APPLICATION OF LCA TO AGRICULTURAL PRODUCTS

1 Core methodological issues
2 Supplement to the *LCA Guide*
3 Methodological background

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Environmental Life Cycle Assessment (LCA) is a method for performing an integral analysis of the environmental impacts of products. It has been investigated to what extent this method is applicable to the field of agriculture. Solutions have been sought to the methodological problems arising when an LCA is carried out on agricultural products. The results are reported in Parts 1, 2 and 3, which can be seen as a supplement to the existing *LCA Guide*. Parts 4a, 4b and 4c (available in Dutch only) describe the experience gained in applying the LCA method to three specific cases: arable farming, dairy farming and bio-energy.

Environmental Life Cycle Assessment/LCA/agriculture and environment

PREFACE TO THE ENGLISH EDITION

This edition is a translation of a report that was published in Dutch in March 1996.

The original edition consists of four volumes: one methodological volume, and three additional volumes describing three different case studies. Only the methodological volume has been translated into English. For this reason, some of the references to the additional volumes have been omitted from this English edition. A list with suggestions 'for further reading', containing mainly publications in Dutch, has also been omitted.

FOREWORD

The Netherlands' 1st National Environmental Policy Plan devotes considerable attention to adopting an integrated approach to environmental problems, based on the consideration that environmental impacts should not be transferred to other environmental media or other links in the product chain. In order to operationalize this policy perspective it is often necessary to inventory and subsequently evaluate environmental impacts 'from the cradle to the grave'. This is particularly relevant in the case of product-oriented environmental policy. It was consequently in this context that the method of environmental Life Cycle Assessment (LCA) was developed. The role and position of this method in product policy is indicated in the government's 'Product and Environment' policy document. Although the method yields an integral assessment of the environmental impact of a given product, it is no substitute for a policy decision. LCA should rather be seen as a policy support tool; it provides a means of charting environmental impacts in a way that can be readily understood, but in the final decision other policy principles and interests must also be taken into account.

Over the past few years the LCA method has been regularly applied to non-agricultural products. Within industry, particularly, LCA is employed for the purpose of environmentally oriented product development, but in the field of environmental policy, too, LCA is a useful tool for structuring the consequences of a proposed policy: for arguing the choice for using one-way or returnable packaging, for example. Another important area of application is the development of criteria for eco-labelling schemes.

To date the LCA method has not often been used for agricultural products. Because the standard LCA method had not been designed for this purpose, in practical applications in this area a number of a inadequacies and bottlenecks were encountered. In practice, debate on several major principles hampered application of LCA, although integral assessment of all environmental impacts is an important facet of agricultural policy and LCA is viewed as a suitable instrument for this purpose.

The Dutch Ministries of Agriculture, Nature Management & Fisheries and Housing, Spatial Planning & Environment therefore commissioned the present study, the objective of which is to render the LCA method more appropriate for agricultural applications. A uniform method for assessing environmental impacts from the cradle to the grave is essential for agriculture, too. On the basis of three cases the researchers have prepared an 'agricultural' supplement to the standard *LCA Guide*.

With the publication of this supplement to the method we anticipate that LCA will be used more frequently in the agricultural sector. Possible areas of application include improving the environmental performance of agricultural products, underpinning eco-labelling criteria and assessing the environmental impact of 'agrification'. In the case of agriculture, too, LCA should be seen solely as a tool for policy support, rather than as a substitute for decision-making. This agricultural supplement will be incorporated in the standard *LCA Guide* as soon as the 'standard' guide is updated in 1997.

The close involvement of the guidance committee in the substance of this project has clearly increased the support base of the results. As project promoters we are very grateful to the committee for their contribution.

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CONTENTS

INTRODUCTION

Dutch agriculture causes a wide range of environmental problems. The government, the agricultural sector, industry and consumers each have their own perspective on these problems. In discussions about the environmental impact of agricultural products horizons are often limited, however. In many cases the discussion revolves around only part of the production chain, with no account being taken of how environmental impacts are passed on to other links in the chain. The number of environmental 'themes' considered is also often limited.

Environmental life cycle assessment

Environmental life cycle assessment (LCA) is a method for performing an integral analysis of the environmental impacts of products. To this end all the environmental impacts of a product or service are charted, 'from the cradle to the grave'.

The LCA method comprises five successive phases. In the *goal definition* it is established with what aim and for whom the LCA study is to be carried out. A precise description is also drawn up of the economic product to be investigated. In the *inventory analysis* a so-called process tree is then drawn up comprising all the processes that together make up the life cycle of the product. The environmental impacts of each of these processes are investigated – in other words, the emissions to and extractions from the environment due to human action. In the *classification and characterization* phase these environmental interventions are then assessed for their potential environmental impacts. In the *evaluation* phase an overall pronouncement is made on the (potential) environmental impacts of the product. In an *improvement analysis*, finally, the scope for improving the overall environmental performance of the product investigated is considered.

Heijungs *et al.* (1992) have written a guide for performing LCAs (subsequently referred to as the *LCA Guide*). In the Netherlands as well as elsewhere this is currently being used a standard. Readers unfamiliar with the LCA methodology are referred to *Beginning LCA. A guide into environmental Life Cycle Assessment* (Van den Berg *et al.*, 1995).

Aim of this study

The LCA method was originally developed for industrial products. If the method were to be applied to agricultural products without due consideration to the specific characteristics of agriculture, there would be a risk of erroneous impressions being gained. The aim of the present study is therefore to investigate the extent to which the LCA method is suitable for use in the agricultural context. Solutions have been sought to the methodological problems encountered in performing an LCA for agricultural products. The results are recommendations for those interested in undertaking an LCA for agricultural products and can be seen as a supplement to the existing *LCA Guide*.

The study is a joint project carried out by three Dutch institutes: the Centre of Environmental Science of Leiden University (CML), the Centre of Agriculture and Environment (CLM) and the Agricultural-Economic Institute (LEI-DLO). The study has been funded by the Dutch Ministries of Agriculture, Nature Management & Fisheries and Housing, Spatial Planning & Environment.

Method
The basic approach of the study has been to base methodological development on experiences gained in a number of case studies. In addition, specialists were consulted in the fields of agriculture and environment as well as LCA. Methodological problems and possible solutions were also put to participants in three workshops.

The case studies fulfilled three functions:
1) To identify problems: what problems are encountered in applying LCA to agricultural products?
2) To assess the merits of potential solutions to these problems in the context of the specific cases. The case studies thus served to support methodological development of LCA and were explicitly not designed to yield any precise insight into the environmental impacts of the products investigated.
3) To serve as a guideline for those considering undertaking an LCA study. The case studies describe precisely the choices made, the bottlenecks encountered and the information gathered.

Given the status of the cases, there has been no peer review of the case studies.

Three cases have been elaborated, *viz.*:
- arable farming;
- dairy farming;
- bio-energy.

Together, these cases provide a good impression of agricultural production in various soil-related sectors. Moreover, the cases have been chosen such that the problems that may be encountered by those implementing an LCA study on agricultural products are covered as comprehensively as possible.

In resolving the methodological problems encountered in the course of the present study we have followed a number of different paths. For some problems the best solution could be chosen by *argumentation*. A case in point is the decision, in considering the deposition of minerals, to take only the technically available fraction as environmental input. Another example is the decision to consider the soil as part of the environmental system, so as to include substance toxicity to soil organisms. For other methodological problems decisions were made partly on the basis of the *experience* gained during the case studies. In each individual case we worked with different types of data, so as to gain an idea of the pros and cons of different data sources. Another example is the decision to work with a mineral balance. The approach formulated in this report is based in part on the practical experience gained during work on the case studies. Finally, we have subjected a number of methodological problems to a *sensitivity analysis*. These were problems for which there are a variety of (apparently) good solutions. The sensitivity analyses indicated the effect of choosing one or other of these solutions, permitting a practical choice of the 'best' solution. These analyses were carried out as part of the case studies. Sensitivity analyses were carried out for five methodological problems:

1) delimitation of the process tree (dairy farming case study);
2) allocation in the case of co-production (bio-energy case study);
3) allocation in the case of recycling (dairy farming case study);
4) the impact of measures in one cropping system on other cropping systems (arable farming case study);
5) incorporation of a time dimension in the 'classification and characterization' phase (arable farming case study).

Report structure
The report on the study entitled 'Application of LCA to agricultural products' consists of four parts, with the following structure.

Part 1, *Core methodological issues*, provides a brief summary of the most essential additions to the *LCA Guide* (Heijungs *et al.*, 1992). This part is intended for the reader interested in obtaining a quick impression of the key points on which the methodology of the original *LCA Guide* has been supplemented.

Part 2, *Supplement to the LCA Guide*, provides a more extensive description of the additions to the original *LCA Guide*. This part of the report is structured in largely the same way as the *LCA Guide*. Because the additions relate exclusively to three of the five chapters of the original Guide, the present document contains only three chapters: goal definition, inventory analysis and classification & characterization. The section structure is slightly different from that of the original Guide, since some of the original sections had to be supplemented to such a degree that without a certain subdivision clarity would have been lost, while other sections required no specific agricultural additions at all and consequently do not appear in the present report. In each section specific mention is made of the section of the *LCA Guide* to which the section refers.

Part 3, *Methodological background*, considers in greater detail the motivation behind the methodological choices made and is aimed primarily at readers interested in the arguments and sources used in arriving at these choices.
Parts 1, 2 and 3 have been published as a single volume.

Parts 4a, b and c (three volumes, available in Dutch only) describe the cases that have been elaborated. Part 4a is concerned with arable farming, Part 4b with dairy farming and Part 4c with bio-energy. As far as possible these cases have been elaborated as LCA studies, but their prime purpose is to support and illustrate the methodological choices made. For example, they do not contain extensive tables of data, as is usual in LCA reports, for this study is not concerned with data and concrete results, but with methods and experience. However, the report structure follows that of the existing *LCA Guide* as far as possible.
Parts 4a, b and c will be of particular interest to (future) practioners of LCA for agricultural products.

1: Core methodological issues

CORE METHODOLOGICAL ISSUES

The *LCA Guide* (Heijungs et al., 1992) can be used for performing an LCA on agricultural products. On a number of issues the existing method needs to be supplemented, however. The present study investigates the problems occurring if the *LCA Guide* is employed for agricultural products. For a number of problems solutions are provided. This brief summary describes the main problems and our proposed solutions. The following issues – arranged according to the steps of the LCA method – are discussed:

GENERAL
- choice of method;

GOAL DEFINITION
- the functional unit for foodstuffs;

INVENTORY ANALYSIS
- the boundary between the economic system and the environmental system;
- delineating the process tree, *inter alia* in relation to capital goods;
- substance flows to and from the soil;
- the choice of data sources in relation to the goal of the study;
- nutrient emissions in relation to the extent to which nutrification constitutes a problem in the areas in which these emissions occur;
- groundwater abstraction in relation to desiccation;
- choice of data sources in relation to the goal of the study;
- allocation;
- crop rotation;

CLASSIFICATION & CHARACTERIZATION
- environmental transport and degradation of substances.

Identified problems for which no solution has yet been found
In the course of the study two types of environmental interventions were identified for which we were unable to find an unambiguous way of including these interventions in the scope of an LCA:
- ecosystem-degrading interventions;
- physical degradation of the soil.

In Part 3 of this report, *Methodological background*, a summary is provided of the discussions that have taken place in the framework of this study within the project team and during the workshops.

Choice of method
The LCA method yields a location-independent, generic analysis of the potential environmental impacts associated with a product or product system. The generic aspect is encompassed in the fact that not one specific product but a product in its generality is analysed. In the case of an LCA study on wheat, for example, analysis is based not on a single grain of wheat from one specific plant cultivated by one specific farmer, but on production of, say, 1000 kg of wheat using a given cultivation method.

With LCA all the environmental impacts occurring during the overall life cycle of a product can be integrally analysed. One limitation of this approach is that the analysis reports only on the *potential contribution* of a product system to a given environmental problem and not on its actual environmental impact, round a specific emission point, for example. LCA is characterized by having a *location-independent design*: in other words, an environment is assumed that comprises homogeneous media. In the agricultural setting, however, differences in local conditions, such as differences in soil type and climatology, may have a major influence on the environmental impacts resulting from a given emission. Two of the problems described below relate to the location-independent design of the current LCA method. Not surprisingly perhaps, two of the solutions proposed by our research team are in the direction of a more location-dependent approach. This said, though, LCA remains a location-independent assessment tool. For this reason LCA is <u>not</u> the appropriate instrument for analyses whereby allowance must be made for actual environmental impacts at a specific site, for example when a water board wishes to gain an idea of the changes in environmental impact in a given area of their operations as a result of a change in cropping methods for a given crop.

LCA should therefore be employed only for a generic analysis of the environmental impacts of a product or product system, and <u>not</u> for analysis of local environmental problems.

1. GOAL DEFINITION

The functional unit for foodstuffs (*LCA Guide* § 1.3.4)
Defining the functional unit for foodstuffs is a matter requiring particular attention. The main reason for this is that providing humans with nutrients is not the sole function of foodstuffs. Foodstuffs also fulfil an important practical, psychological and social function. This is particularly true in the industrialized world, since people there actually consume more than sufficient nutrients. The basic point of departure in comparing food products is *real substitution*.

2. INVENTORY ANALYSIS

The boundary between the economic system and the environment system (*LCA Guide* § 2.1.1)
In the agricultural sector the main function of the soil is economic. It is consequently often seen as part of the agricultural production system. In the standard LCA method, however, all environmental media, including the soil, are considered part of the environmental system. Both definitions can be backed up by arguments and be practically elaborated within LCA. However, a choice must be made between these two approaches. A similar problem arises with agricultural products: like the soil these can be viewed, in part or in their entirety, as part of the economic system or the environmental system.

In this project the following choices have been made:

- The soil should be considered as part of the environmental system. The main argument for this choice is that we wish to view damage to the soil as an environmental impact, because it should be possible to distinguish between systems that differ in their degree of damage to the soil. Substance flows to and from the soil are consequently considered to be environmental inputs and outputs. This does not imply any judgment as to the acceptability of these flows, merely that these emissions should be included in the LCA.
- The harvested portion of the crop forms the economic output of arable farming. The substances contained herein, such as phosphorus and nitrogen, make no contribution to such environmental problems as nutrification. The harvestable part of the crop (grains of wheat or beetroots, for example) is therefore considered to be part of the economic system, while the rest of the crop is taken to be part of the environmental system.

Exception
An exception to these choices can be made in the case of forms of *greenhouse horticulture* whereby natural soil is not used for production. In such cases the whole production system – including the entire crop – is taken to be part of the economic system. The soil obviously remains part of the environmental system.

Deliniation of the process tree and capital goods (*LCA Guide* § 2.1.2)
In order to avoid the problem of infinite regression, in an LCA the process tree must be cut off at various points. Only processes that contribute scarcely if at all to the environmental interventions associated with the functional unit can be omitted. A basic point of departure is that all processes on which information can be readily obtained should be included. Figure 2.1 of the *Supplement to the LCA Guide* (Part 2 of this report) summarizes the processes that may on no account be omitted in an LCA for agricultural products. For the other processes the choice for or against inclusion can be made on the basis of their anticipated contribution to the environmental interventions. In estimating the anticipated contribution a number of criteria can be used:
- the amount of mass involved in the process;
- the amount of energy involved in the process;
- the integral cost price of the process.

These three criteria each have their pros and cons and should therefore preferably be used in combination.

In practice capital goods are often left out of consideration in an LCA, because their contribution to the aggregate environmental score of a product is deemed negligible and there is insufficient time to evaluate this contribution. Another reason for omitting capital goods from an LCA is that there is frequently little difference in their use between two comparable product systems. If per-hectare yields differ, however, this assumption in wrong. In agricultural activities a number of capital goods are employed with a relatively short service life (e.g. farm machinery) and a number of capital goods that require relatively large amounts of material (e.g. farm tracks and roads). In both cases their omission may have a relatively large influence on the final result. The environmental interventions associated with the *production and maintenance of machinery* can therefore not be omitted in an LCA for agricultural products. Likewise, the contribution of *farm tracks and roads* cannot simply be left out of consideration. *Farm buildings* can generally be omitted from the study, except in the case of greenhouse horticulture and in studies whereby farm buildings constitute the main issue.

Substance flows to and from the soil (*LCA Guide* §2.2.1 and § 3.1)
As indicated above, we have opted to take the soil as part of the environmental system. This means that all inputs to the soil (fertilizer and manure) should in principle be classified as nutrifying. Besides emissions of minerals and other substances to the soil, agriculture also involves extraction of these substances from the soil. In the inventory phase of an LCA the quantity of a substance extracted from the soil should therefore be subtracted from the quantity emitted to the soil, because the former has no environmental impact. This is particularly relevant for substances present in fertilizer dressings.

In order to determine what fraction of the mineral supplements ends up in the environment, use is made of a *soil mineral balance*. This can be used to calculate the emission, by subtracting all outputs from all inputs. For annual crops (e.g. most arable crops) the long-term equilibrium situation can be taken as the point of departure; in other words, the assumption is made that, on balance, there is no accumulation of nitrogen. The excess nitrogen is then divided over volatilization, run-off, leaching and denitrification. In the case of perennial crops such as grass the assumption of no long-term accumulation of nitrogen is no longer valid, however, and accumulation should be included in the balance. For phosphorus a similar but simpler balance can be drawn up. In the case of phosphorus, there is accumulation in the soil. The excess phosphorus is divided over leaching, run-off and accumulation.

A similar balance should also be drawn up for other substances added to the soil, such as heavy metals.

Contrary to the approach taken in the *LCA Guide*, in the case of nutrient accumulation in the soil a distinction is made between *problem areas* – where nutrification constitutes a problem (large parts of Western Europe) – and *non-problem areas* – where nutrification does not form a problem (virtually the entire Third World, where soil exhaustion is the problem). In areas where nutrification is not a problem the accumulation of minerals in the soil is not classified as a nutrifying emission. This means that in the inventory phase a distinction must already be made between areas where nutrification is a problem and areas where it is not, on the basis of the location of the emission. Emissions of (soil-supplement) minerals to other environmental media, via run-off, leaching and volatilization, for example, are classified as nutrifying, because these emissions can lead to nutrification of surface waters or of areas in the vicinity of the non-problem area. If it is unknown whether nutrification constitutes a problem in a given area, all emissions should be regarded as nutrifying.

Exception
In this case, too, an exception can be made for soil-free production, such as substrate cropping systems. A soil mineral balance is then superfluous and the procedure of the general *LCA Guide* can be followed.

Groundwater abstraction and desiccation (*LCA Guide* § 2.2.1 and 3.2)
As it now stands, the LCA method gives no consideration to desiccation, because this is considered to be a local problem. In agriculture, however, desiccation does constitute a major environmental problem. In the problem of desiccation, drainage, watertable management and groundwater abstraction are key determining factors. The first two aspects are highly location-specific and are difficult to relate to a functional unit of product. These aspects can therefore not yet be included in an LCA. Desiccation should consequently only be included if it is governed largely by the direct or indirect withdrawal of groundwater in the area in question. As in the case of nutrification, then, with desiccation due allowance should be made for the difference between problem and non-problem areas. Groundwater abstraction in areas where there is no desiccation problem or in areas where groundwater abstraction does not contribute to this problem (for example, in the situation of surface water levels being kept artificially low, thus determining the degree of desiccation) should not be classified as desiccating. If it is unknown whether a groundwater-abstraction process contributes to desiccation, groundwater abstraction should be classified as desiccating.

Choice of data sources in relation to the goal of the study (*LCA Guide* § 2.2.2)
The agricultural sector comprises a large number of individual farm enterprises, no two of which are identical. This means that for direct agricultural processes it is very important, depending on the goal of the study, to consciously opt for *average data, normative or representative data, or data on individual farms*. With each of these choices, but particularly in the case of average data, due allowance must be made for the spread of the results due to the spread of the raw data. Examples of a substantiated choice of data sources include:

- If the goal of the study is to obtain an idea of the environmental impacts associated with milk sold in supermarkets, use can be made of average data on milk production.
- If various different current milk-production methods are being compared, use can be made of average data on the companies applying the various production methods.

- A milk producer who wishes to know which elements of his product (system) have the greatest bearing on the environmental impacts he causes will obviously choose data specific to his own product (system).
- If the government wishes to use LCA to back up a policy to encourage or discourage a given production method, use can be made of normative data that are specific to companies applying the production method in question.

There are various data sources that can be used for an LCA. For data on agricultural production in the Netherlands, for example, use can be made of the LEI agricultural database. A drawback of this database is that it contains data on individual farms only, while in the context of LCA process-level data offer the greatest analytical potential. If data on individual farms are employed, problems arise if any attempt is made to derive the contribution of individual processes to the environmental profile. This is particularly a problem if the functional unit relates to only part of the farm's operations and is linked to a number of specific processes.

Allocation (*LCA Guide* § 2.3)
The *LCA Guide* explains how environmental impacts should be divided over the various inputs and outputs of a process. In agriculture, however, a number of specific cases of co-production and recycling are encountered that require further elaboration. In addition, attention should be paid to the possibility of avoiding allocation by applying the so-called *substitution method*. This method can be used if one of the products of a multi-output process can also be obtained via an alternative process, of which it is the sole output. The environmental impacts of this alternative process are then deducted from those of the multi-output process.

Co-production is common in agriculture. The various parts of the animals and plants produced are often used for different applications. Before allocation is undertaken, it must first be clear that multi-output processes have as far as possible been divided into single-output processes. Only for those processes that cannot be further subdivided should allocation be carried out, and this should be done on the basis of social causality. In other words, the value to society, generally expressed in terms of the turnover (= market value times yield (in units of mass)) of all outputs involved forms the basis for dividing environmental interventions and economic inputs (e.g. electrical power or animal feed) over the various outputs: if one of the two co-products is responsible for 90% of turnover, this product will also be allocated 90% of the environmental impacts.

If *manure* is used in arable farming, recycling is taking place and the environmental interventions associated with the processes involved (storage, transport, processing) should be allocated to the product system that pays for these processes. If payment is collective (e.g. in the case of storage in a manure centre) interventions should be allocated on the basis of the ratio between the cost paid by the arable farmer and the cost paid by the cattle farmer. Again, these rules are based on social causality.

Crop rotation (*LCA Guide* § 2.3)
Agricultural crops are frequently cultivated in a system of crop rotation, with different crops being cultivated in succession on a given plot of land. If a comparison is being made between different crop-rotation schemes, this will cause no extra allocation problems. In practice, though, such a comparison will not often be useful, for LCA is a tool designed for comparing the environmental impacts of various different products. What will most frequently be compared are a product from one crop-rotation scheme and one from another scheme. This gives rise to difficulties, because the various crops and the activities performed in cultivating these crops often also have consequences for the crops grown later in the rotation scheme. Examples include:

- soil fumigation carried out for potatoes, but also benefiting other crops;

– application of organic fertilizers in a given crop, with some fraction of the minerals only being taken up after the following crop has been sown.

These allocation problems cannot simply be ignored in an LCA. The basic point of departure in allocation is: *'Why is a given activity performed?'*. For example: the soil fumigants applied in potato cultivation would not be used if potatoes were not included in the crop-rotation scheme. The environmental interventions associated with the soil fumigants should therefore be allocated entirely to the potatoes, even if benefits accrue to other crops, too. On these grounds, in the case of application of nitrogen fertilizer the associated environmental interventions are allocated to the crop to which the fertilizer dressing is applied, while the environmental interventions associated with application of *phosphate* and *potassium* are divided over the crops on the basis of the recommended dressings for each individual crop. *Organic matter* is allocated on the basis of the share of the various crops in the crop rotation scheme (expressed in terms of space requirements: ha·y). When multiple fertilizers (manure and other animal wastes, in particular) are applied, the emissions occurring up until the moment the minerals reach the soil (emissions during storage, transport and application) are divided over the various crops on the basis of the economic value of the minerals in the fertilizers.

Given the importance of the crop-rotation scheme for further choices within an LCA, it is of major importance that the crop-rotation scheme being used to cultivate the products in question already be indicated in the *goal definition*.

3. CLASSIFICATION & CHARACTERIZATION

Environmental transport and degradation of substances (*LCA Guide* § 3.2)
In the current method used for the *characterization* of toxic substances no allowance is made for environmental *transport and degradation* of these substances, while in the case of various agricultural pesticides these processes may be of decisive importance for the degree to which the toxic potential of these substances leads to potential environmental impacts. For the most commonly used agricultural pesticides new equivalency factors have therefore been developed for the toxicity themes, incorporating intermedia transport and degradation. These equivalency factors have been calculated with the aid of the USES (Uniform System for the Evaluation of Substances) model (RIVM, VROM, WVC, 1994). The new equivalency factors cannot be compared with the factors in the *LCA Guide*, and the scores calculated for pesticides using these factors cannot therefore be added to the scores for other substances, but should be listed separately. In Part 4a of this report (in Dutch) a number of old and new equivalency factors are compared.

2: Supplement to the LCA Guide

1 GOAL DEFINITION

In the goal-definition phase of an LCA the precise goal of the study is described. In basic terms, goal definition involves deciding on the following three elements:
- the application and target group of the study (§ 1.1);
- the depth of the study;
- the subject of the study.

