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The Eco-indicator 99
A damage oriented method for Life Cycle Impact Assessment

Methodology Report

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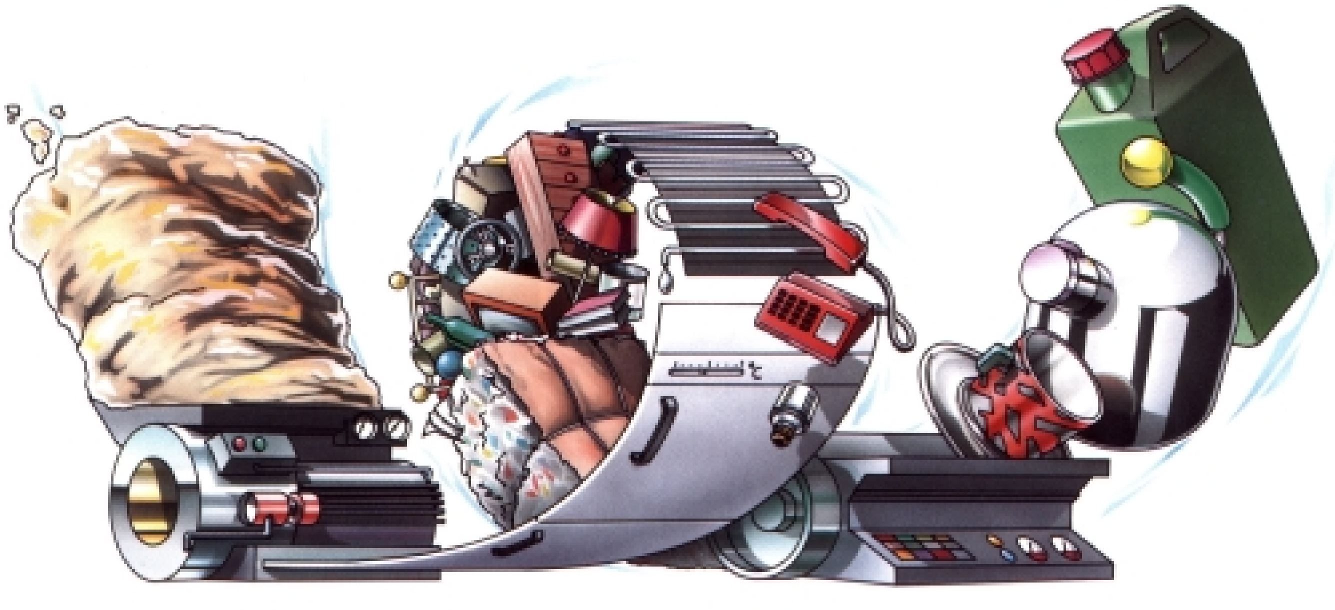