The *application* is generally decided on jointly by the parties commissioning and executing the study prior to commencement of the study. If an LCA study is to be performed successfully, it is important to state in advance what the exact *objective* of the study is and for which *target group* the study is being undertaken. Readers of the final report will also be interested in the identity of the *initiator* of the study. These formal issues should be clearly described in the report, so that readers can place the study in its proper context.

The *depth* of the study is related to the goal and the available time. If the main interest is in the overall picture, or if there is little time, parties may opt for a simplified approach, for example by leaving certain processes or certain environmental impacts out of consideration. Such simplifications should be clearly reported, to avoid readers gaining a wrong impression of the significance of the final results.

The *subject* of the study is of crucial importance. The product that is to be evaluated should be described in detail. Items to be reported on are the *product group* to which the products to be evaluated belong, the *spatial and temporal representativeness* of the products, the *functional unit* (§ 1.2) used as the basis for evaluation and the *products* that are to be evaluated.

Basis: *LCA Guide*[1], *Chapter 1*

Additions: 1.1 The application of the study (addition to § 1.1 of the *LCA Guide*)
1.2 The functional unit (addition to § 1.3.4 of the *LCA Guide*)

1.1 The application of the study

The *LCA Guide* distinguishes three aspects of determining the application of an LCA:
- determining the type of application;
- selecting the target group(s);
- listing the parties involved.

This document furthermore distinguishes a fourth aspect:

[1] The 'LCA Guide' referred to in the present report is the document 'Environmental Life Cycle Assessment of Products - Guide and Backgrounds' (Heijungs *et al.*, 1992).

- choice of instrument.

These choices should preferably be made after the type of application has been precisely established.

Basis: *LCA Guide*, § 1.1

Additions:

1.1.1 Determining the type of application (addition to § 1.1.1 of the *LCA Guide*)

1.1.2 Choice of instrument

1.1.3 Relationship between application, target group and data (addition to § 1.1.2 of the *LCA Guide*)

1.1.1 Determining the type of application

Guinée (1995) distinguishes a number of possible applications of LCA in general. In principle, each of these applications is relevant for agricultural products. In Table 1.1 each of the applications mentioned by Guinée is linked to one or more examples from the field of agriculture.

TABLE 1.1 Possible applications of LCA for agricultural products

general	examples in agriculture
improving certain environmental aspects of existing products	reducing the environmental impact of a given cropping system
designing new products	producing fuel from agricultural products; designing new types of livestock sheds
providing environment-oriented product information	providing environmental information on a given product (e.g. wheat) to companies using this product as a raw material (e.g. industrial bakeries)
comparing the environmental impacts of functionally comparable products	comparing tomatoes grown on a substrate with tomatoes grown outdoors
awarding eco-labels	using LCA (or parts thereof) in the procedure for awarding (agro)eco-labels to agricultural products
creating a policy basis for the approval of new products/technologies	designing environmental criteria for granular fertilizer produced from manure
evaluating the environmental aspects of policy strategies	evaluating (elements of) current EU agricultural policy relative to alternative policy; evaluating the environmental impact of introducing an energy tax

1.1.2 Choice of instrument

Once the type of application has been determined, it is important to establish whether LCA is the most appropriate tool for answering the specific research question, or whether another instrument is perhaps more suitable for the purpose. This choice can be made on the basis of the potential and the limitations of LCA in relation to the potential and limitations of other tools for environmental assessment.

LCA yields information on the environmental impacts of a product. This information can be used for decision-making by government agencies, companies and consumers and thus constitutes a decision-support tool.

LCA has been developed for evaluating the environmental burden of *products* fulfilling a given function. LCA is frequently employed for comparing two or more product alternatives with the same function, for example a tomato grown on a substrate system with a tomato grown outdoors. The objective of LCA is an *integral analysis of all the environmental impacts occurring during the entire life cycle ('from the cradle to the grave') of a product or service.*

As a tool for supporting decision-making, the LCA method has a number of important features:

- LCA enables the extremely complex and extensive data set required for the environmental assessment of a product to be compressed into scores for a limited number of environmental themes, greatly simplifying the assessment.
- LCA ties up well with the concept of integrated substance chain analysis; in agriculture, for example, it is not only the environmental impacts of cultivation of a given crop that are important but also, say, the environmental impacts associated with the production of the fertilizer used on the crop, as well the environmental impacts of processing the waste produced during crop cultivation.
- In principle, virtually every possible environmental intervention can be included in the assessment.

The LCA method also has its limitations:

- In the LCA method the emissions for which a product is responsible are assessed solely on the basis of quantity and *not on the basis of concentration* in the environment. Because the actual impacts caused by an emission do depend on concentration, these can *not* be assessed using LCA. In LCA the *potential environmental impacts* of an emission are determined; for a given emittant these are considered to be proportional to the magnitude of the emission, summed over the entire life cycle.
- In LCA emissions of one and the same substance within various different processes of the life cycle of a product, which may take place at very different locations, are summed *unweighted*. This means that assessment by means of LCA is *location-independent*.
- Because LCA makes an assessment that is independent of location and concentration, this instrument can *not* be used to assess whether (local) environmental standards are exceeded, for example the standards in force for concentrations of substances in the surface water around a given plot of land.
- LCA proceeds from *linear processes*, in other words the environmental interventions associated with production of 2 kg of steel are assumed to have twice the magnitude of those associated with production of 1 kg of steel, which is obviously a simplification of reality: in practice, the first kilogram produced will have a greater or lesser environmental impact than the thousandth. In addition, *within LCA itself* there is no scope for determining the influence of, say, changes in process conditions on the magnitude and nature of the environmental interventions.
- An LCA gives an impression of the differences between the environmental impacts of two different product systems in which one functional unit of product is produced. The fact that the magnitude of the quantity of product to be assessed (the functional unit) can be arbitrarily chosen means that assessment, by definition, relates to marginal changes, implying that the relationships between emissions/extractions and environmental impacts are assumed to be linear. With an LCA a better understanding can be obtained of the environmental impacts of production of one extra functional unit in current or, possibly, imaginary production processes.
- An LCA considers a *single delineated product system*. For example, a comparison is made between two different product systems used for producing 1 kg of tomatoes, with the rest of the world assumed to remain constant. If the entire Dutch horticultural sector were to switch from one product system to the other, this assumption would no longer hold, and other effects would also

occur. For example: assume that one of the product systems uses manure and the other artificial fertilizer. In an LCA study the system using manure would probably score worse on the theme of *nutrification* than the system using artificial fertilizer, because of the mineral losses in the former case (per kg tomatoes). For the Netherlands as a whole, however, there may be less loss of nutrients if tomatoes are grown using manure, because of a reduction in the aggregate input of nutrients in the Netherlands following a reduction in the use of artificial fertilizer.

Besides LCA, there are also other analytical policy-support tools, each with its own specific objective and scope. The main instruments are listed below, accompanied by a short description.

- For assessing the risks associated with a given situation, *risk analysis* is the most appropriate instrument. It is used to evaluate actual, concentration-dependent impacts. The question of whether certain threshold values, for example no-effect concentrations (NEC-values), are being exceeded can be answered with the aid of risk analysis.
- For assessing the most suitable location for a certain economic activity from the environmental point of view, *environmental impact assessment (EIA)* has been developed. This approach focuses above all on the location-specific environmental impacts, although LCA-like methodologies are being employed with increasing frequency to incorporate the rest of the chain in the final evaluation.
- For assessing the environmental impact of a specific economic activity, the *environmental audit* has been developed. This approach focuses on physical aspects such as the substance flows and emissions issuing from an enterprise, but also frequently on such aspects as the degree of environmentally oriented training undertaken within the enterprise and working conditions.
- For comparing two *simple processes*, the environmental impacts of which are anticipated to be negligible in other parts of the chain (or in differences between these), a process-technology approach is the most appropriate, since it can make due allowance for non-linearities, for example the consequences of changes in process conditions.
- If the direct environmental impacts caused by *an individual substance in general* over a certain period of time (for example, 'the annual environmental impacts of nitrogen in the Netherlands') are to be analysed, *substance flow analysis (SFA)* is the most suitable instrument (e.g. Van der Voet, 1996).

The above does not mean that certain elements of the LCA method cannot be incorporated in the other instruments. In particular, the equivalency factors – with which hundreds of emissions and extractions can be aggregated to some twenty environmental impacts – may also be of interest outside the LCA context. LCA can also be used alongside other instruments if decisions are to be made that can be tackled according to several different lines of approach.

1.1.3 Relationship between application, target group and data

In executing an LCA on agricultural products a number of problems are encountered that are connected with the production level to which the data to be gathered should relate. Compared with many other economic activities, agricultural production encompasses a relatively large number of production units (in this case: farms). The production processes – and consequently the associated environmental interventions, too – may differ markedly from farm to farm. Because the environmental impacts of the production of, say, one litre of milk may differ so much from farm to farm, in comparing two different methods of milk production (A and B) it may be important to obtain an impression of the spread in scores among the various farms. Assume that on average milk A scores slightly better than milk B and that the spread in data is very wide. In that case it may be the case that the worst-scoring farms producing milk A score worse than the best-scoring farms producing milk B.

The results of an LCA are thus highly dependent on the choice of data sources. In this context the level of the data sources is of crucial importance: a choice can be made from among data on an individual farm, data on an (imaginary) representative farm, normative data for a given area or a given crop, or an average for a number of (say, Dutch) farms. The data sources used should therefore be chosen after careful deliberation.

Which data source should be chosen is highly dependent on the application of the study, particularly in relation to the target group. This means that the choice of the data sources should link up directly to the *goal definition* of the LCA. In the *LCA Guide* three target groups are distinguished:

- consumers;
- producers;
- government agencies.

Consumers

With the aid of LCA consumers can be informed about the environmental impacts of a certain product, possibly with an eco-label constituting an intermediate stage. This can be achieved by means of information relating directly to the product stocked on the supermarket shelves. One problem in this context is that the milk stocked in the supermarket originates from a large number of different farms, implying a need to perform a separate LCA for each individual farm. In practice, this is obviously unfeasible, and for the intended purpose unnecessary. If the objective of the study is to compare a number of types of milk (dairy farms), it suffices to use LCA to establish the average differences among the corresponding environmental impacts of the various types of milk (dairy farms). In this case, then, use can be made of average data on certain types of farm.

Producers

A producer can be informed as to the major areas of environmental impact due to the products he produces: are these areas to be found within the confines of his own operations, at the sites of his raw materials suppliers, or at those of the processors of his waste? The producer can use this information to reduce the environmental impact of his products. For this purpose use will have to be made largely of data that are specific to the operations of the producer in question.

If the producer wishes to gain an impression of the extent to which the environmental performance (on a variety of environmental themes) of his product/company deviates, in a positive or negative respect, from the average product/company, or from a normative or representative product/company, then data will have to be gathered on the individual product/company, on the one hand, and average, normative or representative data, on the other.

Government agencies

Government policy has an influence on the type of product produced at various levels, as well as on the production method thereby employed. If the government wishes to steer production in a direction causing minimum environmental impact, LCA is one of the tools that are appropriate. An example here is minimization of the environmental impact of horticulture. By means of LCA it can be determined whether, and to what extent, cultivation on artificial substrates should be encouraged. To this end, practical data from operational companies are required, which must then be clustered in one way or another. A complicating factor is that, in drawing up policy, government agencies use not only practical data on the current situation (monitoring) but also future scenarios based on certain policy trends for which certain normative data are required. This means that when LCA is employed to support government policy use will sometimes have to made of a combination of practical and normative data.

1.1.4 Guideline: the application of the study

- Establish the type of application, choosing from the following possibilities:
  1. improving certain environmental aspects of existing products;
  2. designing new products;
  3. providing environment-oriented product information;
  4. comparing the environmental impacts of functionally comparable products;
  5. awarding eco-labels;
  6. creating a policy basis for the approval of new products/technologies;
  7. evaluating the environmental aspects of policy strategies.
- On the basis of the potential and the limitations of the various tools for environmental analysis, verify that LCA is the most appropriate tool for the purpose at hand.
- Follow the guidelines of the *LCA Guide* (§ 1.1) with respect to:
  - the choice of target group or target groups;
  - the listing of those involved in the study.
- On the basis of the choice of application and target group, establish which data sources are suitable for the study at hand. Possibilities include:
  - data on an individual company (farm);
  - normative data (manuals, formulae);
  - data on an (imaginary) representative company (farm);
  - average data for a group of companies (farms), for all Dutch farms, for example.

1.2 The functional unit

If the environmental impacts of two or more products are being compared, it is of major importance to ensure that the basis for comparison is a useful one. The key issue should be the *function*, and not the absolute quantity of product. For example: if two types of wheat (A and B) are to be compared, the protein content may be relevant. Perhaps the environmental impact associated with 1 kg of type A is less than that of 1 kg of type B. If 1 kg of type B has twice the protein content of type A, however, in certain studies (relating to wheat for bread, for example) it may make more sense to compare 1 kg of wheat B with 2 kg of wheat A than to compare the two types of wheat on the basis of mass. A general approach is based not on '1 kg of wheat', but on the function of the wheat. The functional unit might thus be 'the provision of 1000 kg wheat protein'. Obviously, it must then be reported that what are being assessed are 'wheat A' and 'wheat B' from the product group 'wheat'. The functional unit is the basis of comparison, however, and the function is the key issue at stake.

Basis: *LCA Guide*, § 1.3.4

Additions: 1.2.1 The basis of the functional unit: mass or land area?
1.2.2 The functional unit for foodstuffs

1.2.1 The basis of the functional unit: mass or land area?

If two different cultivation methods are being compared, the question may arise whether the functional unit may also be taken to be a unit of area (1 ha ecological wheat cultivation, for example) instead of the usual mass unit of product (1 kg of wheat, for example). In an LCA the environmental interventions accompanying all the relevant processes, occurring at any location in the world, are summed and evaluated. This means that location-specific impacts and concentrations play no role and are not reflected in an LCA. If an LCA study in which a hectare has been taken as the functional unit indicates that cultivation method A scores better than method B, this does not therefore mean that method A is also necessarily better for the immediate surroundings than B. Method B might well be better for the local environment, with method A scoring far better on the environmental aspects relating to the supply side (production of raw materials and so on). This is a strong argument against choosing a unit of land area as the functional unit. If an area unit is nonetheless taken, in addition to spatial requirements temporal requirements should also be incorporated in the functional unit, because the environmental impact associated with use of the land is proportional not only to the area of land but also to the period of time during which the land is used for the purpose concerned. In other words, not a hectare but a hectare·year should in this case be taken as the functional unit.

1.2.2 The functional unit for foodstuffs

Foodstuffs often fulfil more than one function: they are frequently a source of nutrition and a source of enjoyment at one and the same time. In establishing the functional unit of a product with more than one function, the *objective* of the assessment is of decisive importance. The function on which the functional unit is based should stand in direct relation to the assessment objective. In comparing products, *real substitution* should be the issue at stake: the quantity of product the size of one functional unit of the one product alternative should constitute a real substitute for a quantity the size of one functional unit of the other product alternative. In this context quality aspects may also play a role, such as protein content (in cereals or milk, for example) and suitability for a given processing technology or form of consumption (the spreadability of margarine, for example). This means that the functional unit should be a *constant factor*: switching from one alternative to the other should have no influence on the quantity used/functional unit required.

Imagine that a comparison is to be made between beef and pork, the question being made concrete in the form of a comparison between steak and Wiener schnitzel. A slice of meat fulfils various different functions at one and the same time: it is a source of calories, a source of protein and a source of enjoyment. Depending on the function considered, the functional unit can be defined in various ways. If the energy content is the central issue, a given quantity of steak will be compared with a quantity of Wiener schnitzel that supplies the same number of calories; if the function of calorie source is the key issue, then quantities with an equal calorie level are compared. If enjoyment is the issue, the basis for comparison is more difficult to establish, because enjoyment is difficult to quantify. However, one can imagine that in each of these three cases a different quantity of Wiener schnitzel will be comparable with a given quantity of steak. The functional unit can thus not be established unambiguously.

One way to overcome this problem is to fall back on the ultimate aim of the study. The aim of comparing beef and pork might be to inform environmentally aware consumers about the relative environmental impact of a number of kinds of meat, for example, to enable them to make a critical choice. In that case, it is important to find out how much Wiener schnitzel a consumer would eat in practice as a substitute for a given quantity of steak. If, for example, it is found in practice that an average portion of steak weighs 100 grams and an average portion of Wiener schnitzel 150 grams, the obvious approach would be to compare a portion of steak (100 grams) with a portion of Wiener schnitzel (150 grams). If in practice meat purchase is based above all on mass, however, a comparison of 100 grams of steak with 100 grams of Wiener schnitzel is the more appropriate choice. They key issue here is to find a *real substitute* for 100 grams of steak. Depending on the question whether ‘a portion’ or ‘a certain mass’ is the *constant factor* in meat purchase, this may be 100 or 150 grams of Wiener schnitzel.

1.2.3 Guideline: the functional unit

- Choose a functional unit formulated as clearly and in as much detail as possible and relating as far as possible to a complete activity (See *LCA Guide*, § 1.3.4.).
- Ensure that *real substitution* is the central issue: a quantity of product the size of one functional unit of the one product alternative should form a real substitute for a quantity of the other product the size of one functional unit. If real substitution is involved, the functional unit will be a *constant factor*: switching from one alternative to the other will have no influence on the quantity used/functional unit required.
- Ensure that the choice of functional unit stands in close relation to the objective of the study.

2 INVENTORY ANALYSIS

In the inventory analysis the life cycle of the product under investigation is analysed, answering the following questions:

- What are the constituent processes of the life cycle?
- How are these processes connected?
- What are the economic inputs and outputs of each process?
- What are the environmental inputs and outputs of each process?

The answers to the first two questions are given in the form of a so-called *process tree*: a schematic summary of all the processes that go to make up the life cycle, in all their interrelationships. The answers to the third and fourth question are given, for each separate process, on a form that is structured according to a standard *format*. The summed inputs (raw materials) from the environment and outputs (emissions) to the environment are represented in the so-called *inventory table*. The inventory table provides an overview of all extractions from the environment and all emissions to the environment for which one functional unit of product is directly or indirectly responsible. Emissions and extractions are together termed 'environmental interventions'.

Sometimes a process from the life cycle is highly interrelated with other processes lying outside the life cycle. This is the case, for example, with co-production of two products in a combined (economic) process, such as the combined production of wheat and straw. In such cases it is not clear in advance for what fraction of the environmental interventions of the combined process each of the two individual products is responsible. There are rules for determining how these environmental interventions should be distributed: the so-called *allocation rules*. For each process a decision must be made as to which allocation rule is deemed most suitable for the process in question. Because the life cycle of a product generally comprises a very large number of processes, virtually every LCA study involves allocation problems.

The inventory analysis consists of four steps:

- drawing up the process tree;
- entering the process data;
- applying the allocation rules;
- creating the inventory table.

The process tree is a schematic summary of the life cycle of a functional unit. Together, the processes in a process tree constitute the *product system* of the functional unit. In drawing up a process tree decisions must be made as to which processes are part of the product system and which are not. In doing so, three boundaries must be drawn:

- the boundary between the product system and the environmental system;
- the boundary between processes that are relevant and those that are not;
- the boundary between the product system under consideration and other product systems.

Basis: LCA *Guide*, Chapter 2

Additions:	2.1	Drawing up the process tree (addition to the introduction of § 2.1 of the LCA *Guide*)
	2.2	Delineating the boundary between product system and environmental system (addition to § 2.1.1 of the LCA *Guide*)
	2.3	Delineating the boundary between relevant and non-relevant processes (addition to § 2.1.2 of the LCA *Guide*)
	2.4	Entering the process data (addition to § 2.1.2 of the LCA *Guide*)

2.1 Drawing up the process tree

A process tree is a schematic representation of the (economic) processes that go to make up the life cycle of a product, in all their interrelationships. Proper representation of the process tree is of crucial importance, because of the large number of processes making up the majority of product life cycles: the process tree forms the ultimate check on whether all the processes have been properly included in the analysis. Because it has transpired in the past that the description given in the LCA *Guide* of how the process tree is to be drawn up still raises a number of questions, there here follows a further description.

Basis: LCA *Guide*, § 2.1

Additions:	2.1.1	The structure of economic processes
	2.1.2	The interrelations between processes: the process tree

2.1.1 The structure of economic processes

Economic processes are characterized by *inputs* and *outputs*. A clear distinction between *economic* inputs and outputs, on the one hand, and *environmental* inputs and outputs, on the other, is indispensable in drawing up the process tree.

Environmental inputs consist of raw materials or space claimed from the environment for the purpose of an economic process. Examples include phosphate ore or crude oil (abiotic resources), tropical hardwood (a biotic resource) and the space requirements of the enterprise carrying out the economic process (milk production, for example). Raw materials for a process that are not extracted directly from the environment, but are the product of an agricultural production process (straw or maize, for example) or an (industrial) manufacturing process (fodder cake from crushed oilseed rape, for example) are considered to be *economic inputs*.

Environmental outputs consist of emissions of potentially hazardous substances to the various environmental media (air, water and soil) and radiation, noise, heat and light to the environment as a whole. Victims of disasters are also considered to be 'environmental output'.

Produced waste is not considered to be an emission to the environment, because waste always ends up in the environment only after a certain form of processing. This processing may consist of, say, waste incineration or landfilling. The incineration of waste causes environmental emissions; these emissions should be determined or estimated whenever possible and a list of the environmental outputs from the incineration process included in the process tree. In addition, energy may be generated during waste incineration: an economic output of the incineration process, which should be reported as such. A landfill site is taken to be part of the economic system. This means that landfilled waste as such should not be considered an environmental emission. As in the case of waste incineration, landfilling of waste may lead to environmental emissions (through leaching of hazardous substances to the soil and groundwater, for example). In the case of landfill, too, there may be an economic output, if the landfill gas (methane) is collected, for example.

If waste is generated in a (production) process, this should be reported as 'waste to be processed' under the heading of *economic* outputs. In addition, the waste processing process should be described as a separate process.

The *economic inputs* of a process can be divided into economic inputs that bear a cost and economic inputs that accrue proceeds to the process.

Economic inputs that bear a cost are goods, services, materials and energy supplied by other processes: animal feed, artificial fertilizer or electricity, for example. The *economic inputs* of a process are thus, by definition, the *economic outputs* of other processes (animal feed, fertilizer or electricity production).

Economic inputs that accrue proceeds are waste products from another process which are processed in the process under investigation at a charge (pig slurry in the process 'manure processing', for example). Here, too, the *economic input* 'waste to be processed' in a given process (manure processing, for example) is formed by the *economic output* 'waste to be processed' from another process (pork production, for example).

The *economic outputs* of a process can also be divided into economic outputs that accrue proceeds and those that bear a cost.

Economic outputs that accrue proceeds consist of the goods, services, materials and energy supplied by the process. These include not only the primary products that are the immediate focus of the process in question (wheat and milk, for example), but also intentional and unintentional co-products, to the extent that these have a positive market value (straw and beef, for example).

Economic outputs that bear costs consist of the 'waste to be processed' that is not processed in the process under investigation. Only substances and materials without any market value, the processing of which bears a cost, are considered as waste. 'Waste products' with a positive market value are not therefore considered as 'waste to be processed' but as co-products.

The economic outputs of a process are always the economic inputs of another process.

All the economic inputs can ultimately always be traced back to environmental inputs and outputs. Economic outputs lead ultimately either to environmental outputs or to persistent accumulation of substances and materials in the economic system.

2.1.2 The interrelations between processes: the process tree

The *process tree* is an inverted tree-shaped schematic representing all the economic processes that go to make up the life cycle of the product under investigation. These processes are represented by blocks which are connected by arrows. The blocks represent the processes, while the arrows symbolize the economic inputs and outputs of these processes. The schematic is drawn up such that the economic input of a process is always formed by the economic output of a prior process. Each arrow therefore symbolizes an input as well as an output.

Because the overall process tree of a product usually has a very extensive configuration, it is generally advisable to work with a *summary process tree* and *partial process trees*. The summary process tree reviews the complete life cycle, but for ease of reference clusters of processes are combined under an umbrella heading and shown as a single block. Each of these blocks is later elaborated in a partial process tree. Partial process trees may also include clusters of process under a single heading, which are in turn elaborated in a lower-level partial process tree. In this way it is possible to 'zoom in' on the process clusters in the summary process tree in increasing levels of detail.

As mentioned, processes are characterized by inputs and outputs. An output from a process in the process tree generally forms the input for a subsequent process. An input or output will generally consist of a certain quantity of a certain substance or a certain product. However, a service rendered can also be considered as an input or output. For ease of reference, inputs from the environment (extractions) and outputs to the environment (emissions) are usually omitted.

Processes are always labelled. The names of the inputs and outputs are often omitted in a process tree. In some cases it may be clearer to label these, though, for example if a process has multiple economic outputs.

The process tree often forms the basis for inputting processes in a computer program. This means that the names given to processes should be chosen with great care, avoiding any ambiguity. For example, process clusters should never have the same name as partial processes within such clusters, and processes that are different in practice should always be labelled with different names.

2.1.3 Guideline: drawing up the process tree

- Follow the instructions in the *LCA Guide* (§ 2.1).
- Represent processes as blocks and the inputs and outputs of processes as arrows.
- Ensure that the process tree contains only *economic* processes.
- Ensure that the economic output of a process is always the economic input to a subsequent economic process.
- Ensure that the functional unit forms an output of the entire summary process tree.
- Ensure that all the economic processes are consistently and unambiguously labelled:
  - A partial process tree may not have the same name as the key process within the partial process tree.
  - The name of a partial process tree must correspond with the name of the relevant process cluster in the summary process tree.
  - Similar processes that are distinguished individually may not have the same name ('transport', for example).

2.2 Delineating the boundary between product system and environmental system

When performing an LCA it is extremely important to make a clear distinction between the product system under investigation and the environmental system. The product system consists of all the processes involved directly or indirectly in the production of a given product. The environmental system consists of 'the environment', including all the processes occurring there. The product systems and environmental system are connected via raw materials being extracted by product systems from the environmental system (e.g. crude oil, phosphate ore) and pollutants being emitted by product systems to the environmental system (e.g. CO_2, NO_3^-). In addition, product systems can also influence the environmental system in non-material ways, by noise emissions or space requirements, for example.

There is usually a clear physical boundary between the product system and the environmental system. Sometimes this boundary is less clear-cut, though: although the soil is generally defined as part of the environmental system, in the agricultural sector this is often viewed as part of the agricultural product system. In the agricultural context the boundary line between the product system and the environmental system is relevant with regard to two points: the soil and the crop.

Basis: LCA *Guide*, § 2.1.1

Additions: 2.2.1 Agricultural soil: product system or environmental system?
2.2.2 Agricultural crops: product system or environmental system?

2.2.1 Agricultural soil: product system or environmental system?

In the agricultural context the boundary between the product system and the environmental system is difficult to draw, because the soil fulfils both an economic and an environmental function. Although the soil is part of both systems, in performing an LCA a choice must be made as to whether or not emissions to the soil are to be considered environmental interventions. The boundary between the product system and the environmental system is in fact thus defined: in an LCA there is no possibility of an overlap.

If the soil (or part thereof) in agricultural areas is considered not to constitute part of the environmental system, this implies that emissions of, say, crop protection agents or other polluting substances to the soil are not viewed as environmental interventions. Degradation of soil ecosystems in agricultural areas are then not considered to constitute an environment impact and therefore have no influence on the environmental assessment of the product under investigation, for example in terms of the assessment of the effects of hazardous crop protection agents on soil ecosystems. This is at odds with the generally held view that a reduction in soil loading with crop protection agents should be evaluated as positive for the environment. In performing agricultural LCA's too, then, the soil is viewed as part of the environmental system.

2.2.2 Agricultural crops: product system or environmental system?

Agricultural crops remain for a limited period of time in the environmental system and are then harvested, bringing them into the product system. The question is: which of these two systems do the crops belong to? This can best be assessed with reference to an analysis of the fate of emissions impinging directly or indirectly on the crop.

During their growth crops take up minerals from the soil. The greater part of these minerals have been applied to the soil as fertilizer for the crop. In LCA the application of minerals to the soil is viewed in principle as a nutrifying emission to the soil. In the case of crop fertilizer dressings it is inappropriate to consider the entire dressing of minerals as an emission to the soil: after uptake by the crop, some fraction of the minerals applied to the soil will be removed from the land with the crop harvest, thus losing its nutrifying potential. Only parts of the crop remaining behind on the land after harvest, possibly being ploughed under, can still contribute to the nutrification problem because of the minerals they contain. In the case of minerals, then, the best approach would appear to be to consider the harvestable part of the crop – including the minerals absorbed therein – as part of the product system rather than as part of the environmental system. Emissions of minerals to the soil are thus considered to be environmental emissions only to the extent that they are not taken up by the harvestable part of the crop. A cradle-to-grave analysis that also includes the product's consumption phase will show that part of the minerals present in the harvested crop will ultimately end up in the environment, by way of the sewerage system. With modern sewage treatment technology, however, a major fraction of the fixed nitrogen will be denitrified to non-nutrifying N_2.

In addition to minerals from the soil, plants also absorb carbon dioxide from the atmosphere. If the harvestable part of the crop is viewed as part of the product system, the fixation of carbon dioxide in this part of the crop can be seen as a negative emission. If one considers the entire life cycle, however, there is no nett carbon dioxide uptake because the vegetable products are ultimately converted back to carbon dioxide, by way of digestion, sewage treatment and environmental degradation. It is therefore proposed not to include carbon dioxide in the analysis if the entire life cycle is being analysed. If a so-called 'cradle-to-gate' analysis (an analysis of products up the moment they leave the farmyard gate) is being performed, though, this fixation must either be included, or it must be explicitly stated that this fixation is being excluded from the study. If this is not done, there is a danger that if other researchers use the results of the study they will include, say, the emission of CO_2 during combustion of bio-diesel fuel, while the fixation of CO_2 was omitted in the cradle-to-gate analysis.

2.2.3 Guideline: the boundary between product system and environmental system

- Follow the instructions in the *LCA Guide* (§ 2.1.1):
  - Virtually every activity to which costs are attached is an economic process.
  - Waste that is yet to be processed (such as flue gases and sewage water) is assigned to the product system, as are waste processing plants (such as flue-gas treatment units and sewage treatment plants).
  - Treatment steps occurring after a substance has entered the environment do not constitute part of the product system causing the original emission.
  - Landfilling of waste is an economic process.
  - Processes connected with arable and livestock farming, forestry and so on are considered to be economic processes.
- Always consider the soil as part of the environmental system, even when it is agricultural soil.
- Consider the harvestable part of the crop as part of the economic system.
- The basic point of departure is that for each and every process all the environmental and economic inputs and outputs should be included as comprehensively as possible in the definition of the process. In a *cradle-to-grave* analysis the fixation of (so-called short-cycle) carbon dioxide by crops (as a negative CO_2 emission) and the later release of this carbon dioxide (as a positive CO_2 emission) can be left out consideration, because there is no nett emission involved.
- In a *cradle-to-gate* analysis of an agricultural product, on the other hand, the negative emission of (short-cycle) carbon dioxide due to photosynthesis should be included. If it is decided to omit it, this must be explicitly indicated.

2.3 Delineating the boundary between relevant and non-relevant processes

One of the basic problems in analysing the life cycle of a product is that of *infinite regression*: every process refers to prior and subsequent processes. The machine that manufactured the tractor was itself also manufactured, and the waste processing plant that processes the discarded tractor will itself also ultimately have to be dismantled and processed. All these processes are part of a product's life cycle in principle, but is will be clear that they cannot all be included in the analysis and that the contribution of most of these processes to the product's overall environmental impact will be negligible. In addition, the production of virtually every product implies a need for capital goods, buildings and support facilities such as a canteen.

In practice, support facilities, depreciation and maintenance of capital goods are not generally included in the analysis and the process tree is cut off at processes relating only very indirectly to the ultimate production/processing of the product under investigation. In the agricultural context, the environmental impacts associated with the production and maintenance of machinery, farm tracks and roads and buildings are not always negligible, however (see § 2.3.1). General rules for determining the relevance of processes for the ultimate environmental impact of the functional unit are very difficult to give, however.

Basis: *LCA Guide*, § 2.1.2

Additions: 2.3.1 Cut-off criteria for agricultural processes
2.3.2 The relevance of capital goods in the agricultural sector

2.3.1 Cut-off criteria for agricultural processes

The life cycles of agricultural products are made up of specifically agricultural processes and more general processes. With regard to the more general processes, the life cycle of an agricultural product does not, in principle, differ from that of any other arbitrary product. Specific guidelines for discriminating between relevant and irrelevant processes cannot therefore be given for this part of the life cycle. Specifically agricultural processes are processes directly connected to agricultural production. For such processes several guidelines can be given, though, for discriminating between relevant and irrelevant processes. Figure 2.1 presents a checklist of processes that may at any rate *not* simply be left out of consideration. Some of these processes are responsible for major environmental interventions during the phase of actual agricultural production (e.g. manure spreading); other processes cannot be omitted because of the importance of the environmental interventions in the preceding parts of the life cycle (e.g use of feed concentrates).

Processes not included in the checklist are not, by definition, irrelevant. If it is not possible, on the basis of general process knowledge, to estimate the relevance of such a process, a different criterion will have to be employed. One possible criterion is to use an *indicator*, such as the share of the process in the costs, energy/exergy or mass involved in the overall life cycle. The question is whether it is possible, for specifically agricultural processes, to indicate what indicator is most appropriate for this purpose.

The *costs* involved in producing an agricultural product frequently consist largely of labour costs. On the other hand, labour-intensive agricultural processes are generally responsible for only a small part of the overall environmental impact of an agricultural product. If costs are employed as an indicator, then, labour-related costs should first be deducted from the total costs. The *mass* involved in the production of a functional unit is not a good indicator, because processes such as pesticide and energy use would then be left out of consideration, although their environmental impact is certainly not negligible. Mass is consequently not a good indicator, because the substances and materials used in agricultural processes differ substantially in the degree to which the burden the environment per kg of substance or material. The *energy contribution* of a process to the overall agricultural process is relevant for virtually every process taking place in the agricultural sector. The energy contribution of a process to the overall agricultural process can in principle be employed as an indicator for the share of the individual process in aggregate environmental impact. However, it should be borne in mind at all times that this indicator is not appropriate for relatively low-energy processes that are environmentally burdening in other respects, such as pesticide use. This indicator should therefore also be used with due caution, but it still appears to be the best possible indicator.

In practice, in determining the cut-off points use will have to be made of a combination of the three above criteria: cost, mass and energy contribution. For agricultural processes, mass should never be used as a single cut-off criterion.

FIGURE 2.1 Relevant agricultural processes: a checklist

I Crop cultivation

1. fertilizer use
   - artificial fertilizer use;
   - manure use;
   - sewage sludge use;
   - compost use;
   - fossil fuel use;
2. crop protection:
   - pesticide use;
   - fossil fuel use;
3. soil tillage:
   - fossil fuel use;
4. irrigation:
   - water use;
   - fossil fuel use;
5. sowing:
   - seed and seedling use;
   - fossil fuel use;
6. harvesting:
   - fossil fuel use;
   - production of organic waste;
7. capital goods: production and maintenance of machinery, farm tracks and roads and buildings.

II Livestock breeding

1. feeding:
   - roughage use;
   - concentrate use;
   - fossil fuel use;
2. care:
   - shed heating and ventilation;
   - drug use;
   - fossil fuel use;
3. manure-related activities:
   - manure production;
   - manure storage;
   - manure treatment (e.g. drying);
   - manure spreading;
   - manure disposal;
   - fossil fuel use;
4. shed maintenance:
   - shed cleaning;
   - fossil fuel use;
5. milking:
   - milking equipment use;
   - milking equipment cleaning;
   - milk cooling;
   - fossil fuel use.

2.3.2 The relevance of capital goods in the agricultural sector

In the agricultural sector the environmental interventions associated with the use of capital goods (machinery, buildings and farm tracks and roads) are frequently anything but negligible. This was demonstrated by sensitivity analyses undertaken as part of the case on arable farming in this project. In a number of examples investigated in a Danish study (Weidema *et al.*, 1995) it was found that capital goods are responsible for about 15% of the primary energy resources consumed by the product system. The major contribution was from the production and maintenance of farm machinery. In § 2.4.5 it is explained how the environmental interventions associated with the production and maintenance of machinery can be quantified. The construction of farm tracks and roads can also make a significant contribution to the impacts scores of an LCA, because relatively large amounts of fossil fuels are required for road-building and for production of the building materials. Buildings make a relatively minor contribution, because the quantity of materials required for buildings is comparatively small. Nonetheless, buildings may contribute several per cent to the overall environmental impact of the functional unit. In principle, then, buildings should be included in the analysis.

2.3.3 Guideline: the boundary between relevant and non-relevant processes

- Establish for which processes in the process tree information is already available or relatively easy to obtain. These processes should, in principle, not be omitted, even if their contribution is only minor.

If process data are not readily available

- Establish which processes are intimately connected with production of the product under investigation (the functional unit). These processes can be omitted only if there are clear indications that they are of only minor relevance. As a process in the process tree becomes further removed from the direct production of a functional unit, the chance of its becoming less relevant increases.
- Establish as well as possible, using general process knowledge (by consulting policy documents and specialists, for example) which processes probably make a relevant contribution to the environmental impact of the functional unit, for whatever reason. These processes may not be omitted. In this step, where direct agricultural processes are involved, use can be made of the checklist of relevant processes (Figure 2.1) for agricultural products.
- With some processes general process knowledge will indicate that it is above all a small number of very specific environmental interventions or sub-processes that are relevant. One can then opt to include such processes *in part*, by including only these specific environmental interventions or sub-processes. For example, the production of capital goods can be included as a rough approximation, by taking only energy and material requirements into consideration.
- The relevance of including processes can sometimes be established on the basis of experience gained in earlier, similar studies.
- Establish which processes *might* be considered for omission.
- Establish, for each of the processes that might be omitted, what is already known about its emissions and extractions. Estimate the relevance of these environmental interventions. In doing so, make due allowance for what is generally known about the problems associated with the emissions and extractions of these substances, with reference to the following questions:
  - Is the substance emitted on the black list?
  - Is there a risk of the substance becoming depleted?

  If there are indications that problematical emissions or extractions occur, the process can be omitted only if it is quantitatively very insignificant.

- Make a rough estimate of the quantity of *energy* and/or exergy associated with each of the processes in the summary process tree per functional unit and use this information to estimate the total quantity of energy required for the production of the functional unit.
- Make a rough estimate of the *costs* associated with each of these processes per functional unit and determine the cost price of the functional unit.
- Make a rough estimate of the *mass* of the total quantity of material (expressed in units of mass) associated, per functional unit, with each of the processes that might be omitted and estimate at the same time the mass of the functional unit itself. (This is not always feasible: sometimes the functional unit has no unit of mass but is defined, for example, as '1 MJ of electricity'.)
- Establish, on the basis of these estimates for each of the processes that might be omitted, the contribution of the process to the total cost price, the total quantity of energy required for production of a functional unit and the quantity of materials required for the process relative to the mass of the functional unit. If the costs, energy requirements or material requirements are substantial compared with the costs, energy requirements or mass of the functional unit, the process cannot be omitted, unless it can be plausibly argued that the process is still not relevant. For specifically agricultural processes the energy contribution is generally the most relevant indicator.
- Establish the total costs and the total quantity of energy and materials associated with the omitted processes taken together, relative to the cost price, energy requirements and mass of the functional unit. Include this share in a sensitivity analysis as a 'one-sided spread' of the result of the LCA. If the omitted processes are responsible for an estimated 3% of the total quantity of energy used, say, and the total quantity of energy required for the processes that are being included is *X* MJ, then this *X* is an estimated 3% too low, and the spread is therefore '+3%'.

2.4 Entering the process data

As already described in § 2.1.1, each process in the process tree is characterized by both economic and environmental inputs and outputs. When entering the process data all these inputs and outputs must be described in a systematic fashion. A *format* has been drawn up for this purpose (see Appendix 1). When filling out the data, due care must be taken that an economic input always corresponds with the economic output of another process.

Basis: *LCA Guide*, § 2.2

Additions: 2.4.1 Location-specific interventions
2.4.2 Indirect interventions
2.4.3 Agricultural emissions of minerals: the mineral balance
2.4.4 Desiccating emissions
2.4.5 Production and maintenance of machinery

2.4.1 Location-specific interventions

As already mentioned in § 2.1.2, in the LCA method no allowance is generally made for location-specific impacts. When LCAs are used for the agricultural sector, however, a location-specific approach will be necessary for some environmental themes in the characterization step, because for these particular themes the potential impacts will be location-specific. Consequently, the emissions associated with these environmental impacts must be investigated in a location-specific manner. This means that in the inventory step due allowance must already be made for the environmental impacts that will be linked to the inventoried environmental interventions during characterization. The strict division between inventory analysis and classification/characterization thus becomes somewhat blurred.

Emissions of toxic substances

For the characterization of toxicity (human toxicity and ecotoxicity) a new method has been developed based on the USES model (RIVM,VROM, WVC, 1994), in which the degradation rate and the distribution of substances over the various environmental media are taken into due account in the assessment of equivalency factors. In this USES-derived model equivalency factors can be computed for organic chemicals. This model cannot be used to calculate equivalency factors for inorganic substances such as heavy metals and minerals. For this reason, and because it was not feasible in the context of the present project to calculate new equivalency factors for all the substances included in the *LCA Guide*, in the case studies performed in this project the model was used for pesticides only: with pesticides differences in degradation rate are of particular importance. Because the model makes a distinction between emissions occurring in the Netherlands and emissions elsewhere, this distinction must already be made during the inventory of toxic emissions.

Because the USES-derived model can be used only for pesticides for the time being, it is only for these substances that the distinction between emissions within the Netherlands and emissions elsewhere need be made.

Emissions of nutrifying substances

In characterizing potentially nutrifying substances, the *LCA Guide* prescribes making no distinction between locations where nutrification constitutes a potential problem and locations where this is not the case. Here, it is recommended to make such a distinction for emissions of nutrifying substances *to the soil*: emissions of nutrifying substances to the soil are considered to be potentially nutrifying only in areas where nutrification is a problem. For soil-emissions of potentially nutrifying substances, then, a distinction must already be made in the inventory phase between emissions of nutrifying substances in areas where nutrification constitutes a problem and such emissions in areas where soil exhaustion is the problem.

2.4.2 Indirect interventions

Sometimes a substance remains in the environmental medium to which it has been omitted for a short time only. This may mean that the most significant environmental impacts associated with this emission occur in a different environmental medium. The underlying intermedia transport can in principle be modelled in the characterization step: the transport characteristics can be incorporated in the equivalency factors. Some transport processes are highly dependent on specific emission characteristics, however. For example, the percentage of nitrogen that volatilizes to the atmosphere following application of manure to the soil is highly dependent on how the manure is applied. In such cases it is impossible to design a general model of the transport characteristics on the basis of substance properties. In such cases, therefore, the distribution over the various media will have to be specified in the inventory step already. An emission of manure ammonium to the soil will consequently be inventoried, to a greater of lesser extent – depending on the method of application – as an emission of ammonia to the atmosphere. In the case of pesticide emissions that originally occur in the air (in the form of spray), an estimate will already have to be made during the inventory phase of their distribution over air, crop, soil and surface water in the initial minutes following application. This depends not only on the method of application (by spray boom or by aerial spraying) but also on the height of the crop, for example. A drawback of already including the transport parameters in the inventory phase is that no allowance can be made for the residence time of the emitted substance in the medium where the emission originally occurs, and consequently nor for the associated environmental impacts. As yet, there is no way to avoid this problem.

2.4.3 Agricultural emissions of minerals: the mineral balance

The application of fertilizers during crop cultivation is accompanied by emissions of minerals that originally enter the soil system, but frequently subsequently become distributed by complex mechanisms over the various environmental media and the crop. In such cases it may simplify the issue to employ a mineral balance of the soil, in order to visualize the ultimate flows to the various environmental media[2]. Because physico-chemical intermedia transport processes – such as leaching and run-off – have not yet been modelled in the characterization step for inorganic substances, these should also be incorporated in the inventory stage.

Besides direct application of fertilizers, other kinds of mineral inputs can also be included in the mineral balance, to the extent that the functional unit can reasonably be held responsible for any emissions that might result from these inputs. For example, it is to be anticipated that when a farmer applies fertilizers to his crops he will make due allowance for natural inputs due to mineralization of 'old' organic matter, nitrogen fixation by legumes and deposition, at least to the extent that the minerals released are available to the crop. The natural supply of minerals can therefore be included in the mineral balance, to the extent that these minerals are biologically available to the crop.

The ultimate aim of the mineral balance is to obtain an overview of the inputs and outputs of minerals to and from the soil. The balance may involve various different compounds. In calculating the emissions, due allowance should be made for the form in which the minerals are reported in the balance; this does not have to be the form in which they are released as an emission. For example, volatilization of 1 kg N is equivalent to an emission of 1.2 kg NH_3.

Nitrogen

Below it is indicated which inputs and outputs should be included in the mineral balance for nitrogen.

input	output
seed and seedling use	crop harvesting
artificial fertilizer use	ammonia volatilization
manure use	denitrification
compost use	run-off to surface water
sewage treatment plant sludge use	leaching to groundwater
deposition[3]	leaching to surface water
nitrogen fixation by legumes	
mineralization[2]	accumulation of organically bound nitrogen

Mineralization and accumulation of organically bound nitrogen in the soil cause changes to the stock of organically bound soil nitrogen. The input of bound nitrogen is due to dressings of manure and other animals wastes (N_r), to crop residues left behind on the field (N_w) and to binding of mineral nitrogen to crop residues and green manure crops. In the case of arable crops a steady-state situation is frequently assumed, whereby mineralization is equal to accumulation. The assumption is thus made that there is no nett accumulation of nitrogen in the soil. For perennial crops, such as grass, the steady-

[2] The mineral balance of *the soil* differs from the mineral balance of the *farm*, as administered in the manure records currently kept on Dutch farms. In this context it is not the farm that is important, but the soil. In establishing the mineral balance of the soil the methods and data of the farm manure records may certainly be a useful source of information, however.

[3] To the extent that it is available to the crop in question. On this point see Aarts & Middelkoop (1990) and Middelkoop & Aarts (1991).

state situation has often not yet been reached and there is still a nett accumulation of organically bound nitrogen. In the characterization this accumulation scores as a nutrifying emission to the soil.

On peaty soil mineralization generally exceeds accumulation. In areas where the soil is not fertilized but is being exhausted (in a number of developing countries, for example) negative accumulation may occur with certain crops, resulting in a gradual decrease in the stock of organically bound nitrogen in the soil.

For nitrogen the following emissions to air, water and soil hold:

emissions to air:
- volatilization (NH_3);
- denitrification (N_2; N_2O);

emissions to water:
- run-off to surface water (NO_3^-);
- leaching to groundwater (NO_3^-);
- leaching to surface water (NO_3^-);

emissions to soil:
- accumulation of organically bound nitrogen in manure, crop residues, through binding to crop residues and binding to green manure crops.

Appendix 2 provides a review of Dutch data sources that can be used to quantify the various items in the mineral balance.

Phosphate

For phosphate the situation is simpler than for nitrate, because phosphate is subject to fewer degradation processes in the soil. The phosphate balance consequently comprises fewer items than that for nitrogen. Below a list is given of the items that should be included in the phosphate mineral balance.

input	**output**
seed and seedling use	crop harvesting
artificial fertilize use	run-off to surface water
manure use	leaching to groundwater
compost use	leaching to surface water
sewage treatment plant sludge use	accumulation of phosphate
deposition	
mineralization	

In the Dutch situation there is frequently accumulation of phosphate. In areas where the soil is not fertilized but is being exhausted (in a number of developing countries, for example) negative accumulation may occur with certain crops, resulting in a gradual decrease in the stock of phosphate in the soil.

For phosphate the following emissions to air, water and soil hold:

emissions to air:

none;

emissions to water:
- run-off to surface water (PO_4^{3-});
- leaching to groundwater (PO_4^{3-});
- leaching to surface water (PO_4^{3-});

emissions to soil:
- phosphate accumulation.

The phosphate emission can be established by deducting the phosphate contained in the economic outputs from the overall phosphate input. This emission must then be divided over accumulation, leaching and run-off. This distribution is governed by the phosphate condition of the soil. First the phosphate accumulates, until the soil is saturated; only then does leaching occur to any substantial degree.

Appendix 2 provides a review of Dutch data sources that can be used to quantify the various items in the phosphate mineral balance.

2.4.4 Desiccating emissions

Man can influence the natural water table in different ways:

- by direct abstraction of groundwater;
- by abstraction of groundwater through use of tap water;
- by artificial transport of rainwater to surface water;
- by artificial transport of rainwater to the sewer system;
- by supplementation of rainwater with irrigation water residues;
- by cultivating crops with a high evapotranspiration rate;
- by withdrawal of water with crop harvests;
- by controlled rainwater discharge;
- by artificial modification of surface water levels.

In the Netherlands, agricultural activities are one of the main causes of the desiccation problem. When assessing the environmental impacts of agricultural products, then, the question of how much these products contribute to desiccation should always be addressed. (This obviously does not mean that agricultural products always make a major contribution to the problem of desiccation: this contribution depends on the specific situation, such as the type of agricultural product being assessed.)

When inventorying the environmental interventions associated with a product, the entire life cycle of the product should be taken into due account. When assessing an agricultural product, then, in principle every process in that product's life cycle should be inventoried with respect to its desiccating action. With most agricultural products, actual cultivation of the product is likely to be the overriding process in terms of dessication.

In the problem of desiccation, the governing factors are drainage, water table management and groundwater abstraction. The first two aspects are highly location-specific and difficult to relate to a functional unit of product. Consequently, these aspects cannot yet be included in an LCA. What can be included, though, are the direct and indirect abstraction of groundwater related to a functional unit of product. In areas where desiccation does not constitute a problem and in situations where groundwater abstraction does not contribute to dessication (when the water table is kept artificially low, for example), groundwater abstraction should not be classified, however.

Direct groundwater abstraction
Some production processes make direct use of groundwater. This is the case with production of beer or mineral water, for example, and with irrigation of agricultural crops. In the agricultural context, direct groundwater abstraction comprises the groundwater pumped up on the farm site for the purpose of direct production of the functional unit. Direct groundwater abstraction can be expressed in m^3 water$\cdot$year^{-1} for a process or m^3 water for a product.

Table 1 gives estimates of the average abstraction of groundwater for irrigating various crops in the Netherlands. If the crop yield per ha·year is known, the quantity of irrigation water pumped up per kg product can be readily calculated from this figure. It should be borne in mind that the values given are averages. There is a considerable spread around these averages; the level of irrigation is highly dependent on soil type, for example. In location-specific studies, therefore, location-specific estimates should be employed.

TABLE 1 Estimate of the average abstraction of groundwater for sprinkling of various crops in the Netherlands

crop	abstraction ($m^3 \cdot ha^{-1} \cdot year^{-1}$)
grass	615
maize	667
potatoes	1000
sugarbeet	875
other crops	600

Besides being used as irrigation water, there are various other possible applications of groundwater within the context of agricultural production: during the manufacture of pesticides, for instance. All items accruing benefits to the functional unit in question should, in principle, be included in the LCA.

$$\textit{direct abstraction of groundwater by agricultural product } [m^3] = \textit{abstraction of groundwater for irrigation} + \textit{abstratction of groundwater for other purposes within production process}$$

Groundwater abstraction through use of tap water

In the Netherlands, approx. 65% of tap water consists of groundwater (Braat, 1989). Tap water use therefore contributes indirectly to groundwater abstraction. In the Dutch situation, then, groundwater abstraction due to tap water use can be calculated simply by multiplying the volume of tap water used by 0.65. For processes taking place elsewhere, information will first have to be obtained on the percentage of local tap water consisting of groundwater. A parallel approach can then be followed.

In the Netherlands:

$$\textit{abstraction of groundwater through tap water use } [m^3] = 0.65 \times \textit{tap water use}$$

In general:

$$\textit{abstraction of groundwater through tap water use } [m^3] = \frac{\textit{\% tap water groundwater}}{100} \times \textit{tap water use}$$

Artificial transport of rainwater to surface water

Artificial transport of rainwater to surface water occurs mainly via drainage. In addition, run-off from paved surfaces or surfaces from which the natural vegetation has been partly or completely removed may also play a role.

In the Netherlands agricultural land is one of the land categories where drainage occurs. Other, industrial sites may also be drained. The drainage water from agricultural land originates from all forms of precipitation, *i.e.* both rainwater and irrigation water. Only the discharge of rainwater may be considered as potentially having a desiccating effect.

Improved discharge and drainage of agricultural land is considered to be one of the main causes of the desiccation problem (RIVM, 1993). In practice, though, this term will often be difficult to quantify, on the one hand because of its relationship with the water management regime in the area in question, and on the other because information will often be lacking on what percentage of a certain type of farmland is drained. If it is feasible to make a quantitative estimate of the volume of drainage water relative to the functional unit, it is proposed to include this term in the LCA. If this is not feasible, there is no other choice but to exclude it.

In the field situation natural precipitation can frequently not be distinguished from artificial supplements in the form of irrigation water. Such a distinction is important for establishing the degree to which certain processes lead to withdrawal of groundwater and rainwater from the soil, however. If process-specific data is not available, the quantity of groundwater and rainwater withdrawn can be estimated by proceeding from the assumption that the ratio between the amounts of natural rainwater and irrigation water withdrawn from the soil is the same as the ratio in which they are added to the soil. To this end the ratio between the average annual rainfall and the average annual irrigation water supplement can be used.

Artificial transport of rainwater to the sewer system
Artificial transport of rainwater to the sewer system occurs when the water that runs off from paved surfaces is retained in a sewerage system. Discharge of rainwater to the sewer takes place on numerous public and private sites, including roads and buildings. In the life cycle of agricultural products this term will probably always be negligible in comparison with desiccating interventions occurring in the actual agricultural production process. Because this term is often difficult to quantify in terms of the functional unit, moreover, it is proposed to leave rainwater discharge to the sewer out of consideration in LCAs on agricultural products.

Supplementation of rainwater with irrigation water residues
It is common agricultural practice to supplement natural rainfall with artificial irrigation, either by sprinkling or by surface irrigation. As is the case with natural rainfall, this artificial precipitation can, in principle, seep down to the water table[4]. The volume of water thus returned to aquifers can be calculated by subtracting from the volume of groundwater and rainwater abstracted for the purpose

[4] If groundwater is abstracted for the purpose of irrigation, seepage will consist partly of the water originally abstracted. There will consequently be no nett addition to the volume of groundwater. However, this abstraction is less than one would expect on the basis of abstraction statistics. Because in calculating the quantity of groundwater abstracted no allowance is usually made for the fact that a fraction of the water is ultimately returned to aquifers, this fraction must be considered as an addition (in the sense of negative abstraction). The 'gross abstraction' can thus subsequently be corrected.

of irrigation the volume of irrigation water taken up by the crop and the volume of irrigation water otherwise lost:

$$\textit{seepage to groundwater } [\mathrm{m}^3] = \textit{abstraction of irrigation water} \\ - \textit{uptake of irrigation water by crop} \\ - \textit{run-off of irrigation water} \\ - \textit{blow-off of irrigation water} \\ - \textit{evaporation of irrigation water} \\ \textit{(spray and soil and crop surface)} \\ - \textit{drainage of irrigation water}$$

Cultivation of crops with a high evapotranspiration rate

In agricultural areas and artificially forested areas, the natural vegetation has been replaced by 'artificial' crops. As a result of the growth of agricultural output, there has been a major increase in evapotranspiration from agricultural crops over the past few decades. Evapotranspiration from agricultural crops is consequently a desiccating intervention, at least to the extent that evaporation of groundwater and rainwater uptake is involved. Evaporation of irrigation water – even if this water is groundwater – cannot be considered a desiccating intervention, because the abstraction of groundwater for the purpose of irrigation has already itself been taken to be a desiccating intervention. An estimate will therefore have to be made of the percentage of evapotranspiration directly involving groundwater and rainwater. If there are no pertinent data available, the share of rainwater in overall evapotranspiration can be estimated by adopting the following method. In doing so, by sheer necessity the quantity of groundwater taken up by the crop itself must be left out of consideration.

As in built-up areas, in agricultural areas, too, rainwater discharge plus crop evapotranspiration must be corrected by subtracting the amount of evaporation that would have taken place under natural conditions. A simple solution is to consider as 'non-desiccating' the direct evapotranspiration of groundwater and rainwater from crops that occurred prior to 1950. The extra amount of crop evapotranspiration is then considered to be desiccating – to the extent that groundwater and rainwater are involved. Because evapotranspiration prior to 1950 is not linked directly to a quantity of product, but to an area of land, the extra evaporation from agricultural crops should first be calculated per hectare.

$$\textit{extra evapotranspiration of groundwater and rainwater per ha } [\mathrm{m}^3 \times \mathrm{ha}^{-1}] = \\ \textit{evapotranspiration of groundwater and rainwater per ha crop} - \textit{evapotranspiration before } 1950$$

Subsequently, a simple conversion can be made to obtain the extra evapotranspiration due to the functional unit (f.u.) in question:

$$\textit{extra evapotransp. of groundwater and rainwater } [\mathrm{m}^3] = \frac{\textit{extra evapotransp. of groundwater and rainwater per ha}}{\textit{no. of f.u. per ha}}$$

The assumption that the evapotranspiration prior to 1950 can be considered non-desiccating is valid for the Dutch situation, because the problem of desiccation first emerged clearly in the Netherlands around 1950. For agriculture under conditions deviating strongly from the Dutch situation – in the tropics, for example – a different solution will have to be found (*This is similar to the problems associated with nutrifying emissions*).

In the Netherlands, total evapotranspiration increased by an average of 650 m^3 per ha·year during the period 1951-1980 (following De Wit, 1991). If the crop-specific difference between current evapotranspiration and evapotranspiration prior to 1950 is not known, this figure can be used as an estimate.

Withdrawal of water with crop harvests

In addition to evapotranspiration from agricultural crops, crop harvesting is also a desiccating intervention: water fixed in the plant is removed. Here, too, only withdrawal of groundwater and rainwater may be taken to be desiccating: withdrawal of irrigation water cannot be considered desiccating, because this would entail double counting.

The quantity of groundwater and rainwater removed with the crop can be calculated as follows:

$$\textit{withdrawal of groundwater and rainwater via harvest}\ [\mathrm{m}^3] = \frac{1}{\textit{specific mass of water}} \times (\textit{total mass of harvest} - 0.4 \times \textit{mass of harvested dry matter}) \times \textit{fraction of rainwater in absorbed water}$$

The first term has been added to balance the dimensions: these are 1 $m^3 \cdot tonne^{-1}$. The mass of the harvest should therefore be expressed in tonnes.

Calculation of overall withdrawal of groundwater

The overall withdrawal of groundwater can be calculated as follows:

$$\begin{aligned}\textit{withdrawal}\ [\mathrm{m}^3] = {} & \textit{direct abstraction of groundwater} \\ & + \textit{abstraction of groundwater through tap water use} \\ & + \textit{drainage of rainwater} \\ & - \textit{seepage to groundwater} \\ & + \textit{extra evapotranspiration of groundwater and rainwater} \\ & + \textit{withdrawl of groundwater and rainwater via harvest}\end{aligned}$$

2.4.5 Production and maintenance of machinery

As mentioned earlier, in § 2.3.1, the environmental interventions resulting from the production and maintenance of farm machinery are often non-negligible. The interventions associated with machinery production can be divided into interventions associated with resource extraction and production of the parts required for machinery production on the one hand, and interventions associated with the manufacture and assembly of the machinery on the other. The main issues here are the production of the materials and the energy required to produce, maintain and repair the machinery.

To determine what fraction of the interventions associated with machinery production and maintenance is due to a given agricultural activity, the following formula can be applied to each intervention:

$$I_a = \frac{C_a}{C_m} \times I_m$$

where:

I_a = intervention due to the activity [unit]
C_a = capacity of the machinery required for the activity [hours]
C_m = total capacity of the machinery [hours]
I_m = total intervention associated with the production and maintenance of the machinery [unit]

In practice, data on the interventions associated with the production and maintenance of a given item of machinery will not always be available. In that case use can be made of indices for the quantities of energy resources, iron ore and limestone required to produce and maintain an item of machinery, together with indices for the emissions to air and water occurring during production of the machinery. The inputs of iron ore and limestone are then taken as direct environmental inputs, while the energy resources are viewed as economic inputs. A set of figures is given in Appendix 3, with the indices for the inputs of energy resources, iron ore and limestone given per kg machinery.

2.4.6 Guideline: the process data

- Follow the instructions in the *LCA Guide* (§ 2.2). A form for entering the process data according to the *LCA Guide* format is included in Appendix 1.
- If the distribution of a certain emission over water, soil and air is very dependent on the emission characteristics, then already estimate the distribution in the inventory phase. This should always be done for the application of fertilizers and crop protection agents.
- To establish the distribution of minerals over water, soil and air, employ a mineral balance, as described in § 2.4.3.
- In inventorying desiccating interventions, always include at least the following processes:
  - direct groundwater abstraction;
  - use of tap water;
  - artificial transport of rainwater to surface water;
  - extra crop evapotranspiration (relative to the 1950 level);
  - withdrawal of water with the crop harvest.

  The total withdrawal of groundwater can be calculated as follows:

$$\begin{aligned} \textit{withdrawal}\ [\mathrm{m}^3] = {} & \textit{direct abstraction of groundwater} \\ & + \textit{abstraction of groundwater through tap water use} \\ & + \textit{drainage of rainwater} \\ & - \textit{seepage to groundwater} \\ & + \textit{extra evapotranspiration of groundwater and rainwater} \\ & + \textit{withdrawl of groundwater and rainwater via harvest} \end{aligned}$$

- Include the production and maintenance of machinery by using the method outlined in § 2.4.5.

2.5 Allocation of environmental interventions

In the inventory phase the share of the functional unit in each process in the life cycle is established. For some processes this is simple to calculate. However, if a process fulfils multiple functions (has multiple outputs) that do not all relate to the product under investigation, matters become more complicated. The environmental interventions of that process will then have to be divided over the various functions according to a certain distribution key. It is thus in multifunctional processes that allocation constitutes a problem. In practice, three such situations occur (Huppes, 1992; Consoli *et al.*, 1993):

- co-production;
- open-loop recycling;
- combined waste processing.

Co-production refers to a process having multiple tradeable outputs; such a process is sometimes termed a *multi-output process*. An example of a multi-output process is combined production of milk, meat and leather (and possibly other products) in dairy farming. The share of milk production (rather than beef or leather production) in the environmental interventions of dairy farming is not immediately clear.

Recycling is the re-use of a waste product in a new production process. If the waste product is re-used in the same product system, this is referred to a closed-loop recycling. A case in point is the use of cow manure in the cultivation of maize, which is used a cattle feed. In the case of closed-loop recycling there is no allocation problem, because the recycling process occurs within one and the same product system. If waste products from a given product system are used in a different product system, it is referred to as open-loop recycling. Open-loop recycling often involves some form of reprocessing, with waste from one product system being worked up to a raw material for the following product system. Reprocessing then fulfils at least two functions: waste processing and production of a new raw material. A case in point is the processing of manure from intensive livestock farming to granular fertilizer for use in horticulture and arable farming. It is not immediately obvious how the environmental interventions of the manure processing plant are to be divided over the product systems 'livestock farming' and 'horticulture and arable farming'.

Combined waste processing refers to waste flows from various different production processes being processed together. An example is combined incineration of domestic waste and waste from the office, shop and services sector. The emissions from the waste incineration plant must then be divided over these two systems.

What is allocated?

The distribution key for allocation can be applied to three sets of items:

- the environmental inputs and outputs of processes (extractions and emissions);
- the economic inputs and outputs of processes ((intermediate) products, services and waste to be processed);
- (sub)processes.

The ultimate aim is to allocate environmental inputs and outputs. Allocation of economic inputs and outputs or (sub)processes is in fact an intermediate step.

An example of direct allocation of an environmental output is allocation of livestock-shed emissions to milk, meat and leather.

An example of allocation of an economic input is allocation of chicken feed to meat and eggs. Here, the ultimate aim is to allocate the emissions and extractions occurring in the chicken feed life cycle.

An example of allocation of a subprocess is allocation of wheat sowing to wheat and to straw. Here, too, the ultimate aim is to allocate the emissions and extractions occurring in the life cycle of the sowing process.

To what is allocated?

Allocation is always to the economically valuable outputs of processes (products or services: commodities) and *not* to processes, persons or agencies, *nor* to waste products. Emissions of crop protection agents during potato cultivation are thus allocated to the potatoes, and not to potato cultivation, the potato farmer, the agricultural cooperative or the potato foliage.

With agricultural products it is above all in the cases of co-production and recycling that allocation is an important issue. Until now allocation of combined waste processing has not led to any specific problems in LCAs on agricultural products and use can consequently be made of the general guidelines in the *LCA Guide*. Here, we therefore consider allocation in the cases of co-production and recycling only.

Basis: *LCA Guide*, § 2.3

Additions:
2.5.1 Allocation versus substitution
2.5.2 Allocation in the case of co-production of agricultural products
2.5.3 Allocation in the case of recycling: livestock wastes
2.5.4 Allocation in a crop rotation scheme

2.5.1 Allocation versus substitution

In the past few years a general consensus has emerged on the procedure to be followed for allocation:

Step 1: Split multi-output processes as far as possible into single processes. An example is dairy farming, where both meat and milk are produced. The milking of the cows must be separated from meat production and should be allocated to the milk. Medications given to the cows, in contrast, are often relevant for the production of both meat and milk.

Step 2: If, for a given process, the degree to which the process is used for each of the individual products is quantifiable, these data can be used for allocation. A case in point is the marketing activities undertaken by a firm selling a variety of products: it is then fairly straightforward for staff to keep a record of the number of phone calls they make for each product. In the case of agriculture this step is probably less important.

Step 3: If certain emissions are physically related to one of the products in question, these emissions should be allocated to that product.

Step 4: Only if there are still problems remaining after the first three steps have been followed may allocation be based on value to society. This is best expressed as *turnover: market price times yield (in mass units).*

Substitution

In some cases agricultural products normally produced in combination with other products may also be produced in an alternative manner, in a single-output process. This is the case with glycerine, for example: this product can be produced from oilseed rape or synthetically (Oegema & Pasma, 1994). When it is produced from oilseed rape, glycerine is a co-product, alongside rapeseed oil and animal feed; synthetic glycerine production, on the other hand, is a single-output process. In such cases allocation can be avoided by deducting the environmental interventions of the single-output process from the environmental interventions of the multi-output process. In the example cited the environmental interventions of synthetic glycerine production are thus deducted from those of the co-production of rapeseed oil, animal feed and glycerine, to yield the environmental interventions of joint production of the rapeseed oil and animal feed.

In general, however, there are no alternative single-output processes for agricultural products produced in co-production. In such cases, then, allocation rules must be employed to establish the environmental interventions for which each individual product is responsible.

Besides allocation on the basis of social causality, substitution of environmental interventions by the interventions of a comparable co-product is a good possibility. One requirement here is that the co-product in question must indeed be considered by society to form a real substitute for an already existing product, which is not generally produced as a co-product. (It should be added, though, that this is often difficult to establish.) In applying this method it should be assessed on a case-by-case basis whether the system does not become too large and unwieldy.

2.5.2 Allocation in the case of co-production of agricultural products

Co-production is a frequent occurrence in agriculture. In dairy farming, for example, production of milk, meat, calves and leather cannot each be considered in isolation. In arable farming the entire crop rotation can be seen as a multi-output process, although in this case a large number of processes can be distinguished which are geared to a single crop and for which allocation can be based on physical causality. Some crops in themselves yield more than one product: cereal and straw, for example. In such cases there is clearly co-production.

In allocating indivisible multi-output processes the aim is to establish the extent to which each of the co-products can be held responsible for the sum total of environmental interventions of the process (and upstream processes). The art is to find an optimum yardstick for the degree to which a given product can be held 'responsible'. What is 'good' in this respect cannot be objectively established, however: it depends on what is meant by 'responsibility'. In most discussions the concepts of 'physical' and 'economic' responsibility are distinguished. Although the physical units are generally readily measurable and stable data, this does not necessarily mean that they form a true reflection of the (partial) responsibility of a given output for a certain production process taking place. Allocation on the basis of a physical quantity is consequently only justifiable if the chosen quantity reflects some degree of causality, for the energy content of jointly produced fuels with a comparable utilization capacity, for example. If there is no physical causality, there is no sense in taking a physical basis for allocation and it is better to seek a yardstick for social causality. Social causality can be quantified as the turnover (in dollars, say) = market price (in dollars/kg) times yield (in kg). It is not always easy to establish the true market price, especially if subsidies are involved. In such cases one solution may be to employ average world market prices.

Society's valuation of a product is subject to change. This means that proportional allocation of multi-output processes may vary with time. This is not a weak point, but in fact a strong point of this type of allocation. It indicates the partial responsibilities of co-products may change: if one co-product becomes more expensive in relation to another, the production process is carried out more for the former co-product. For example: if gold could suddenly be extracted from pig manure, pigs would no longer be kept for their meat but for their manure. Most of the environmental interventions would then be allocated to the manure, even though the physico-chemical reality of the process remains unchanged.

2.5.3 Allocation in the case of recycling: livestock wastes

In the agricultural context manure (and livestock wastes in general) is an important example of a product that is 'recycled'. It is proposed to allocate the environmental interventions associated with manure production to livestock farming and those associated with manure use to arable farming.

Transport and storage of manure can be seen as upgrading processes, whereby a waste product from livestock farming is converted into a valuable resource for arable farming. In principle, it is proposed to allocate the environmental interventions associated with these processes to the product system paying for these processes. If there is joint payment by several different parties (in the case of storage at a manure centre, for example) the environmental interventions are allocated on the basis of the ratio between the cost of waste processing paid by the livestock farmer and the cost of manure as a resource paid by the arable farmer.

2.5.4 Allocation in a crop rotation scheme

In arable farming crops are often cultivated successively in a crop rotation scheme: 33% potatoes, 33% sugarbeet and 33% wheat, for example. In practice, there is considerable variation in the design of crop rotation schemes. Often processes carried out primarily for one of the crops will also have consequences for the cultivation of other crops in the scheme. In potato cultivation, for example, a soil fumigant is sometimes used and this is also of benefit to the following crop: wheat, say. This could be an argument for allocating some fraction of the environmental interventions associated with the soil fumigant to wheat cultivation. We do not opt for this approach, however. The reason for this is that the main question to be addressed in the context of a crop rotation scheme should be: What is the reason for a given activity? In the above example the soil fumigant is used because this is essential for potato cultivation; if potatoes were not included in the crop rotation scheme, the soil fumigant would not be used. The associated environmental interventions should therefore be allocated entirely to potato cultivation.

Similarly, fertilizers are often taken up by crops other than those to which they were originally applied. Depending on the fertilizer regime, uptake and uptake efficiency, only part of the minerals will be taken up by the crop to which the fertilizer is applied. The remainder will remain in the soil or be transported to the atmosphere or water. The fraction remaining in the soil will be available to the following crops in the crop rotation scheme. Here, too, the allocation question must be answered on the basis of the reason for application. The answer to this question may differ substantially from mineral to mineral. Phosphate and potassium, for example, are often applied as fertilizer for the entire rotation scheme, benefiting all the crops in the scheme. In contrast, virtually all the nitrogen applied (apart from the small *Nr* fraction from manure) benefits the crop to which it is applied.

In practice, this leads to the following recommendations:

1. In principle, the emissions of nitrogen are allocated to the crop to which the fertilizer is applied. The sensitivity analysis performed in the arable farming case study indicates that allocating nitrogen fertilizer application and the associated environmental interventions in their entirety to the crop of application (instead of using allocation rules for the N_r fraction) has only a negligible effect on the results. Only in special cases is it therefore necessary to divide the emissions due to the *Nr* fraction

over the other crops in the rotation scheme (based on the share of the crops in the scheme, expressed in ha·y). This is the case, for example, when two crops are being compared, one of which is fertilized solely with artificial fertilizer and the other solely with manure.

2. The emissions associated with a phosphate and potassium dressing are divided over the various crops on the basis of the recommended dressing reported in agricultural handbooks. In this way due allowance is made for average uptake and uptake efficiency. It would be easier to allocate on the basis of individual crop uptake, because these data have already been gathered during the inventory phase. However, the sensitivity analysis performed in the arable farming case study shows that this has too great an impact on the final results.

3. The emissions associated with a dressing of (effective) organic matter are allocated to all crops on the basis of the share of the crops in the rotation scheme. Organic matter benefits the soil structure and its general condition, and all crops benefit accordingly. The sensitivity analysis performed in the arable farming case study shows that omission of this allocation has too great an impact on the final results.

4. The nitrogen in crop residues may be of benefit to all the crops in a rotation scheme. The emissions of nitrogen from crop residues are therefore not allocated to the crop from which these residues arise, but divided over all the crops in the rotation scheme. Emissions from crop residues can only be omitted if:

- each crop in the rotation scheme yields the same amount of minerals in crop residues;
- two cultivation systems with equal yields of a single crop are being compared, in the context of one and the same rotation scheme.

A complication arises if multiple fertilizers such as manure are applied. Manure contains nitrogen, phosphate, potassium and organic matter. It is not so much the emissions occurring after application that form the problem, but rather the environmental interventions up to and including application: the economic processes *manure storage, manure transport* and *manure application*. For example: the emission of CO_2 due to tractor usage cannot simply be allocated to the crop to which the manure is applied because only the nitrogen from the manure is allocated directly to the crop. This can be solved by assigning an economic value to each fraction of the manure and then allocating on the basis of this value.

2.5.5 Guideline: allocation of environmental interventions

- Follow the instructions in the *LCA Guide*.
- Split multi-output processes as far as possible into single-output processes on the basis of physical causality, in order to avoid the need for allocation.
- For each individual process, make a choice as to the basis of allocation (not for the entire process tree, in other words).
- It is possible to opt for substitution instead of allocation if there is a realistic alternative single-output process for the co-product.
- If allocation is necessary, then opt for allocation on the basis of social causality. This form of causality is best reflected in pure turnover (market price times yield, minus any subsidies). If a physical quantity can also be found that reflects social causality well, this quantity can also be used as the basis of allocation.
- If there are several possible bases for allocation, at the end of the study perform a sensitivity analysis to establish the extent to which the final result of the study depends on the choice of allocation basis.

3 CLASSIFICATION & CHARACTERIZATION

In the *LCA Guide* the inventory analysis is followed by *classification*. Under the influence of international discussions, the terminology has been somewhat modified, however (Consoli *et al.*, 1993). Consequently, this chapter is now called 'Classification and characterization'.

In the classification according to the new guidelines (Consoli *et al.*, 1993) all the environmental impacts associated with each environmental intervention are listed. This is in fact a methodic step, which has already been elaborated in the *LCA Guide*. It is obviously also possible, in a specific product study, to list for each intervention in the inventory table what environmental impacts the intervention has. In practice, it is sometimes found that classification is used to indicate which environmental impacts are to be included in the assessment, and which not. (In the *LCA Guide* this is referred to as 'Selection of the problem types'.) In characterization the environmental profile is drawn up and the impact scores normalized, if so desired. The term 'classification factors' has been replaced by 'equivalency factors'. The term 'characterization factors' is also sometimes encountered.

Basis: *LCA Guide*, Chapter 3

Additions:
- 3.1 Nutrification (addition to § 3.2 of the *LCA Guide*)
- 3.2 Desiccation (addition to § 3.2 of the *LCA Guide*)
- 3.3 Toxicity (addition to § 3.2 of the *LCA Guide*)

3.1 Nutrification

If the mineral balance indicates that accumulation of minerals in the soil occurs with a certain crop, this does *not* score if cultivation takes place in an area where nutrification does not constitute a problem. Although nutrification is a problem virtually throughout Europe, in certain areas in developing countries where soils are being exhausted it is not. However, emissions of minerals (added as a soil dressing) to other environmental media, via run-off, leaching and volatilization, for example, are still classified as nutrifying, because these emissions may lead to the nutrification (eutrophication) of surface waters or of adjacent areas where nutrification does constitute a problem.

3.2 Desiccation

The result of inventorying the desiccating interventions represents the number of m^3 of groundwater withdrawn, summed over the life cycle of the functional unit. In characterization, this figure is taken directly as the impact score:

$$\textit{desiccation}\ [\mathrm{m}^3] = \textit{groundwater withdrawal}\ [m^3]$$

As in the case of nutrification, when considering desiccation due allowance should be made for the difference between problem areas and non-problem areas. Groundwater abstraction in areas where desiccation is not a problem or in areas where groundwater abstraction does not contribute to this problem (in the case of surface water levels being kept artificially low, making water level management the main cause of any desiccation, for example) should not be taken to be a desiccating intervention. If it is unknown whether desiccation is a problem in the area being considered, groundwater abstraction should be classified as desiccating.

3.3 Toxicity

For a number of crop protection agents new equivalency factors have been calculated for the toxicity themes *human toxicity* and *aquatic ecotoxicity*, incorporating cross-media transport and degradation. These factors have been calculated with the aid of the modified USES model (see Part 3: Methodological Background § 3.1.2). Since these values have not yet been validated, they are not included in this report[5].

3.4 Guideline

- Follow the instructions in the *LCA Guide*.
- In areas where nutrification does not constitute a problem, set the equivalency factor for emissions of nutrifying substances to zero.
- In areas where desiccation does not constitute a problem, set the equivalency factors for direct and indirect withdrawal of groundwater to zero.
- Make due allowance for the fact that the equivalency factors in the *LCA Guide* are based solely on intrinsic toxicity and that the environmental behaviour of substances (transport and degradation) has been ignored in the calculation of these factors.
- Make due allowance for the fact that impact scores for human toxicity as well as aquatic and terrestric ecotoxicity can be calculated with different types of equivalency factors. Two different lists of equivalency factors are the list in the *LCA Guide* and the list in the report of Guinée *et al.* (1996). Take care not to add up results that have been calculated with different types of equivalency factors, since these are not compatible.

5 After the publication of the Dutch original of this report, another report about this subject has been published (Guinée *et al.*, 1996). In this report, which is the result of a joint project of CML and RIVM, new equivalency factors for human and ecotoxicity, taking environmental fate into account, are included. These new equivalency factors – which are based on the RIVM USES-model – are not compatible with the old factors; the two sets of factors differ in their dimensions, for instance. The score of crop protection agents on the toxicity themes calculated with the new factors cannot therefore be added to the scores of other substances calculated by the old method. When using these new equivalency factors the resulting effectscores should therefore be listed separately.

APPENDIX 1 LCA data format

main page

format LCA process data

Centre of Environmental Science
Leiden University, The Netherlands
version 7.2 ; 1993 November 30
file = format72.bld ; by LVO/NWB/AWS

•for explanation see next page•

see also *Environmental life cycle assessment of products, Guide & Backgrounds - October 1992*

1 format

1.1 name or institute . . . .
. . . .
1.2 date (YY MM DD)
1.3 availability of data: public/confidential
. . . .
1.4 page: 1 of .... (number of pages)
1.5 comment (about 1.1-1.2-1.3-1.4) none / see page......

2 process

2.1 *name or code* . . . .
2.2 representativeness
•scale: global / continental / national / local / company
namely . . . .
•dating: historical processes / 60's / 70's / 80's / 90's / 19.. / future processes
•duration: . . . . (hr)
•capacity: . . . . (.................per year)
•status: empirical measurements / best practicle means / models / allocated / theoretical estimations / secondary analysis of literature
else: . . . .
•allocated process . . . . no/yes;
allocation basis: . . . . mass/ economic value/ else:
. . . .
2.3 quality
•clarity: including . . . . calamities, transport, other:
. . . .
excluding . . . . calamities, transport, other:
. . . .
•accuracy: mass balance / energy balance / independent sources / not verified
•completeness of environmental input: lacking / unknown / zero / complete
•completeness of environmental output: lacking / unknown / zero / complete
•completeness of economic input: lacking / unknown / zero / complete
•completeness of economic input: lacking / unknown / zero / complete
2.4 sources
•*nature of source: company/independent body/secondary checked/not checked*
namely . . . .
•age of source: .... (yr)
2.5 overall assessment
2.6 comment (about 2.1-2.2-2.3-2.4-2.5) . . . . none / see page.....

3 economic input

A inputs with costs for this process

3.1 goods . . . . none / unknown / see page......
3.2 services . . . . none / unknown / see page......
3.3 materials . . . . none / unknown / see page......
3.4 energy . . . . none / unknown / see page......

B inputs with proceeds for this process

3.5 waste to be processed: . . . . none / unknown / see page......

5 economic output

A outputs with proceeds for this process

5.1 goods . . . . none / unknown / see page.....
5.2 services . . . . none / unknown / see page.....
5.3 materials . . . . none / unknown / see page.....
5.4 energy . . . . none / unknown / see page.....

B outputs with costs for this process

5.5 waste to be processed: . . . . none / unknown / see page.....

4 environmental input

4.1 resources

4.1.1 abiotic resources . . . . none / unknown / see page......
4.1.2 biotic resources . . . . none / unknown / see page......
4.1.3 energy carriers . . . . none / unknown / see page......
4.2 space . . . . none / unknown / see page......

6 environmental output

6.1 emissions to air . . . . none / unknown / see page.....
6.2 emissions to water . . . . none / unknown / see page.....
6.3 emissions to soil . . . . none / unknown / see page.....
6.4 radiation . . . . none / unknown / see page.....
6.5 sound . . . . none / unknown / see page.....
6.6.1 heat to air . . . . none / unknown / see page.....
6.6.2 heat to water . . . . none / unknown / see page.....
6.7 light . . . . none / unknown / see page.....
6.8 calamities . . . . none / unknown / see page.....

7 balances . . . . none / unknown / see page......

7.1 mass balancing item
...... kg input (3+4) ratio out/in:......
...... kg output (5+6)

7.2 energy balancing item
......MJ input (3+4) ratio out/in:......
..... MJ output (5+6)

8 comments/other interventions none / unknown / see page.....

additional page

ormat LCA process data

entre of Environmental Science Leiden University, The Netherlands

rsion 7.2 ; November 30, 1993

format

.1 name or institute ....

....

.2 date (YY MM DD)

.4 page: .... of .... (number of pages)

2 process

2.1 name or code ....

/4/5/6/7/8 economic / environmental

nput / output / comment / balance

3/4/5/6/7/8 economic / environmental

input / output / comment / balance

Short explanation of the format *This page forms the back-side of the blanco format and contains all the items of the process data format, with a short explanation for each item.*

file = format72.exp/ad version 7.2 931130

1 FORMAT

1.1 name or institute: the one who filled in the format

1.2 date YYMMDD: the data of filling in this format, in the order year/month/day

1.3 availability: indication whether the data are confidential or can be used for other studies

1.4 page: number of pages used for this particular process

2 PROCESS

to be entered for every single input/output if necessary

2.1 name or code: a clear name for this particular process

2.2 representativeness:

•scale: the scale indicates whether the process represents a global, continental, national or local average or whether the process is typical for the company concerned

namely: a specification of the area the data are representative for (for example "western world")

•dating: the approximate date for which the process is representative

•duration (hours): the time required to produce the volume described

•capacity (ton/yr or pieces/year):the amount of product to be produced in a standard amount of time

•status: to indicate whether the process actually exists and has been measured, is derived from other data (e.g. by extrapolation) or is a design definition

•allocated process: to indicate if process data refer to processes that have already gone through an allocation procedure

•allocation basis: to indicate how the allocation was done

2.3 quality:

•clarity: a process is defined clearly when it is clear which parts of the operation are included and excluded (e.g. transport; accidental emissions)

•accuracy: indication whether the data are accurate or verified with other sources or by mass balance or energy balance calculations

•completeness: a list of all possible inputs or outputs, with an indication of data completeness. (Thus it becomes clear if data that are not mentioned are lacking, unknown or zero, and if data that do have a value represent a complete input or output.)

2.4 sources:

•nature of source: to indicate the authority of the data. Generally, data collected by an independent body have more authority than company data, and data that are secundary checked have more authority than data that are not checked.

namely: the name of the source should be entered here

•age of the source: an indication of the date the data were collected (which does often not correspond with the date of publication of these data)

2.5 overall assessment: overall assessment of the set of process data on the basis of the representativeness and quality of the data described. The assessment of the accuracy and completeness of the data in particular will determine the overall assessment.

3 ECONOMIC INPUT

Inputs are economic if they are the outputs of another production process.

A inputs with costs for this process

3.1-3.4 goods, services, materials, energy: a complete list of the economic inputs (in SI units if possible) with costs for this process

The direct use of fossil fuels should not be mentioned here: this is considered as an environmental input. Only energy that is obtained from other, energy producing processes (electricity, thermal heat) should be mentioned as an economic input of energy.

B inputs with proceeds for this process

3.5 waste to be processed: a complete list of all the inputs of waste from other processes (in kg) that is used for this process *'Waste' from other processes that has to be paid for by this process is considered as a 'material'.*

4 ENVIRONMENTAL INPUT

4.1.1-4.1.3 abiotic and biotic resources and energy carriers: a complete list of abiotic (in kg) and biotic (in kg, ...) raw materials and energy carriers (in kg, m^3) that are used as an input for this process

4.2 space: in m^2•year (not in m^2):

$$\text{space use}(m^2\cdot yr) = \frac{\text{material use(kg)}\cdot \text{area}(m^2)}{\text{annual production}(\frac{kg}{yr})}$$

5 ECONOMIC OUTPUT

Outputs are economic if they are the inputs of another production process.

A Outputs with proceeds for this process

5.1-5.4 goods, services, energy, materials: a complete list of the outputs that are produced by this process with proceeds for the process (useful products)

B Outputs with costs for this process

3.5 waste to be processed: a compete list of the outputs of waste (in kg) that are produced by this process and have to be processed in other processes (e.g. waste incineration)

'Waste' that is produced by this process and that is sold to another process is considered as 'material'.

6 ENVIRONMENTAL OUTPUT

6.1-6.3 emissions to air/water/soil: a complete list of emissions to the environment (in kg); if radio-active elements are emitted, isotopes should be specified

6.4 radiation: ionizing radiation to the environment due to sources of radiation within the economic system; to be specified: sort of radiation (α, β, γ/röntgen, neutron); radiation energy per emitted particle (MeV); number of emitted particles per sort of radiation with a certain amount of energy per particle (description of the spectrum). Shielding effects and energy damping effects of shielding materials should be accounted for. If storage conditions of radio-active material do not change, the total radiation emitted until all the material has decayed to non-radio-active elements is calculated from the radio-active series. If storage-conditions do change, or if the radio-active material is eventually emitted to the environment after having been part of the economic system for a certain amount of time, radiation has to be calculated for each time period in which the material has been stored under equal conditions.

6.5 sound: in Pa^2•year (not in dB):

$$\text{noise}(Pa^2\cdot yr) = \frac{\text{material use(kg)}\cdot 4\cdot 10^{-10}(Pa^2)\cdot 10^{(dB/10)}}{\text{annual production}(\frac{kg}{yr})}$$

6.6 heat: waste heat in MJ

6.7 light: in lm•hr

6.8 calamities: in number of victims

7 BALANCES

7.1 mass balance: as a control: the total weight of inputs should correspond with the total weight of outputs; the output/input ratio should equal 1.

7.2 energy balance: ibidem, but difficult to obtain, because energy output (e.g. friction heat) will often not be quantified as a process output.

8 COMMENTS / OTHER INTERVENTIONS

to indicate interventions that cannot be quantified (e.g. use of groundwater, leading to soil dehydration) and general comments about the process

APPENDIX 2
Data sources for the mineral balance

Below an indication is given of how to quantify the various items in the mineral balance. Which figures are ultimately used is highly dependent on the case study being undertaken.

Almost all the data sources listed relate to the Netherlands. Besides the sources mentioned below, the LEI agricultural database may also serve as an important source of information in the Dutch situation. One drawback of this source, though, is that the data are at the enterprise level rather than the process level and that the cost of use may rapidly rise.

INPUT ITEMS

Seed and seedling use

quantity of seeds and seedlings per crop:

- *Handboek voor de Akkerbouw en de Groenteteelt in de Vollegrond*;
- *Kwantitatieve Informatie voor de Akkerbouw en de Groenteteelt in de Vollegrond*.

mineral contents of seeds and seedlings:

- *Project Mineralenboekhouding (1994)*.

Artificial fertilizer use

normative data:

- *Kwantitatieve Informatie voor de Akkerbouw en de Groenteteelt in de Vollegrond* (arable crops and arably grown horticultural crops);
- *Handboek voor de Rundveehouderij (1993)* (grassland and maize fields).

empirical data:

- *Oudendag* et al. *(1995)*;
- *Poppe* et al. *(1994)*.

Manure use

- *Hoogervorst (1991)*;
- *Leneman* et al. *(1995)*;
- *Poppe* et al. *(1994)*.

The quantity of nitrogen in manure can be divided into three fractions:

N_m: mineral nitrogen (directly available)
N_e: organically bound nitrogen (available within a year)
N_r: highly organically bound nitrogen (available in the longer term)

Compost and sludge use

- *Gehalten van mineralen in Project Mineralenboekhouding (1994)*.

Deposition

- *Project Mineralenboekhouding (1994)*.

Nitrogen fixation by legumes

- *Project Mineralenboekhouding (1994)*.

Mineralization

- *Project Mineralenboekhouding (1994)*;
- *Vellinga* et al. *(1993)* (grassland).

Mineralization is highly dependent on regional factors, such as soil type and (for grassland) groundwater level.

OUTPUT ITEMS

Crop harvest

crop yield and crop mineral content:

- *Kwantitatieve Informatie voor de Akkerbouw en de Groenteteelt in de Vollegrond*;
- *Project Mineralenboekhouding (1994)*.

Ammonia volatilization

Ammonia volatilizes from manure, a process that is highly dependent on the method of manure application and ploughing in and on the type of manure. Ammonia volatilization takes place in livestock sheds and during manure storage, but also after application or excretion on farmland. For the soil mineral balance only the latter types of volatilization are relevant. Volatilization from livestock sheds and manure storage facilities is elaborated in the case study on milk. Ammonia volatilization from farmland after manure application can be calculated using the following formula:

$$NH_3\ volatilization = emission\ coefficient \times N_{\min fraction} \times N_{total\ manure}$$

where:

emission coefficient	=	fraction of the directly available nitrogen that volatilizes (with manure, dependent on the method of ploughing in, in combination with the type of manure)
$N_{\text{min fraction}}$	=	fraction of the nitrogen that is directly available to the plant (for artificial fertilizer 100% and for manure dependent on the type of manure)
$N_{\text{total fertilizer}}$	=	total quantity of nitrogen in the manure dressing.

Denitrification

Denitrification is highly location- and study-specific. For this reason no general guidelines can be given for calculations on denitrification.

Run-off to surface water

Run-off is highly location- and study-specific. For this reason no general guidelines can be given for calculations on run-off.

Leaching to groundwater and surface water

Leaching is highly location- and study-specific. For this reason no general guidelines can be given for calculations on leaching.

Accumulation of organic nitrogen and phosphate

- *Handboek voor de Akkerbouw en de Groenteteelt in de Vollegrond*;
  (arable crops and arably grown horticultural crops);
- *Handboek voor de Rundveehouderij (1993)* (grassland and maize fields).

For annual crops a steady-state situation is assumed:

mineralization = accumulation

For perennial crops such as grass the steady-state situation does not apply: with cultivation of these crops there is accumulation of minerals in the soil. This accumulation can be calculated by adding the quantities of highly bound nitrogen in the various types of fertilizer dressings to the nitrogen in crop residues (corrected for ammonia volatilization).

APPENDIX 3
Inputs for the manufacture of farm machinery

The figures in the table below provide a first approximation of the quantities of materials and energy required for the production of 1 kg of farm machinery. However, it is far more recommendable to retrieve figures relating specifically to the machinery under investigation.

Inputs for the manufacture of farm machinery ***per kg machinery***	
Environmental inputs	
– iron ore	1.44 kg
– limestone	0.32 kg
Economic inputs	
– electricity	3.3 kWh
– mineral oil	0.77 l
– diesel oil	0.29 l
– natural gas	0.19 m^3

3: Methodological Background

1 GOAL DEFINITION

1.1 Application of the study

In practice, it is central government agencies and industry that make greatest use of LCA. One of the purposes for which governments employ LCA is to undertake comparative product studies for awarding so-called *ecolabels* to products (by means of which government policy is implemented via critical consumers). Industries are using LCA with growing frequency to analyse their products, generally with a view to product improvement. In many cases the results of such studies are not made public, because they contain sensitive corporate information, or because the company is afraid that process data will go on to lead a life of their own and continue to be used even after the environmental impact of the process in question has meanwhile been reduced. Paradoxically enough, this is in fact one of the reasons that outdated or unreliable process data sometimes have to be used in LCA studies. In the world of LCA the generally poor availability of process data is viewed as a major obstacle.

1.1.1 Possibilities for applying LCA for the agro-eco-label

One of the possible applications of LCA to agricultural products is in the context of the agro-eco-label. LCA can be used in a variety of ways fo this purpose:

- as an assessment criterion for one or more environmental themes in the life cycle of an agricultural product;
- as an assessment criterion for one or more environmental themes in the agricultural part of the life cycle of an agricultural product;
- alongside, or as a refinement of, other assessment criteria.

One of the most essential characteristics of LCA is that it considers both the entire *life cycle* of a product and the entire *environmental spectrum*. This means that any *shifting* of environmental impacts from one part of the life cycle to another and from one environmental theme to another is avoided. In principle, one can opt to use LCA in its pure form for establishing assessment criteria for agricultural products. For each of the environmental themes being assessed a limit value is then set for the impact score, which may not be exceeded. For each of the products to be assessed an *environmental profile* must then be made, by establishing the impact score over the entire life cycle for all the environmental themes. In the environmental profile of potatoes, for example, data are then also incorporated that relate to the production of soil fumigants. This kind of approach is the purest, but also the most laborious.

One possible simplification that can be introduced is to substantially reduce the number of environmental themes assessed. This can be done on a case-by-case basis, on the basis of knowledge concerning the problems associated with the product in question. It may be known that the main problems occurring in the life cycle of the potato, for example, relate to ecotoxicity, acidification and nutrification. The environmental profile can then be limited to these three themes. In doing so there is a risk of problems being shifted to other themes (the greenhouse effect, for instance), and due attention should always be paid to this risk.

An even greater simplification is to introduce a limit on the part of the life cycle to be investigated. In the case of the potato, for example, one might opt to consider potato cultivation only, omitting the production of fertilizer, manure and pesticides. Strictly speaking, it is then no longer a *life cycle* assessment that is being carried out. The LCA method can also yield valuable information even when only a single process is being analysed, however, because the contributions of the process to the various environmental problems can be quantified. Here again, though, there is a risk of problems being shifted to other processes in the life cycle.

1.1.2 Applications of LCA in relation to data sources

The agricultural sector comprises an enormous number of farm enterprises and a parallel number of specific situations. Various sources can be used for gathering average data. In the Dutch context, one such source is the LEI agricultural database, although it can be queried whether this can be taken as being sufficiently representative for this type of data and for the sub-populations that may need to be distinguished. It should be borne in mind that in 'calculating' the environmental burden the result will then be an average, with the environmental score of an individual product deviating positively or negatively from this average. In order to make a meaningful pronouncement on the various products or product systems, the spread around the average will also be relevant: represented, for example, by the environmental score of the best ten enterprises, the worst ten enterprises and the general average for the product (group) in question. By incorporating this spread in the analysis, the reliability of a pronouncement on the environmental burden can be verified. The type of statement referred to here is 'these products (origin, cropping system, type of product, produced in season *X*, etc.) are produced with less environmental impact than those (comparable) products'.
If there is a large overlap between the spread of the environmental impact of the various products, the resultant difference in the (average) environmental score will not hold for a large proportion of the products and, on the basis of this information, the consumer wil run the risk of making the 'wrong' choice when it comes to purchasing an individual product.

Example
In the figure below, with the number of products on the *y*-axis and the environmental impact per product on the *x*-axis, production method A has less environmental impact on average than production method B. Because there is little spread relative to the difference, in the left-hand figure there is far less likelihood of a product produced by method A being less environmentally benign than a product produced by method B than in the case of the right-hand figure. A statement on differences in environmental impact is therefore more definite in the first case than in the second.

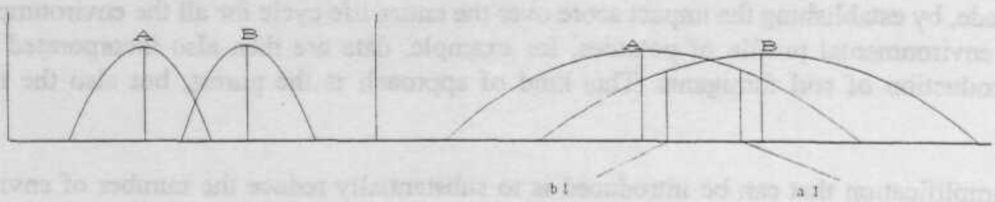

Figure 1.1 Influence of spread on the results of an LCA

In comparing products produced by two different production systems, there are three possibilities with respect to the type of production systems:

1. The production systems already exist in practice;
2. One of the two production systems do not yet exist in practice;
3. Neither production system yet exists in practice.

In each case data should be used that approximate reality as far as possible. If one of the two production systems does not yet exist in practice, an erroneous impression may be gained if the average current situation for the existing production system is compared with the future production system, for which only normative data are available. Whether the wrong impression is obtained depends on how the normative data have been drawn up: do they represent a worst-case approximation, a best approximation of the average situation, or a best-case approximation? In the first and last case a worst-case and best-case situation, respectively, should also be taken when gathering data on the existing production system. When comparing an existing production system with a future production system, the choice of data sources is therefore a matter to which extra attention should be paid.

1.1.3 Application of LCA: a discussion

LCA is an instrument that is still in the process of development. The results of an LCA study performed with the aid of the methods available today should therefore be used with due caution. It should be borne in mind that the modelling is still fairly coarse in some respects, sometimes giving a distorted view of reality. Some people are of the opinion that LCA may not be used for product assessment until development of the method is complete, because the consequences of such an assessment may have a far-reaching impact, even though they may be based on a false view of reality. It is indeed true that, with widespread use being made of LCA, there are no guarantees that an erroneous decision will not be made now and again. However, it should be borne in mind that in practice *not* including the results of an LCA will probably more often lead to wrong decisions being made. The results of an LCA study can be considered as an approximation of reality. Use of these results increases the chances of the right decision being made. LCA thus appears to be a valuable decision-support tool, with which the environmental burden of products can be effectively reduced.

2 INVENTORY ANALYSIS

2.1 The boundary between product system and environmental system

The soil

In the LCA method a sharp distinction is made between the product system (also known as 'the economic system' or 'the economy') and the environmental system. In practice, however, it is not always feasible to draw a sharp dividing line: the two systems overlap to some extent. The question is whether this 'grey' area should be included in an environmental assessment such as LCA, or whether some kind of intermediate solution is perhaps possible.

In the life cycle of arable farming products the soil constitutes part of both the product system and the environmental system. Because LCA is a tool for assessing impacts occurring in the environmental system, the most consistent approach would appear to be to include impacts taking place in the soil in this assessment, too, regardless of the question of whether the soil is indeed part of the product system. However, this implies that the use of substances applied to or in the soil for the purpose of the agricultural process must immediately be classified as an environmental intervention. Some people consider this erroneous, because the use of fertilizers and soil fumigants, say, is a conscious, functional activity inherent to the agricultural process. There are various different options for delineating the boundary between the product and the environmental system in the context of the soil. Here, we elaborate three such options:

- delineation on the basis of agricultural functionality;
- delineation on the basis of the concept of 'multifunctionality';
- delineation on the basis of threats to the soil environment.

In the case of delineation on the basis of agricultural functionality, it is assumed that that part of the soil belonging directly to the agricultural product system does not constitute part of the environment. There may, however, be emissions from that part of the soil to other parts of the soil that do belong to the environment. If this approach is adopted, the boundary between the product system and the environmental system runs straight through the soil. There must therefore be delineation in both the horizontal and the vertical direction.

Delineation in the vertical sense refers to the soil depth that is still considered to be part of the product system: to the depth of the furrow, to the bottom of the rooting zone, to the water table, to a certain depth (1 metre, for example) or completely.

Delineation in the horizontal sense refers to the degree to which field margins such as ditches, waterways, banks, hedgerows and tree lines are taken to belong to the product system or the environmental system.

If (part of) the groundwater and/of surface water in ditches and possibly waterways are classified as part of the product system, it should be realized that not only part of the medium *soil* but also part of the medium *water* are then classified as no longer belonging to the environmental system.

In Dutch government policy sustainable use of the soil is often defined as soil use that does not impair the *multifunctionality* of the soil. Maintaining the multifunctionality of the soil implies that the soil is used in such a way that the soil satisfies the reference values after, at most, two years following discontinuation of this usage. Application of this concept for delineating the boundary between the agricultural product system and the environmental system implies that interventions in the soil are considered to be environmental interventions only to the extent that they still remain an intervention after elapse of two years. For emissions of pesticides, for example, this means that only that fraction of the emission remaining in the soil after two years is taken to be an emission to the soil.

Emissions of hazardous substances to the soil lead to potential degradation of the *soil environment*, including soil ecosystems. The degree to which the soil environment is threatened is not dependent on the agricultural functionality of the emission. For example, use of soil fumigants – which are useful and perhaps even unavoidable for farmers – may have the side-effect of damaging the natural soil ecosystem. If the soil environment is the key issue in deciding whether or not an emission to the soil constitutes an environmental intervention, the question of whether the soil is also part of the (agricultural) product system is irrelevant: every emission that is potentially damaging to the soil environment[6] is taken to be an environmental intervention.

In an environmental assessment such as LCA the question of the extent to which the environment is threatened by human activity should be the key issue – and not the question of whether or not humans have a right to undertake such activity. From an environmental point of view, then, it is only on basis of threat to the soil environment that the boundary between the product system and the environmental system may arguably be delineated.

Delineation on the basis of agricultural functionality proceeds from the basic point of departure that certain emissions should not be taken as environmental interventions because they are functional and unavoidable. The implicit assumption behind such an approach is that threats to or degradation of the environment are, by definition, indefensible and consequently 'wrong'. This is by no means true, however. Continued human habitation of the Earth is inevitably accompanied by a certain degree of threats to and degradation of the environment. This also holds for virtually all human activity, including industrial and agricultural production processes. However, the fact that environmental threats and degradation are, to a certain extent, justified and unavoidable does not mean that such activities may not be viewed as (potentially) damaging to the environment.

In the case of delineation on the basis of the concept of 'multifunctionality' the emphasis is on the irreversibility of an intervention: an intervention is only taken to be an environmental intervention if it is irreversible. An intervention is taken to be reversible if it is no longer perceptible after two years. The frequency with which the intervention occurs is irrelevant. Annual use of a certain pesticide that would lead to a certain continuous concentration in the soil would not therefore constitute an environmental intervention if two years after discontinuation of use no pesticide residues were to remain in the soil. In this option a permanent intervention with permanently detrimental consequences for the soil environment would therefore not always have to be classified as an environmental intervention. However, the fact that the soil is being 'sustainably' used does not automatically mean that this form of usage does not damage the environment. A farmer who uses the soil sustainably acts in a manner that can be socially – and perhaps also environmentally – justified. Social and environmental justi-

6 'Soil environment' is here taken to mean solely the *natural* soil environment. Pest control is not necessarily an environmental intervention, as long as the natural composition of the soil ecosystem is not threatened as a result. In the context of LCA pest control is in fact assessed with regard to its degree of specificity: only the degree to which the natural ecosystem is also threatened alongside the pest organisms plays a role in the assessment.

fication constitutes no guarantee that the environment will not be threatened or damaged in any way, however.

To a certain extent threats to and damage to the environment are unavoidable. The aim of environmental assessment is not to pronounce judgement on these but to minimize them. The key issue in an environmental assessment of agricultural products is not the absolute but the relative degree to which the environment is threatened or degraded ('less is best'). The fact that a certain form of crop cultivation is relatively benign with regard to the local soil environment can be evaluated with regard to its ecological merits only if it is recognized that this soil environment is a permanent part of the environmental system. The soil should consequently be taken in its entirety as part of the environmental system. This is the case only if delineation is based on the threat to the soil environment.

The crop

Harvested crops are clearly part of the product system rather than the environmental system. As a start, then, the most logical approach therefore appears to be not to consider emissions to the harvestable parts of crops as emissions to the environment. In doing so a complication may arise, however: emissions to crops may, as they pass through the product system, ultimately end up in the environmental system, and thus indirectly have an environmental impact.

Via the sewer system or sewage treatment plants, the minerals taken up by crops may end up in surface waters, where they may contribute to the problem of nutrification. Consideration might therefore be given to considering minerals taken up by crops *a priori* as emissions to surface waters. However, the environmental interventions ensuing from application of functional fertilizer dressings would then be set on a par with those resulting from over-fertilization, whereby excess nutrients are transferred directly to surface waters via leaching and run-off. It is debatable whether such an assessment is realistic.

Pesticides adhering to the harvestable parts of crops may ultimately end up in human foodstuffs and thus contribute to the impact 'human toxicity'. The most consistent approach would be to consider 'human food' as a separate environmental medium. Some part of the harvestable part of crops would then again be considered as part of the environmental system, but emissions to this medium would then score solely on the theme 'human toxicity'. Minerals from manure and fertilizer would consequently not partly disappear from the environmental system, but end up in a medium where they would no longer score on the theme 'nutrification'. In addition, such an approach would require a separate exposure model: the quantity of emission impinging on the edible part of the crop will contribute to human exposure to a limited extent only, since pesticides initially emitted to the edible parts of the crop are largely lost as a result of run-off, degradation, rinsing, peeling and cooking. For nitrogen, one would in principle have to make due allowance for the compounds in which this mineral is converted in the crop: some nitrogen compounds – such as nitrate – have a human-toxic impact; others are non-hazardous, or even form an essential part of the human diet. Because it is thus no straightforward matter to model human exposure via the medium 'human food', it is proposed to leave this out of consideration for the time being, and take the crop in its entirety as part of the product system.

Carbon dioxide fixed by the plant might, in principle, be considered as a negative emission. However, one would then have to allow for the fact that this fixation is only temporary: after consumption and digestion of crops, the fixed carbon dioxide is again released. Although there is thus no nett fixation of carbon dioxide, temporary fixation may be beneficial from the environmental viewpoint. This is particularly true of long-term fixation. On a global scale, for example, a substantial quantity of carbon dioxide is kept permanently 'out of the atmosphere' by the existence of rainforests. Until the duration of fixation has been modelled in LCA, temporary fixation cannot yet be properly included in the analysis. It is therefore proposed to leave out of consideration entirely the relatively short-term fixation of carbon dioxide in agricultural crops.

2.2 Delineation of the boundary between relevant and non-relevant processes

2.2.1 Status report

The *LCA Guide* (Heijungs *et al.*, 1992) provides no direct answer to the question of when a process should, in general, be considered relevant. The problem is analysed and a number of general guidelines are given for indications that a process is indeed relevant and consequently cannot be omitted.

It is impossible to provide a rule of thumb as to when which processes can be omitted. As a basic guideline, the *LCA Guide* states that depreciation and maintenance of capital goods may *not* be omitted if the associated costs constitute a substantial fraction of the price of the product in question. The opposite does not hold at all: the price of a product is not proportional to its environmental consequences, and a relatively minor share in the price may certainly be accompanied by a major environmental impact.

An initial indication of the contribution of a process to the environmental impact of a functional unit can be based on expert judgement (Fava *et al.*, 1991). If such judgement is not available, or yields insufficient information, supplementary approaches must be used.

If a rough estimate can be made of the environmental interventions associated with the process that may possibly be omitted, a sensitivity analysis can be employed to estimate the contribution of the process to the environmental impact of the functional unit (Vigon *et al.*, 1993). There are proposals to neglect processes that are anticipated to contribute only slightly (less than 5%, for example) to each of the environmental impacts analysed (Fava *et al.*, 1991; Vigon *et al.*, 1993). In the *LCA Guide* it is remarked that this does not appear to be a satisfactory solution: according to this line of reasoning, once a process tree has been subdivided far enough into sub-processes, *each* of these sub-processes can be omitted. Husseini & Kelly (1994) overcome this problem at the intervention level with their proposal to sum each of the estimated emissions over the whole life cycle and set the requirement, for each of these summed emissions, that the neglected processes together do not exceed a certain percentage of the aggregate emission.

If process emissions are difficult to estimate, an alternative approach is to work with one or more *indicators*. For processes relating to auxiliary raw materials Husseini & Kelly mention two possible indicators: their share of the total mass of the product system, and their share in overall energy requirements. Here, too, they propose a cumulative method: the processes to be included should together have a certain minimum share in the overall mass and the energy requirements. With regard to mass, they propose a threshold of no less than 90%; for energy requirements, they even advise a threshold of at least 95%, because in many life cycles energy consumption is a major contributor to overall environmental impact.

One method often used in practice is to simply omit certain categories of process, such as production and maintenance of capital goods (Vigon *et al.*, 1993).

2.2.2 Discussion

In estimating the relevance of processes with respect to their share in the environmental burden associated with the functional unit over its entire life cycle, expert judgement is particularly important, because a purely methodic approach is by definition impossible. Such judgement may be supplemented with a sensitivity analysis.

Any estimation method that is not based entirely on specific process know-how involves a risk of process-specific contributions being erroneously neglected. This holds particularly for the use of indicators, because these are precisely non-process-specific.

If process-specific know-how forms insufficient grounds for judging whether a process can be omitted, a rough estimate of the environmental interventions of the process followed by a sensitivity analysis appears to the best way of obtaining relatively accurate information. However, decisions on whether or not to neglect certain processes with a relatively minor share in the overall environmental impact of a product should always be based on the summed contributions of all processes taken together. In this way the pitfall is avoided of large numbers of processes being split into sub-processes, each of which makes a negligible contribution but which together are responsible for a significant share in the aggregate environmental impact of the product under investigation. In this context summation at the impact level (*i.e.* after characterization) appears to be a better approach than summation at the intervention level, as proposed by Husseini & Kelly (1994), because some interventions make a far greater contribution to a given environmental impact than others.

A practical problem encountered in using estimates of the environmental interventions of processes coming into consideration for omission is that it is difficult to gain an impression of the percentage contribution of a process to the various environmental problem categories without first analysing the environmental interventions of the process. Once such an analysis has been carried out, the main motive for omitting the process (saving the effort of a laborious analysis) disappears. In practice, therefore, this method will frequently prove unworkable. In that case the use of indicators provides an alternative. In adopting this approach, however, one should always remember that indicators, by definition, do not give a complete picture of process-specific environmental interventions. Use of an indicator brings with it the inherent risk of non-indicator-related environmental interventions being wrongly omitted.

Omission of certain categories of processes, such as production and maintenance of capital goods, is defensible only if it can reasonably be assumed that such processes make only a minor contribution to the environmental impact of a functional unit. An assessment on this point can be based on the results of LCA studies on products similar to the functional unit under investigation, or on specific analyses. Knoepfel (1994) has performed this kind of analysis on the contribution of capital goods to various environmental impacts for several categories of products. One of the results found was that for some environmental impacts the share of capital goods in the environmental burden of transport was over 50%. Production and maintenance of capital goods may therefore never simply be omitted.

2.3 The process data

2.3.1 Desiccation

Desiccation is a new topic in LCA. Practice will have to show the extent to which desiccating interventions are quantifiable. A decision will also have to be made on a case-by-case basis as to whether the theme of *desiccation* should be included in the analysis. If the product under investigation is only a minor contributor to the desiccation problem, it may be opted to omit this theme from analysis. For agricultural products this will not generally be the case.

Because of climatological variation and differences in soil type, evapotranspiration from crops is very location-specific, and due allowance should be made for this fact in the inventory phase. The reference situation need not be determined on a location-specific basis, however: it is sufficient to establish an average level of groundwater abstraction that can be considered non-desiccating. Although the reference level is the same for all crops on a per-hectare basis, this does not mean that the reference can simply be omitted. In the first place, the reference level need not be the same for different crops on the basis of the unit associated with the functional unit (e.g. kg). Secondly, the existence of a reference level has an influence on how the impact scores stand in relation to one another. And thirdly, evapotranspiration constitutes merely one possible desiccating intervention and the score for this intervention should be comparable with the score for other interventions. If the reference were to be omitted, crop evapotranspiration would score too high relative to other interventions, such as the use of tap water.

In Part 2 of this report a fairly arbitrary and pragmatic choice was made to use the level of 1950 as a reference level for evapotranspiration, based partly on the fact that the desiccation problem emerged in the Netherlands only after 1950 and partly on the fact that information on this point is available. However, this does not answer the question of whether this approach is the most appropriate.

Desiccation is a highly regional environmental problem. It can be questioned whether groundwater abstraction should be considered as a desiccating intervention in areas where desiccation does not constitute a problem. As an illustration: in areas where the water table is artificially maintained, such as areas below sea level, there is little logic in speaking of desiccation. In such areas groundwater abstraction may have a negative environmental impact, as a result of groundwater shortages being balanced with an intake of water from elsewhere which may contain nutrifying or otherwise polluting substances. If a consistent approach were to be taken, this would mean classifying groundwater abstraction as a nutrifying intervention in such cases. For the time being, however, we propose to inventory groundwater abstraction as a desiccating intervention regardless of the location.

2.3.2 Ecosystem-damaging interventions

General

Virtually every environmental intervention has a potentially detrimental effect on ecosystems. To the extent that ecosystem degradation is linked to emissions of potentially hazardous substances, inventory thereof has already been covered. Besides emissions, purely physical interventions may also have a damaging effect on ecosystems. As an example, the habitat of certain species may be threatened by physical, spatial changes to the natural environment whereby the natural vegetation is replaced by agricultural land. The degree to which such changes pose a threat to ecosystems depends on how the space is used. Space use will generally pose more of a threat to natural ecosystems as that use becomes more intensive and artificial. Although some forms of land use may lead to the total disappearance of the natural ecosystem, scope is also hereby created for other ecosystems. A case in point is the replacement of forest by pasture, whereby forest birds disappear but meadow birds acquire a new niche. In such cases, too, ecosystem damage may be considered less serious than in the case of disappearance of the natural ecosystem without scope hereby being created for other ecosystems.

Status report

According to the *LCA Guide*, damage to ecosystems and the landscape[7] is caused by a change in the *type of space use*: (more or less) natural systems are replaced by less natural systems. In the inventory phase data are gathered on:

- the absolute extent of the space use (m^2);
- the duration of the space use (year);
- the significance of the space use.

The absolute degree of space use and the duration of the space use for which one functional unit can be held responsible are intimately connected. For example, a form of potato cultivation that yields 60 tonnes of potatoes per hectare can be said to be responsible for the use of 167 m^2 space during one year, equivalent to $5.3 \cdot 10^9 m^2 \cdot s$. The notion of 'space use' thus refers to the combination of land use and time frame, and is expressed in $m^2 \cdot s$. In this case, then, the functional unit '1 tonne of potatoes' is associated with space use amounting to $5.3 \cdot 10^9 m^2 \cdot s$.

The significance of the space use is related to the type of space use for which the process is responsible: a meadow, for example, will generally be able to support a more valuable ecosystem than an airport. In addition, the type of space use prior to the space use being assessed is also of importance: a factory built on a former airport site will cause less damage than a factory built on a former meadow. In the *LCA Guide* five types of ecosystem and landscape are distinguished, following an IUCN report:

- natural systems (type I);
- modified systems (type II);
- cultivated systems (type III);
- built systems (type IV);
- degraded systems (type V).

For a description of each of the types the reader is referred to the *LCA Guide* (Backgrounds, p.36).

According to the *LCA Guide* the inventory should establish, for each process, which type of ecosystem is involved, and which type of ecosystem this has replaced. What is inventoried, then, are *changes* from one type of ecosystem to another. Ten types of change are distinguished: I→II, I→III, I→IV, I→V, II→III, II→IV, II→V, III→IV, III→V and IV→V. Each of these types of space use are inventoried separately in terms of space use. (The reverse changes, leading to environmental improvement, are inventoried as 'negative space use'.) Knoepfel (1995) also employs a method based on changes from one type of land use to another.

As a deviation from the *LCA Guide*, it is here proposed to base the assessment of damage to ecosystems on the *type* of land use in combination with the duration, and not on landscape *changes*. The underlying reason is that landscape changes are one-off occurrences that cannot readily be quantitatively associated with the functional unit.

Discussion

In terms of substance, in analysing physical damage to ecosystems a distinction can be made between one-off changes from one ecosystem category to another, on the one hand, and continuous use within a given category, on the other. An objection to using phase changes for the inventory of ecosystem degradation with the aid of system transitions is that one-off changes cannot be quantitatively related in the inventory to the size of the functional unit. In the *LCA Guide* an attempt is made to perform such

7 In the *LCA Guide* 'damage to ecosystems' and 'damage to the landscape' are evaluated together. The evaluation criteria are based mainly on ecosystem damage, however. Degradation of the landscape value of a given area is associated with aesthetic values and is consequently difficult if not impossible to quantify objectively. For this reason we restrict ourselves here to ecosystem damage.

a quantification. Application of the method proposed there has the undesired consequence that the intervention can in principle be 'bought off' through very short-lived use for a certain purpose, however. For example: a planned fertilizer factory places the future plant site (a meadow: category III) at the disposal of the municipal authority for use as a car park (category IV). The factory (category IV) is subsequently built on a 'former car park' (also category IV). The space use to be attributed to the fertilizer plant during the production phase is therefore zero. The transition from III→IV is allocated to those using the car park. This is an easy way for the fertilizer producer to duck part of his environmental responsibilities and he may score better than his competitors on the theme 'damage to ecosystems', without his fertilizer actually having contributed less to environmental degradation.

The key problem with using changes from one type of ecosystem to another as a basis for equivalency factors for ecosystem damage is that a one-off intervention (some degree of destruction of an ecosystem) is allocated to an undefined quantity of product. In the context of LCA, therefore, landscape changes can be included in a qualitative sense only. One way of doing this is to place a 'flag' (qualitative remark) if a change occurs that can be assigned directly to the product in question.

Continuous use of land can, in principle, be linked quantitatively to the functional unit. In the inventory phase, land use can be expressed in *ha* × *year*, with a (qualitative) indication of the type of land use. In the characterization phase a weighting factor can then be allocated to this type of land use[8], to arrive at a simple impact score.

In the IUCN scheme, farmland comes under category III of the five types of ecosystem. In comparing various methods of agricultural production, such as extensive versus intensive cropping, this categorization is consequently too coarse: it is desirable to have a means of rendering visible improvements within category III (use as farmland). Preferably, various different equivalency factors should be developed for various types of ecosystem within category III, in order to refine the assessment. Analogously, refinements might also be introduced in the other categories.

In the context of evaluating ecosystem damage due to agriculture, a three-stage procedure – from coarse to fine – would appear to hold the greatest promise:

1. categorization in one of the five IUCN categories;
2. within category III: categorization in terms of crop;
3. per crop: categorization on the basis of cropping intensity.

In the inventory phase the first two steps pose no problem: they represent no more than a qualitative description of the land use. (In the characterization phase *quantitative* weighting factors can be assigned to such *qualitative* descriptions, allowing a quantitative evaluation to be made in the form of an impact score.)

The third step requires more thought. If the inventory were to be limited to the first two steps, there would be no difference in the inventory phase between a hectare of an intensively cultivated crop and a hectare of an extensively cultivated crop. Converted to a mass basis, this would mean that one kilo

[8] Damage to ecosystems due to continuous use of land is thus tied directly to the use of land surface. In the context of LCA this comes under the category of *resource consumption*, not *damage*. Three kinds of resources can be distinguished: deposits (non-renewable stocks that can be depleted), funds (stocks that can be augmented and that may either be used sustainably or become depleted) and flows (stocks for which the level of augmentation is independent of the intensity of use and that cannot therefore become depleted). An example of the last category is sunlight: even if all the available sunlight were to be used, the rate of augmentation remains unchanged. Use of land surface is an example of a flow: the surface itself is not lost through utilization and remains available for ever in principle.

of an extensively grown crop would involve a greater intervention than one kilo of an intensively grown crop, because one kilo of extensively grown crop requires more space (ha x year) than one kilo of intensively grown crop. A quantification of this nature is intuitively erroneous: one hectare of extensively grown crop occupies the space less exclusively than one hectare of intensively grown crop, thus leaving more (physical) space for ecosystems. It is this difference that should be quantified in the third step.

If one assumes that the extra space (farmland) *utilized* by one kilo of extensively grown crop and the extra space (for ecosystems) created by this crop (between the plants) compensate one another, the ecosystem-damaging interventions of the two methods of cultivation should be equal, on a kilo-for-kilo basis. In that case, the *physical yield* (mass) per hectare is the most appropriate yardstick for cropping intensity.

If one wishes to incorporate other parameters besides the difference in density between intensive and extensive cropping in the assessment, matters become more complicated. For example, extensive cropping may be associated with a greater average number of tractor-kilometres driven per hectare of crop (and thus, to a greater degree, per kilo of crop) than intensive cropping. In this sense, the 'extensive cropping' may thus be *more intensive* than intensive cropping! Because more tractor-kilometres increase the risk of physical degradation of ecosystems, through disturbance or damage, the 'degree of intensity' should increase with the number of tractor-kilometres.

It is possible, in principle, to chart further physical differences between intensive and extensive cultivation and construct, from all these differences, a single factor to serve as a yardstick for the intensity of a given cropping system in relationship to the physical damage to ecosystems. While this may be the most accurate approach, the other side of the coin is that it is also extremely laborious. The question can be posed whether it might not be feasible to adopt a simplified approach, by taking an (indirect) *indicator* for the intensity rather than a 'true' yardstick. An indicator of this kind would have to provide a good reflection of the relationship between the intensity of cultivation and the physical damage to ecosystems, without all the factors involved in this relationship having to be individually recognizable. In elaborating the indicator due attention would have to be paid to the fact that it should relate exclusively to the *physical* intensity of cultivation, i.e. excluding the intensity of *chemical* operations, such as fertilizer and pesticide use, because in LCA the effects on ecosystems associated with chemical operations are already taken into account in relation to other environmental themes. Use of an *effect* indicator, such as a measure of the nature value (Buys, 1994), does not therefore appear suitable for this purpose. The search must consequently be for an indicator of *pressure* that is representative for the relationship between the cropping intensity and the physical pressure on ecosystems and that incorporates no superfluous elements. Indicators incorporating the price of products are not suitable, since price bears no relationship to the intensity of production.

2.3.3 Physical degradation of the soil

Soil degradation can be evaluated with respect to three sets of properties:
- chemical soil properties;
- physical soil properties;
- biological soil properties.

These three sets of properties are obviously interrelated. We shall discuss each set in turn and indicate how degradation of these properties can be accounted for in LCA.

Degradation of chemical soil properties
Degradation of chemical soil properties is already partly included in LCA. Inputs and outputs of fertilizers and other chemicals are already inventoried and characterized/classified. What are left are such aspects as alteration of soil acidity (pH) due to cultivation procedures. To account for these, new equivalency factors would have to be developed in LCA. If farmers take measures to counter this acidity, this should feature as a negative score (= positive for the environment) on this theme.

Degradation of physical soil properties
This item should include such aspects as:
- degradation of the soil structure;
- degradation of the soil profile;
- degradation of the relief;
- soil erosion and displacement;
- soil subsidence.

Degradation of the soil structure
Degradation of the soil structure is easy to describe from the agricultural angle. What soil degradation involves from an environmental point of view is less clear, however. Three options for classifying and characterizing degradation of the soil structure are discussed below:
- pressure and pressure distribution;
- density;
- structure index.

Pressure and pressure distribution
Degradation of the soil structure due to physical/mechanical interventions (heavy machinery) leads to soil compaction. The pressure distribution on the soil can be calculated on the basis of machinery weight, tyre pressure and width and soil water content. It can then be derived how many square metres have been subjected to a given pressure. By setting a limit value for the pressure, it can be established how many square metres have been subjected to a pressure exceeding the limit value.

Density
The mineral portion of the three soil components clay, sand and loam each have a density of 2.65 g/cm^3. Nonetheless, undisturbed soil (dry weight) may exhibit considerable variation in density. This variation is due to the amount of air in the soil (the pore volume) and the fraction of organic matter. A good soil structure is generally characterized by a high pore index and a high level of organic matter. Cropping procedures resulting in an increase in soil density could thus be classified and characterized as having an environmental impact. The opposite holds for cropping procedures leading to a decrease in soil density (loosening up compacted ruts, for example). The result would then be the nett quantity (m^3) of compacted soil.

Structure index
Assessment may be extended beyond pressure and density. In soil science the soil structure of farmland is often indicated in terms of a structure index: a measure of the looseness of the soil. On the basis of a visual assessment the soil is then assigned to a certain class.

Degradation of the soil profile
The options for degradation of the soil profile are very furrow-oriented and thus agriculturally oriented. From the environmental point of view it is perhaps more interesting to consider the entire soil profile. This may be relevant in soil protection areas. Examples of degradation of the soil profile are deep subsoil ploughing, rupture of impermeable layers and sanding over bulb-growing fields. Degradation of the soil profile might be assessed in classes varying from no to serious degradation.

Degradation of the relief

Besides degradation of the soil profile, superficial degradation might also be included. Here, too, assessment can be in terms of classes.

Soil erosion and displacement

Erosion can be measured as the quantity of eroded soil (m^3). Data are available for various cropping systems, *inter alia* from an anti-erosion standardization study carried out in the Dutch province of Limburg. For water erosion, the water flow rate and the absorption capacity of the soil are important. For wind erosion, the wind speed is important. In practice research data need to be converted to potential erosion.

Soil subsidence

Soil subsidence is most familiar in the vicinity of operational gas fields. In the agricultural context, soil settling or shrinkage is the most relevant phenomenon. As a result of drainage clay, and to a far greater extent peat, shrink in volume. Some Dutch polders have settled by 50-100 cm over the past 100 years. Here, due allowance should be made for the overlap with the theme of desiccation.

Degradation of biological soil properties

The numbers and diversity of soil organisms vary considerably with soil type. The following key factors are of influence in this context:
- availability of nutrients, particularly organic matter;
- moisture;
- acidity;
- soil tillage;
- presence of toxic substances: pesticides, heavy metals, etc.

Degradation of biological soil properties due to application of toxic substances is indicated in LCA in the score for terrestrial ecotoxicity. Cropping procedures may have either a positive or a negative effect on soil life.

2.4 Allocation of environmental interventions

2.4.1 Allocation in the case of co-production – status report

According to the *LCA Guide* there are in principle two ways to divide processes occurring within a combined production process over the products produced:

1. causal allocation;
2. overall apportioned allocation.

Causal allocation proceeds from physical causality. Physical causality generally relates to individual processes within a combined production process: some processes are performed for all of the products produced, while others relate specifically to one single product. The obvious approach is to allocate the latter type of process solely to the product to which it relates. A case in point is straw-baling in the combined production of wheat and straw: this process is performed solely for the straw, and can therefore be causally allocated to the straw.

If there is no physical causal relationship between products and emissions, certain rules will have to be employed to apportion the emissions to the various co-products by means of *overall apportioned allocation*. There are various options available. According to the *LCA Guide* in this type of allocation the function is the key issue at stake, as a measure of the *social causality* of the process by the product in question. The notion of function can be quantified in a variety of ways, depending on the nature of the process, thus providing a variety of possibilities for drawing up a basis for allocation. Examples include physical quantities such as mass, area or number of units, and economic quantities.

There has been considerable international debate on which method of allocation is the most correct. As yet there is no international consensus on this point. The various standpoints will therefore now be discussed individually.

First of all, it is necessary to define and delineate the term 'allocation', which is not seen equally broadly by all. In the broad sense of the term, allocation encompasses everything involved in apportioning the right environmental interventions in the right quantities to the right products. In its narrow sense, allocation relates solely to apportioning the environmental interventions of indivisible processes to the outputs of those processes. In this narrow view, what is described as 'causal allocation' in the *LCA Guide* is not allocation at all in many cases, but is concerned precisely with avoiding allocation. Because the present document is primarily a supplement to the *LCA Guide*, here we shall employ the broad definition, with reference being made to the narrow definition where appropriate.

The following forms of allocation are distinguished:

1. causal allocation on a physical, chemical or technical basis;
2. causal allocation on a social basis (generally quantified as economic value);
3. allocation on the basis of physical quantities with no reference to natural causality (mass, volume, area, energy, exergy, number of molecules);
4. allocation on the basis of an arbitrary numerical distribution.

In addition, a method has been developed to avoid allocation, by widening the system boundaries of the product system under investigation; this method is sometimes referred to as the 'substitution method'.

Most people are generally agreed that preference should be given to *causal allocation on a physical, chemical or technical basis*, even though this is not always referred to as allocation. Vigon *et al.* (1993) propose splitting off as many subprocesses as possible prior to the allocation procedure, to avoid allocation. Huppes (1993 and 1994a) makes a distinction between splitting off single subprocesses and splitting off subprocesses contributing to one and the same quantifiable function. The former category of subprocesses are assigned to the process to which they relate: a case in point is the compression of chlorine in the combined production process for chlorine and caustic soda. Huppes does not term this allocation. Subprocesses that have been split off as contributing to one and the same quantifiable function are allocated on the basis of the share in this function of each of the co-products. An example is allocation of operation of a packaging machine to the packaging of several different dairy products (milk and yoghurt, for example) in identical packages. This is termed 'allocation on the basis of function'; Huppes (1993 and 1994a) takes 'physical causal allocation' to mean a different type of allocation that is relevant only in the context of recycling and combined waste incineration.

Allocation on the basis of a form of economic value was first used by Basler & Hofman (1974). This method is the subject of heated international debate. Many advocates of this approach see it as a way of quantifying the *social causality* of a process: a process is carried out to create socially desired products, and the economic value is a measure of this social desire (Assies, 1992; Pedersen & Christiansen, 1992; Huppes, 1993 and 1994a and b; Clift, 1994; Finnveden, 1994a and b; Vrije Universiteit Brussel, 1994). Others make no explicit mention of social causality, but cite the argument that the economic value of the manufactured products is the ultimate driving force behind every economic process (Udo de Haes, 1993a; Ekvall, 1994). A final argument cited for allocation on an economic basis is that only in that case is the distinction thus made between co-products consistent with the difference between products and waste, which also rests on economic grounds (Vrije Universiteit Brussel, 1994; Guinée, 1995).

A widely recognized drawback of using the economic value as the basis for allocation is that this value may be subject to major fluctuation (Ekvall *et al.*, 1992; Huppes, 1992; Belgian Biomass Association, 1993; Frischknecht *et al.*, 1993; Huppes, 1993; Vigon *et al.*, 1993; Ekvall, 1994; Henshaw, 1994; Huppes, 1994a). Some opponents of allocation on an economic basis consider this drawback so serious that they employ it as the main argument against this form of allocation (Boustead, 1994; Schricker & Goldhan, 1994). One practical solution to this problem may be to work with average market prices over a prior period of, say, five years.

Finnveden (1994a) holds the view that economic criteria can also be used as an allocation basis without allocation being based on prices: if the value is grounded mainly in, say, mass, volume or energy, these criteria can be considered as indicators for the economic value and used as an allocation basis.

Allocation on the basis of physical quantities without reference to natural causality is the method most frequently used in practice if physical causal allocation is unfeasible. It is generally then opted to base allocation on mass. Among its advantages, advocates of this method cite the fact that it is easy to use and yields stable results, because physical quantities do not vary with time (Vigon *et al.*, 1993; Boustead, 1994). It is also stated that these quantities are related to the chemistry and physics of the process (Consoli *et al.*, 1993) and that allocation on the basis of mass is a widely accepted practice (Vigon *et al.*, 1993). In many documents mass is recommended as an allocation basis without any argumentation, however (Fava *et al.*, 1991; Keoleian, 1993; Klöpffer, 1994). Opponents point out the arbitrary nature of the choice of a physical allocation basis in cases where there is absolutely no causal relationship (Huppes, 1992; Finnveden, 1994a). This is illustrated by the fact that even a choice for mass as an allocation basis is by no means unequivocal: in the case of the chlorine/caustic soda process, for example, it makes a considerable difference whether allocation is based on the mass of sodium, the mass of sodium hydroxide, or the mass of sodium hydroxide including water of crystallization (Huppes, 1992). It may be remarked that it is better to use an unstable measure of a relevant variable than a stable measure of a variable with little or no relevance (Huppes, 1992; Ekvall, 1994).

Both advocates and opponents of allocation on the basis of physical quantities without reference to natural causality point out that allocation on the basis of mass can lead to undesired results if the mass of the main product is relatively low compared with the mass of the by-products, or, phrased differently, if there is no good correlation between the chosen physical property and the economic value (Assies, 1992; Ekvall *et al.*, 1992; Huppes, 1992; Henshaw, 1994; Klöpffer, 1994; Udo de Haes, 1993a). In such cases it is sometimes recommended to make an exception to allocation on the basis of physical aspects, and employ allocation on the basis of economic value (Henshaw, 1994; Klöpffer, 1994). Udo de Haes (1993a) writes: 'In terms of principle, then, allocation on the basis of economic value can be said to be the most accurate approach'.

Allocation on the basis of an arbitrary numerical distribution is not often proposed. In practice one sometimes sees a combined production process being allocated in its entirety to the main product. This is viewed as a method that can be employed as a first approximation in so-called 'screening LCAs' (*i.e.* rough, abridged LCA studies) (Huppes, 1994b; Running, 1994).

Avoidance of allocation is feasible by extending the system boundaries: if product A is to be compared with product A', and B is a co-product of production of A (but not of A') one looks for a dedicated production process for B and takes the environmental interventions of this process as the environmental interventions to be allocated to B in the combined production process for A + B. The underlying reasoning is that production of B as a by-product of A generates a quantity of B that no longer needs be produced in a separate process. Some fraction of the dedicated production of B is thus avoided, and with it the associated environmental interventions. It thus seems justified to deduct these avoided environmental interventions from those of the combined production process of A + B, or add them to the environmental interventions of the production process for A'. In the latter case the comparison of A with A' has been extended to a comparison of 'A + B' with 'A'' + 'B'. In principle, the two approaches come down to one and the same thing. In the LCA context this method was first proposed by Heintz & Baisnée (1992), was elaborated further by Ekvall *et al.* (1992) and Tillman *et al.* (1994) and is also advocated, *inter alia*, by Pedersen Weidema (1993). Outside the LCA context, too, one encounters this approach in assessment methods (Belgian Biomass Association, 1993). The method can only be applied if the by-product forms a real substitute for a product that is normally produced in a dedicated process. One drawback of the method is that the system to be studied will often become too large and complex to work with. For this reason it is proposed to combine this method with allocation (Ekvall *et al.*, 1992; Ekvall, 1994; Tillman *et al.*, 1994).

Within the SETAC (Society for Environmental Toxicology And Chemistry) – the principal international forum for LCA – opinions are divided on the allocation of indivisible processes. The most recent discussions appear to be leading towards allocation on the basis of socio-economic causality (Clift, 1994; Finnveden, 1994b; Huppes, 1994b). In the American and Canadian literature, particularly, the case is still sometimes argued for allocation on the basis of mass, however (Keoleian & Menerey, 1993; Vigon *et al.*, 1993; Husseini & Kelly, 1994). The 'Code of Practice' drawn up by the SETAC also refers solely to allocation on the basis of physical quantities (Consoli, 1993). In a working group set up at a UNEP conference on 'Life Cycle Assessment and its Applications' organized in 1993, however, it was explicitly stated that the omission of economic criteria as a basis for allocation in the case of co-production in this 'Code of Practice' was in error (Udo de Haes, 1993b). There are few if any reports of international discussions on avoiding allocation by substitution of by-products.

Because there is currently no internationally agreed universal procedure for allocation in the case of co-production, some authors stress that decisions will have to be made on a case-by-case basis for the time being, opting for the most appropriate allocation method for the particular circumstances (Clift, 1994; Rønning, 1994). Others propose a stepwise procedure, whereby various allocation methods are assessed for their suitability under the circumstances in order of preference (Pedersen Weidema, 1993; Finnveden, 1994a; Huppes, 1994a).

Given the lack of a universally accepted allocation method, for the time being sensitivity analyses will have to be used to determine the influence of the choice of method made.

2.4.2 Allocation in the case of co-production – discussion

The key issue involved in allocating an indivisible multi-output process is to establish the extent to which each of the co-products can be held responsible for the aggregate environmental interventions of the process (and upstream processes). The art is to find an as good as possible measure for the degree of responsibility. What is 'good' in this respect cannot be determined objectively, however: it depends on what one means by the term 'responsibility'. In most discussions the notions of 'physical' and 'economic' stand in opposition to one another in assigning responsibility. It is therefore useful to subject these two notions as a yardstick for 'responsibility' to further analysis.

In arguing for allocation on the basis of physical quantities it is stated that such values are an optimum reflection of the physico-chemical reality of the process in question. In arguing for allocation on the basis of economic value it is stated that this value is an optimum reflection of the social causality of the process. The first question that can then be posed is which concept constitutes a better measure for 'responsibility': physico-chemical reality or social causality? By referring to an example a better impression can be gained of the concrete elaboration of these concepts.

Example
We analyse the co-production of meat and manure in the process 'pig farming', with the aim of ascertaining what share of the environmental interventions of pig farming should be allocated to the meat and what share to the manure.

First of all we endeavour to find a physical measure that forms as true a reflection as possible of the physico-chemical reality of the process. Assume that a pork pig consumes 750 kg of fodder a year and supplies 250 kg biomass (mainly meat) and 1250 kg of (economically marketable) manure over the same period. This manure originates not only from the fodder, but also from drinking water and leakage and cleaning water and so on. It cannot be ascertained what mass of manure is related to what quantity of consumed fodder. The notion of 'wet mass' cannot therefore be used as a yardstick for the physico-chemical reality of the process. In the case of 'dry mass' the situation is different. We proceed from the approximation that the dry-matter content (d.m.) of pig fodder is 90% and that pork has a dry-matter content of 20%. It is known that the dry-matter content of the manure is 10% on average. This means that 675 kg (d.m.) pig fodder generates 50 kg (d.m.) meat and 125 kg (d.m.) manure a year. It follows that of the 675 kg available dry matter, 50 kg is converted into meat and 125 kg into manure. The remaining 500 kg of dry matter has been used for the pig's catabolism and has therefore been converted into carbon dioxide and water. These quantities reflect (an aspect of) the physico-chemical reality of the process. Allocation is to meat and manure. If we taken the mass of dry matter as the point of departure, we should allocate 50/(50 + 125) x 100% = 29% of the environmental interventions to the meat and 125/(50 + 125) x 100% = 71% to the manure. The identified masses and dry-matter contents can be readily measured, and the result of this allocation method is fairly stable with time: as long as the process technology remains unaltered, so too does the allocation method.

Now we try to find a measure for the social causality of the process 'pig farming'. It can obviously be said that pigs are kept exclusively for their meat and that manure can at best be considered a 'useful by-product'. In that case all the environmental interventions should be allocated to the meat. The allocation yardstick is then intuitive and based on the general fact that in our society meat is far more 'socially desirable' as a product than manure. A correcter approach is to quantify this social desire. For convenience, we here take the market price as our point of departure. We assume a market price of approx. Dfl. 15.00 per kg for pork and a price of approx. Dfl. 0.003 per kg for manure. Per kg dry matter this corresponds to Dfl. 75.00 for the meat and Dfl. 0.03 for the manure, a ratio of 2500:1 for meat:manure. In itself the market price can be fairly readily measured. It is not stable, though: if demand for pork drops sharply and demand for manure increases, the relative social causality of the two products changes.

Although the physico-chemical reality is often easy to measure and stable, this does not appear to imply that it is a good reflection of the (partial) responsibility of a given output for the occurrence of a production process. Allocation on the basis of physical quantities, with no reference to any form of causality, therefore appears indefensible under all circumstances. If there is no physical causality, one should therefore seek to find some measure of social causality. Social causality cannot always be quantified unequivocally: for example, there is a difference between the social valuation – expressed as turnover (= market price times yield (in, say, guilders)) – and the economic motive – expressed as nett profit, even though these two notions are related. Sometimes neither the pure market price nor the profit can be accurately determined – in the case of subsidies, for example. In that case a physical quantity may sometimes be a better indicator of social causality than the price. This holds only for co-products for which the social desirability is reflected in a common physical quantity, for example the energy content of co-produced fuels with a similar utility value.

Social valuation of a product is subject to change. This means that the proportional allocation of multi-output processes may vary with time. This is not a weak, but rather a strong point of allocation methods on this basis, for it indicates that partial responsibilities of co-products may change with time: if one co-product becomes more expensive relative to another, the production process is carried out more for the former co-product. For example: if pig manure should suddenly prove to be a source of gold, pigs would no longer be kept for their meat, but for their manure. Rightly, the bulk of the environmental interventions would then be allocated to the manure, even though the physical reality of the process remains unaltered.

Besides allocation on the basis of social causality, substitution of environmental interventions by the interventions of a comparable co-product is also a good option. One requirement here is that the co-product in question must form a true substitute, in social terms, for an already existing product, production of which is more common by means of a dedicated process rather than as a co-product. (Although it should be added that this will frequently be a difficult matter to establish.) In using this method it will have to be decided on a case-by-case basis whether the system does not grow so large as to become unworkable.

2.4.3 Allocation in the case of recycling

Recycling implies that two or more product systems are linked: wastes from one product system serve – possibly after some form of upgrading – as raw materials for a subsequent product system. In general, there are then three kinds of processes that must be allocated:

- processes associated with the extraction and processing of the primary raw materials;
- processes associated with the processing of the final waste product;
- processes on the borderline between the two product systems, associated with the processing of a waste product from the first process to a product of value for the subsequent process.

When recycling is involved, there will frequently be two product systems: a primary product system, which yields recyclable waste, and a secondary product system, which makes use of secondary raw materials recovered from the primary product system. In principle, however, three or more product systems may also be thus interlinked. This is referred to as a cascade of product systems.

In principle, two kinds of allocation are conceivable for the environmental interventions associated with the extraction and processing of primary raw materials:

- full allocation to the primary product in the cascade;
- division of the environmental interventions over all the products comprising the cascade.

Similarly, for the environmental interventions associated with the processing of the final product, the following kinds of allocation can be applied:

- full allocation to the final product in the cascade;
- division of the environmental interventions over all the products comprising the cascade.

For (upgrading) processes on the borderline between two product systems, finally, three modes of allocation can be distinguished:

- full allocation to the product generating the recyclable waste;
- full allocation to the product using the recycled material as a raw material;
- division of the environmental interventions over the product generating the recyclable waste and the product using (upgraded) recycled material as a raw material.

In the *LCA Guide* the following allocation rules are used in the case of recycling:

- environmental interventions associated with the extraction and processing of the primary raw materials are allocated to the primary product in the cascade;

- environmental interventions associated with the processing of the final waste product are allocated to the final product in the cascade;
- environmental interventions associated with the (upgrading) processes on the borderline between two product systems are preferably allocated to the product using the (upgraded) recycled material as a raw material.

With regard to (upgrading) processes on the borderline between two product systems, Udo de Haes (1993a) proposes dividing the environmental interventions over the two products in question on the basis of the so-called *economic turning point*. The economic turning point is the point where the value of the material to be recycled switches from negative to positive: with recycling, a waste product (with a negative economic value) is worked up to a raw material (with a positive economic value). Processes occurring prior to the economic turning point are allocated to the primary product; subsequent processes are allocated to the secondary product. In this approach collection will frequently be allocated to the primary product: it is then in fact considered to be a form of 'waste processing'.

It is often a difficult matter to establish the precise position of the economic turning point, because it occurs in the middle of a process, for example. In such cases it is proposed to allocate the process in which the turning point occurs on the basis of the 'overall apportioned allocation' method. One way to do this is on the basis of the ratio between the (negative) economic value of the material entering the process and the (positive) economic value of the material leaving the process. This method leads to problems if the economic value of the input and/or output cannot be adequately determined, however, for example in cases where the upgrading process is subsidized by the government.

3 CLASSIFICATION & CHARACTERIZATION

3.1 Processes in the environment

3.1.1 General

In the LCA method emissions of hazardous substances are evaluated for their potential harmfulness for the environment. The step in the LCA method in which emissions of hazardous substances are linked to environmental impacts is termed *characterization*.

The potential environmental hazard of a substance depends on the environmental medium in which the substance is present. Copper in surface waters, for example, has completely different effects from copper in the soil. In evaluating an emission of copper, LCA makes due allowance for this difference: an emission of a given quantity of copper to surface waters is evaluated differently from an emission of the same quantity of copper to the soil.

In the first place emission of a hazardous substance to a given environmental medium generally leads to direct impacts within the same medium. Because substances virtually always undergo processes in the environment, other impacts may also occur, however, and the direct impacts will usually disappear in the course of time. The main environmental processes are as follows:

- degradation to non-hazardous waste products: the substance slowly disappears and consequently ceases to cause environmental impacts;
- conversion to substances that are also hazardous: the substance leads indirectly to environmental impacts in the same environmental medium or in other environmental media;
- migration to other environmental media: the substance leads indirectly to environmental impacts in other environmental media;
- triggering of chemical reactions, catalysis and other processes involving other substances: the substance leads indirectly to environmental impacts in the same environmental medium or in other environmental media;
- reabsorption of the substance in a product system: the substance disappears from the environment and consequently ceases to cause environmental impacts.

The *LCA Guide* makes no allowance for most of these processes. With the aid of the USES model developed by the Dutch National Institute of Public Health and Environmental Protection RIVM (RIVM, VROM, WVC, 1994) for organic substances a model is developed here for incorporating cross-media transport and degradation within the various environmental media.

3.1.2 Modelling potential impacts with the USES model

General

In the current LCA method (Heijungs *et al.*, 1992) cross-media transport and degradation of emitted substances are not incorporated in the impact assessment. In general terms, it can be said that this is a complex area of research that is only just starting to be explored. In practice the main problems in this area are anticipated with studies in which emissions are being assessed of substances exhibiting major differences in terms of transport and degradability. Because this is the case for pesticides, for example, in the context of agricultural LCAs there is a relatively urgent need for modelling these issues. For this reason initial steps in this direction have been made in the present supplement to the *LCA Guide*. It must be stressed that these are merely initial steps; developments in this field are still in full swing. Although the model described here is still of a very provisional nature, it is operational in principle. For a number of pesticides new equivalency factors for human toxicity and aquatic and terrestrial ecotoxicity could therefore be calculated. It should be stressed that the software developed for this purpose could not be validated within the context of the present project, and there may therefore still be some errors. Consequently, this software is not yet available. It is emphasized that the new equivalency factors should be seen as merely indicative.

Use of a risk evaluation model for LCA

General

In risk analysis, models are used for simulating transport and degradation of substances in the environment. In principle these same models can also be used in LCA. In doing so, however, it should be borne in mind that LCA differs fundamentally from risk analysis in a number of respects.

The difference between LCA and risk analysis

In an LCA the emissions resulting from the production of a functional unit of product are assessed for their (potential) harmfulness. The production of a functional unit of product does not cause a continuous emission flux, but a single, momentary emission pulse. An emission pulse can of course be seen as contributing to the total annual emission flux. A functional unit of product thus contributes to the annual risk associated with the total emission flux. If a factory has an annual output of, say, 365 functional units of product, one functional unit can be held responsible for a share of one day per year (or 4 minutes per day, or 10 seconds per hour) in the overall (continuous) risk due to this factory.

The life cycle of a product frequently involves a very large number of processes. Each of these processes is accompanied by certain risks, and the functional unit contributes to each of these risks. The magnitude of this contribution differs from process to process. For example, electrical power will often be required for the production of a functional unit. In most cases, however, the contribution of a functional unit to the overall risks associated with a power station will be very small (0.0000000001 second per year, for example).

Because of the multitude of processes encountered in an LCA, in performing a risk analysis of a functional unit of product an extremely complicated picture would be obtained of a multitude of contributions to a multitude of different risks. In practice, this is simply not practicable. In LCA, therefore, an entirely different approach is adopted from that employed in risk analysis: the key issue here is not the *concentration* of the substance in the environment, but the *quantity* of substance emitted.

In contrast to concentrations, quantities can be summed. In practice, the emissions of a given substance for which a functional unit of product can be held responsible throughout its entire life cycle are aggregated. This is done for each individual substance. The outcome of this exercise is an *intervention table*. (The intervention table also comprises a list of extraction processes, and non-substance data such as the production of noise and heat.)

The emission of a certain quantity of given substance is termed an *intervention*. An example of an intervention is *the emission of 5 kg* SO_2. An intervention from the intervention table is generally made up of a large number of contributions from various processes in the life cycle (which are in turn also termed interventions). An intervention from the intervention table thus seldom occurs at one single location, and it is therefore obviously impossible to establish a direct link between such an intervention and a certain environmental concentration. This means that an intervention from the intervention table cannot generally be related to a certain risk. *The outcome of an LCA thus has no bearing on risks!*

In a risk analysis, the question of whether the NEC (no effect concentration) is exceeded plays an important role. Because an intervention from an LCA intervention table is not related to a certain concentration, there is no question of the NEC being exceeded or not. Risk analysis is often based on the premise that there is no risk as long as the NEC is not exceeded. In practice, this means that risks can be eliminated by dilution, since this lowers the concentration. In LCA it is not the risk of a given emission that is assessed, but the contribution that an emission makes to various current environmental problems. Every emission of a hazardous substance is viewed as making its own contribution to these problems, regardless of whether this emission results in a threshold value being exceeded. Dilution makes no difference to the assessment: *dilution is no solution for pollution.*

The USES risk assessment model

In 1994 RIVM launched the USES risk assessment model (RIVM, VROM, WVC, 1994). This model can be used to calculate the relationship between the magnitude of a continuous emission flux to a given environmental medium and the concentrations in that environmental medium and in other, successive media. The model assumes a steady-state situation: the concentrations in the various environmental media are assumed to be constant.

To establish the concentration of a substance in a given environmental medium following from an emission to another environmental medium requires models of the transport processes occurring between the various media. These models have been implemented in USES, making use, *inter alia*, of multimedia environmental models such as those developed by Mackay (1991).

The USES model is a tool that can be used to evaluate the risks associated with given emission fluxes. With regard to ecotoxicity, these risks are related directly to the concentrations in the various environmental media. For calculating human-toxicological risks, use should be made of an exposure model linking the concentrations in the various environmental media to the direct or indirect intake from each of these media.

The USES model is actually a combination of several environmental models, a number of which had already been published by RIVM. More specifically, the USES model combines emission models, diffusion models, exposure models, effect models and risk-assessment models. These model categories are embodied in five modules. (The USES model also contains a module for data input.) No use has been made of the emission model. To model cross-media transport and degradation for the purposes of LCA the diffusion model has been employed. In addition, the modules for exposure and effect are of interest for modelling the human-toxicological and ecotoxicological impacts.

Emission

In the USES computer model emissions cannot be input directly. Instead, an indication must be given of the branch of industry and type of product involved. The associated emissions are then computed internally on the basis of an extensively modelled database. This means that there is only a limited number of products for which USES can perform calculations: the user must make a selection from a standard list. Neither is the user at full liberty to deviate from the modelling assumptions incorporated in the model. Such a model is not sufficiently flexible for the purposes of LCA. The model has therefore been adapted to allow emissions to be input rather than an indication of the type of product. In other words, while in the original USES model data on the production process are input, the adapted model requires direct input of emissions.

Diffusion

For calculating environmental concentrations of emitted substances, the USES model makes use of three models: two local models and one regional/continental model. In the context of LCA, it is the regional/continental model – named SimpleBox – that is of interest.

The USES model is concerned with modelling the risks associated with the toxicity of emissions. In principle, however, SimpleBox is a general diffusion model that is not linked specifically to toxic effects or risks. In principle, then, for LCA a diffusion model based on SimpleBox can therefore be employed for all emission-related problem categories. Because the current USES model (RIVM, VROM, WVC, 1994) can be used for organic substances only, however, such a broad field of application is not yet feasible in practice. In the following, then, the focus is on the modelling of human-toxicological and ecotoxicological impacts within LCA.

Structure of the SimpleBox model

The basic function of the SimpleBox model is to model substance flows between and degradation within various environmental media. The following environmental media are distinguished:

- air;
- water;
- agricultural soil;
- industrial soil;
- natural soil;
- suspended matter;
- sediment;
- aquatic biota;
- groundwater.

Direct emissions can occur in the first five media only. In addition, emissions can find their way indirectly to any of these media, as a result of substance flows between the various media.

SimpleBox distinguishes two types of substance flows: diffusion flows and advection flows.

Diffusion flows are flows connected with the natural tendency for substances to achieve physico-chemical equilibrium in their distribution over media. Because diffusion is an equilibrium process, it is also a two-way process. Phrased differently, every diffusion flow is accompanied by a diffusion flow in the opposite direction. In a situation of physico-chemical equilibrium these two flows have the same magnitude. In a steady-state situation this need not necessarily be the case at all.

Advection flows are flows occurring under the influence of external physical, geological and climatological forces. Examples include sedimentation, rain, wind and the flow of rivers. Advection flows by no means always have a counterpart in the opposite direction, because the forces in question usually act in one direction only.

To calculate the absolute magnitude of the various advection and diffusion flows resulting from a given emission flux is an extremely complex mathematical exercise, because it is dependent on the concentrations of the substance in question in the media between which these flows occur. In turn, these concentrations depend, directly or indirectly, on all other advection and diffusion flows between all the media. The ratio between the advection or diffusion flow and the concentration in the medium the flow is leaving can be expressed in a fairly simple formula, however. For this reason the SimpleBox model does not compute advection and diffusion *flows*, but advection and diffusion *coefficients*. These can be considered as the 'specific advection' and 'specific diffusion' of the substance in question in a given medium.

Together, the various media make up an environmental system. The largest conceivable environmental system is the environment of the earth as a whole, but smaller systems, in the form of countries or continents, can also be defined in this way. In that case, the environmental system is connected to other environmental systems, and substance flows may occur not only within and between the media of an environmental system, but also between the various different environmental systems. Two environmental systems are described concretely in the USES model: the environmental system of the Netherlands and the environmental system of Western Europe.

Besides environmental systems there are also other systems to which an environmental system may be connected. In the first place, there may be connections to economic systems, including factories and water treatment plants, for example. Generally speaking, flows from these economic systems to the environmental systems are termed emissions. In the second place, in many models certain parts of the earth are not included in the modelled environmental system. In the SimpleBox model old, deep sediments and the seas and oceans are not modelled as part of an environmental system. Deeper-lying groundwater is modelled as part of the environmental system in certain respects but not in others.

Substances may find their way into an environmental system as a result of either transport from a different environmental system or emission from the economic system. Substances may disappear from an environmental system through transport to a different environmental system, through transport to deep sediments or deeper groundwater or through degradation (Figure 3.1).

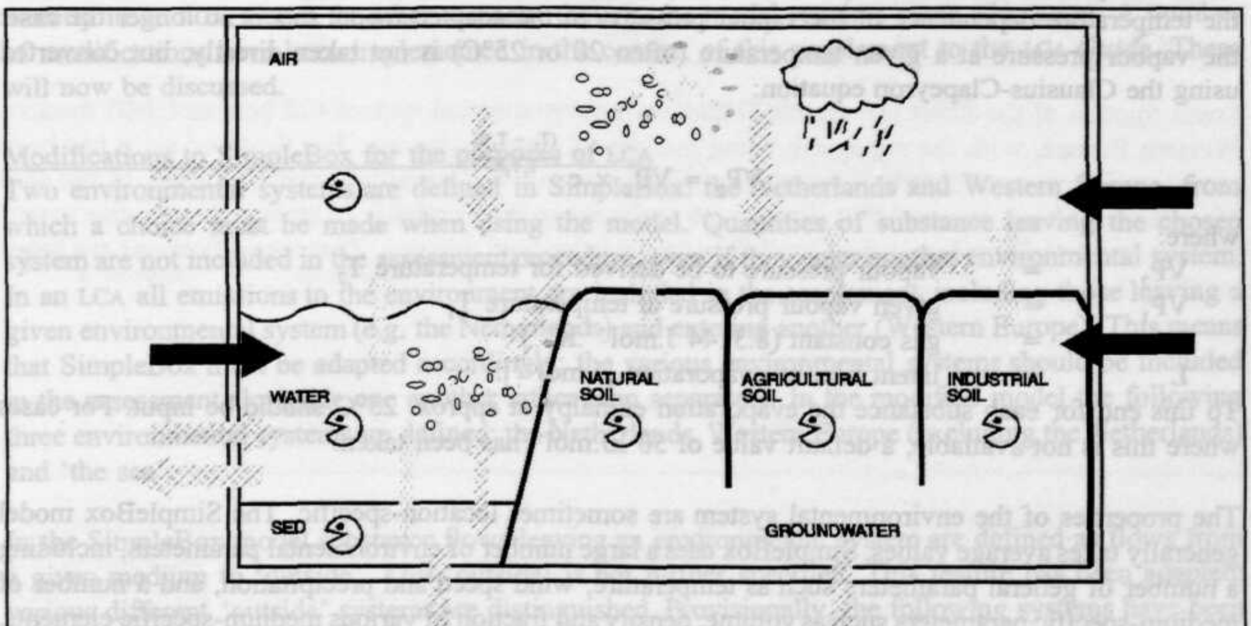

Figure 3.1 Substance flows modelled in SimpleBox (Source: RIVM, VROM, WVC, 1994)

In the steady-state situation, the quantity of a substance leaving the environmental system is the same as that flowing into the environmental system. The same holds for each individual environmental medium. For a given emission flux to a certain environmental medium the SimpleBox model can calculate the anticipated ultimate concentration in each environmental medium due to the emission flux. In risk analysis this concentration is frequently termed the PEC (predicted environmental concentration).

Input parameters for SimpleBox

How a substance is distributed over the various environmental media is dependent on the physico-chemical properties of the substance, on the one hand, and on a number of properties of the environmental system, on the other. The following physico-chemical properties of a substance must be available for use in the SimpleBox model:

- molecular weight;
- octanol-water partition coefficient and water solubility; if one of these parameters is lacking, an estimate can be made if the melting point is known;
- vapour pressure.

The SimpleBox model uses a default value for vapour pressure. This value has been arbitrarily chosen and it is therefore better not to employ it for the purposes of calculation (Jager, 1995).

It should be borne in mind that the vapour pressure of a substance is in fact temperature-dependent. In the SimpleBox model a temperature of 12°C is generally assumed, but no allowance is made for the temperature dependence of the vapour pressure. In the adapted model this is no longer the case: the vapour pressure at a given temperature (often 20 or 25°C) is not taken directly, but converted using the Clausius-Clapeyron equation:

$$VP_2 = VP_1 \times e^{\frac{(T_2 - T_1)L}{T_1 T_2 R}}$$

where:

VP_2	=	vapour pressure to be derived for temperature T_2
VP_1	=	given vapour pressure at temperature T_1
R	=	gas constant (8.3144 $J.mol^{-1}.K^{-1}$)
L	=	latent heat of evaporation [$J.mol^{-1}$].

To this end for each substance the evaporation enthalpy at approx. 25°C should be input. For cases where this is not available, a default value of 50 $kJ.mol^{-1}$ has been taken.

The properties of the environmental system are sometimes location-specific. The SimpleBox model generally takes average values. SimpleBox uses a large number of environmental parameters, including a number of general parameters such as temperature, wind speed and precipitation, and a number of medium-specific parameters such as volume, density and fraction of various medium-specific elements. In the two environmental systems described in SimpleBox average values have been defined for these parameters.

Besides information on flows between the various media information is required on degradation rates (half-lives) within the individual media. The half-lives are dependent on both substance properties and properties of the environmental system and consequently differ from substance to substance as well as from location to location. For an individual substance (and for each medium) the literature does not therefore usually give a fixed half-life, but rather a range within which the half-life lies, depending on local conditions. These are often wide ranges, with the upper and lower bounds sometimes differing by a factor of up to 200. In contrast, the SimpleBox model requires input of fixed values. However, no indication is given of how to construct such values from the available ranges. A possible, provisional approach is to use the geometric mean. If no half-life values are available, the SimpleBox model uses default values. The following half-lives are required:

- half-life for photochemical degradation in air;
- half-life for biological degradation in surface water or – if unavailable – the result of the 'pass ready' test (test for rapid biodegradability);
- half-life for hydrolysis in surface water;
- half-life for biological degradation in soil.

These model parameters are used to perform modelling calculations that yield the distribution of the concentrations of a substance in the various environmental media in relation to the emission of the substance to a certain medium.

Use of the SimpleBox model for LCA

In the SimpleBox risk-evaluation model, the concentrations of a substance in the various media of an environmental system are calculated as a function of the emission flux of the substance to a given environmental medium. In an LCA, on the other hand, emissions are expressed in terms of absolute quantities. For the purpose of LCA, what must be calculated are the concentrations of the substance integrated over time. In other words, the calculations required for an LCA are different from those performed for the purpose of risk assessment. However, the proportional distribution of an emitted substance over the various media is independent of whether the emission is a flux or a pulse (Heijungs, 1995). The codes used in SimpleBox to calculate these proportions can consequently be adopted directly in an LCA model. In other respects, though, the model requires some adaptation. A number of modifications have been implemented in the context of this supplement to the *LCA Guide*. These will now be discussed.

Modifications to SimpleBox for the purposes of LCA

Two environmental systems are defined in SimpleBox: the Netherlands and Western Europe, from which a choice must be made when using the model. Quantities of substance leaving the chosen system are not included in the assessment procedure, even if they enter another environmental system. In an LCA all emissions to the environment are included in the assessment, including those leaving a given environmental system (e.g. the Netherlands) and entering another (Western Europe). This means that SimpleBox must be adapted accordingly: the various environmental systems should be included in the assessment alongside one another rather than separately. In the modified model the following three environmental systems are defined: the Netherlands, Western Europe (excluding the Netherlands) and 'the sea'.

In the SimpleBox model substance flows leaving an environmental system are defined as flows from a given medium to 'outside'. This 'outside' is not further specified. This feature has been adapted: various different 'outside' systems are distinguished. Provisionally, the following systems have been distinguished:

1. 'outside' relative to the environmental system of the Netherlands:
    - the West European environmental system;
    - the sea;
    - deep sediment;
    - deeper aquifers.
2. 'outside' relative to the environmental system of Western Europe:
    - the Dutch environmental system;
    - the sea;
    - deep sediment;
    - deeper aquifers.

In the modified model only deep sediment and deeper aquifers are excluded from the environmental system; flows to the other 'outside' systems are considered to be flows *within* the environment in the modified model.

In the SimpleBox model the sea is not defined as (part of) an environmental system: provisionally, direct and indirect emissions to the sea are not incorporated in the risk assessment procedure. For the purposes of LCA the sea has been added to the model as a separate environmental system. The area of this 'sea' system has – extremely arbitrarily – been set as the area of Western Europe. This was feasible because the magnitude of an environmental system does not influence the impact score of an LCA: a larger system merely means a greater degree of dilution. For the depth of the mixing layer a value of 50 m has been taken. Besides seawater the environmental system 'sea' also comprises the media 'air' (the air column above the surface of the sea), 'suspended matter', 'sediment' and 'biota'. Table 3.1 shows the basic parameters of the 'sea' system. The other parameters derived from these have been calculated entirely analogously to the corresponding parameters for the 'Netherlands' and 'Western Europe' environmental systems in the original SimpleBox model.

TABLE 3.1 The basic parameters of the environmental system 'sea'

variable [unit]	symbol	default value
total area of the system 'sea' [m^3]	SYSTEMAREAsea	$4.0 \cdot 10^{12}$
water fraction of the area of sea [-]	$Fareasea_{water}$	1.0
mixing depth of the seawater [m]	$DEPTHsea_{water}$	50

Some fraction of the flows leaving the Netherlands environmental system will be transferred towards Western Europe, with the remainder being transferred towards the sea. For flows of air it has been provisionally assumed that half move towards the sea and half towards Western Europe. In the case of water it has been assumed that all flows go to the sea in their entirety. In technical terms this means that the 'export' formulae of the SimpleBox model have been adapted. These modifications are listed in Table 3.2.

Table 3.2 Modifications to the formulae for 'export'

original model	modified model
$IMPORT_{air} = ADV_{outside-air} \cdot Coutside_{air}$	$IMPORT_{air} = ADV_{aircon-air} \cdot C_{aircon} + ADV_{airsea-air} \cdot C_{airsea}$
$IMPORT_{aircon} = ADV_{outside-aircon} \cdot Coutside_{aircon}$	$IMPORT_{aircon} = ADV_{air-aircon} \cdot C_{air} + ADV_{airsea-aircon} \cdot C_{airsea}$
—	$IMPORT_{airsea} = ADV_{air-airsea} \cdot C_{air} + ADV_{aircon-airsea} \cdot C_{aircon}$
$ADV_{outside-air} = \frac{V_{air}}{TAU_{air}}$	$ADV_{aircon-air} = \frac{1}{2} \cdot \frac{V_{air}}{TAU_{air}}$ $ADV_{airsea-air} = \frac{1}{2} \cdot \frac{V_{air}}{TAU_{air}}$
$ADV_{air-outside} = ADV_{outside-air}$	$ADV_{air-aircon} = ADV_{aircon-air}$ $ADV_{air-airsea} = ADV_{airsea-air}$
$ADV_{aircon-outside} = ADV_{outside-aircon}$	$ADV_{aircon-airsea} = \frac{1}{2} \cdot \frac{V_{airsea}}{TAU_{air}}$ $ADV_{airsea-aircon} = \frac{1}{2} \cdot \frac{V_{aircon}}{TAU_{air}}$

To calculate nett sedimentation in Western Europe the modified model does not make direct use of the formula for nett sedimentation employed for the Netherlands; instead, the formula is slightly adapted to West European conditions:

$$\begin{aligned} NETsedratecon = (&PROD_{susp} \cdot \frac{AREAcon_{water}}{AREA_{water}} \\ &+ SUSPCONC_{SLS} \cdot EFFLUENTcon_{stp} \\ &+ EROSION_{soil} \cdot (AREAcon_{nat} + AREAcon_{agr} + AREAcon_{ind}) \\ &\cdot Fsolid_{soil} \cdot RHOsolid \\ &- SUSPCONC_{surf} \cdot (EFFLUENTcon_{stp} + WATERrunoffcon)) \\ &\cdot \frac{1}{Fsolid_{sed} \cdot RHOsolid} \cdot \frac{1}{AREAcon_{water}} \end{aligned}$$

For calculating nett sedimentation in sea, too, this formula has been slightly adapted:

$$\text{NETsedratesea} = (\text{PROD}_{\text{susp}} \cdot \frac{\text{AREAsea}_{\text{water}}}{\text{AREA}_{\text{water}}} + \text{SUSPCONC}_{\text{surf}} \cdot (\text{EFFLUENTcon}_{\text{stp}} + \text{WATERrunoffcon})) \cdot \frac{1}{\text{Fsolid}_{\text{sed}} \cdot \text{RHOsolid}} \cdot \frac{1}{\text{AREAsea}_{\text{water}}}$$

In the SimpleBox model flows to deeper aquifers are considered as flows leaving the environmental system. The concentration of a substance in the deeper aquifers is calculated, however: not directly, but as a function of the calculated concentration in the agricultural soil. This approach has been directly adopted in the modified model.

The output of the original SimpleBox model consists of the concentrations in the various media of the environmental system in which (part of) the life cycle of a substance occurs (the Netherlands or Western Europe). The modified model yields the time-integrated quantity of every substance associated with the production of a functional unit of the product that enters the various media of every environmental system (the Netherlands, Western Europe and 'the sea'). In the characterization phase of an LCA these medium-specific quantities can be aggregated to give the total emission of the substance in question to every environmental medium.

Exposure

One of the main differences between LCA and risk analysis concerns the nature of the exposure being assessed. In the case of risk analysis, what is involved is continuous exposure to a constant concentration resulting from a continuous emission. Because in an LCA emission pulses rather than emission fluxes are assessed, we are here concerned with temporary exposure to a concentration that decreases with time.

In LCA product-related emissions are calculated solely on the basis of loads; in other words, there is no data available on the time span over which the various emissions take place, nor on the degree of diffusion. This is of influence on the scope for assessing exposure to emitted substances in LCA, and the restrictions to which this is subject.

Depending on the degree of diffusion and the associated degree of dilution, an emission of a given magnitude may lead to exposure of a relatively small area – and consequently a relatively small population – to a relatively high concentration, or exposure of a relatively large area – and consequently a relatively large population – to a relatively low concentration. If the population is evenly distributed, the concentration level is inversely proportional to the size of the exposed population. For the purposes of LCA assessment, it makes no difference whether a large population is exposed to a low concentration or a small population to a high concentration: the exposure factor is calculated such that it is proportional to both the concentration and the size of the exposed population. In the modified model it is assumed that the Netherlands has a population of 15 million and Western Europe a population of 157 million.

A given emission may take place over a short or a long period of time. An emission occurring within a relatively short time span will lead to relatively short exposure to a relatively high concentration, while an emission drawn out over a relatively long time span will lead to relatively long exposure to a relatively low concentration. The concentration level is inversely proportional to the duration of exposure. Again, this makes no difference for assessment in the context of LCA: the exposure factor is proportional to both the concentration and the duration of exposure to that concentration.

In the current LCA model the exposure of aquatic and terrestrial ecosystems and of humans is calculated. This structure has been retained for the time being. For aquatic and terrestrial ecosystems the assumption has been introduced that exposure to a substance is proportional to the concentration in the media water and soil, respectively. Human exposure is calculated from the intake of pollutants via the consumption of air, drinking water, crops, meat, fish and dairy products. The modified SimpleBox model is employed to compute the (time-integrated) concentration of a given pollutant in the atmosphere.

In the original USES model drinking water consumption is based on a worst-case situation: the anticipated concentration in groundwater and in surface water treated by two methods are compared and the worst case assumed. In the modified model this assumption has not been retained: now the point of departure is that 35% of the drinking water is prepared from groundwater and 65% from surface water. For consumption of drinking water prepared from surface water consumption following treatment is assumed. The influence of this drinking-water treatment is computed separately for each individual substance on the basis of the USES drinking-water treatment codes for storage in open reservoirs and dune filtering. It is assumed that these two systems are used with equal frequency.

In the USES model the concentrations in fish, meat and dairy products are computed from the concentrations in agricultural soil and surface water using substance-specific bioconcentration factors. These calculations have been retained unchanged in the modified model.

In the original USES model no distinction is made between oral and inhalatory exposure. In the modified model these two exposure routes are distinguished separately.

Effect
As a measure of the human toxicity of a substance the USES model takes the NEL (no effect level). This value is calculated from the toxicity data obtained in human and animal studies, which are multiplied by certain safety factors – depending in the quantity and quality of the data. In LCA the NEL is employed as a measure of toxicity: the lower the NEL, the greater the toxicity. How this value is obtained is described in the *LCA Guide*. In contrast to the USES model, the modified model distinguishes between an oral and an inhalatory NEL, as is done in the *LCA Guide*. In calculating the NEL from toxicity data the modified model deviates from the original USES model on one point: if only subchronic toxicity data are available, the chronic data are derived from these by multiplying them by a safety factor of 0.1.

As a measure of the aquatic and terrestrial toxicity of a substance the USES model takes the NEC (no effect concentration). In LCA the NEC is used as a measure of toxicity, analogously to the NEL. In the modified model the NEC values are computed analogously to the procedure of the USES model. If there are insufficient data available for the terrestrial ecosystem, these are estimated using the equilibrium partitioning method, which is also included in the USES model.

Risk evaluation
Concentrations of substances in the environment bring with them a risk of toxic effects if organisms are exposed to levels exceeding the 'no effect level'. For aquatic and terrestrial ecosystems this means that there is a risk if the ratio PEC/NEC exceeds unity.

To assess whether the calculated concentrations involve a risk to human beings, these concentrations must be used to calculate the resulting exposure levels. This calculation is based on the quantities consumed daily by a human being from each medium, for example the volume inhaled and the fish consumption. For each exposure route this daily intake resulting from an emission can be compared with the threshold value, or NEL (no effect level), below which no adverse effects on public health are anticipated.

LCA does not involve any calculations of risk. Nonetheless, in order to assess the relative gravity of various emission pulses of various substances, a mode of calculation is employed that is reminiscent of modes commonly used in the field of risk analysis. For each medium and for each substance, a time-integrated concentration is calculated, resulting from a certain emission pulse of the substance or from a number of summed emission pulses. To calculate the relative threat posed to a given ecosystem by such an emission pulse, the calculated integral is then compared with the NEC for the ecosystem in question. For each substance emitted, the ratio integral/NEC provides a so-called effect score for the 'toxicity' with respect to the aquatic or terrestrial ecosystem. Impact scores for various different substances can be summed to yield an overall impact score for the toxicity to the ecosystem in question.

For human beings, the calculated integrals and the intake from the various environmental media are used to calculate an exposure level. To this end, a single, integrated dose is calculated rather than a daily dose. This dose is multiplied by the number of people exposed and then compared with the NEL. Here, too, impact scores are calculated. For each individual substance, these impact scores relate to the various media to which humans are exposed, but these can ultimately be summed to yield a single, overall impact score for human toxicity.

Although LCA and risk analysis are thus fundamentally different tools, for many calculations the same set of instruments can largely be employed.

The modified USES model can be used for constructing new equivalency factors that incorporate the distribution of the various substances over the various environmental media. Because three environmental systems are distinguished in the modified model, already in the inventory table a distinction must be made between processes taking place in the Netherlands, processes taking place in the rest of Western Europe and processes taking place at sea. No model has yet been developed for processes taking place in the rest of the world. As a provisional solution it is proposed to use the West European model for this purpose. This means that in calculating the distribution of each substance over the various environmental media West European geographical and climatological conditions have been provisionally assumed.

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