product ecology consultants

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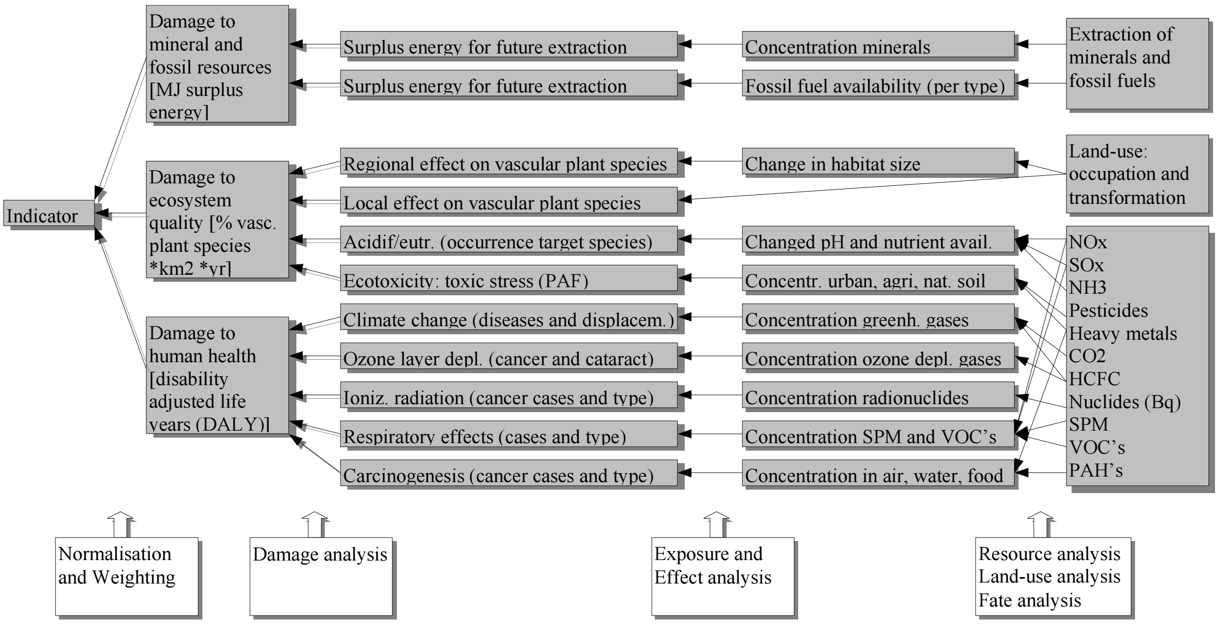
22 June 2001
Third edition



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PREFACE

ACKNOWLEDGEMENTS

SUMMARY

Other reports from this project are:

- The Eco-indicator 99 Annexe Report
- The Eco-indicator 99 Manual for Designers

Reading advise

In order to make a quick assessment of the method, we suggest you concentrate on chapter 1, without the text boxes and chapter 2, as this chapter gives an overview. You can stop here, but for a better understanding we suggest to read also paragraph 3.1, 4.1 and 4.2, 5.1 and 6.1. If you are also interested in weighting and normalisation chapter 7 and 8 should be read.

Some methodological issues are described in textboxes with a layout as this text. These texts can be skipped if desired.

We tried not to represent all the data in this report, as this would sometimes result in sheer endless tables. Such tables can be found in the separate Annexe report.

Preface

Sustainable production and consumption can only be achieved if all market actors take their own responsibility. The ultimate goal is therefore taking into account environment in every decision making process by industry, retailers and consumers. This is a steadily growing process that needs to be fostered by sufficient incentives both from the demand as the supply side. To this end a comprehensive set of policy instruments has been developed in the Netherlands under the label of Integrated Product Policy (IPP).

At the centre of IPP is the introduction of Product Oriented Environmental Management System [POEM] which is being developed in a concerted action both by industry and by government in recent years. The objective of POEM is to establish a systematic drive for continuous improvement of the life cycle environmental performance of products within all sorts of enterprises by integrating environmental aspects in strategic management decisions.

POEM has to be seen as an elaboration of Environmental Management System that focus particularly on product development and product (re)design. The complexity of the decision process involving all environmental aspects means very often an unbridgeable gap for designers. Although life cycle assessment [LCA] is a good tool to assess the environmental performance of a product, and although it is widely used by designers, LCA is time consuming and costly. Designers have to make many decisions especially when designing complex products. Moreover the results of LCA are mostly not straightforward in favour of one product or material design over the alternative one. Results of LCA have to be interpreted or weighed. The Eco-indicator 95 methodology is an LCA weighing method specially developed for product design. This method has proved to be a powerful tool for designers to aggregate LCA results into easily understandable and user-friendly numbers or units, the so-called Eco-indicators.

The Eco-indicator ’95 methodology is being used very often by designers but is criticised by environmental experts at the same time because some environmental aspects were not accounted for in the method. The new Eco-indicator 99 method includes many more aspects and is therefore more complex than the 95 version but the resulting Eco-indicators are still the same user-friendly units.

The weighing system between the different environmental aspects - the core of the Eco-indicator method - has also been changed. The 1995 Eco-indicator used the so-called Distance-to-Target approach. This method was criticised because there was no clear-cut objective way to define sustainable target levels. This problem is in the present Eco-indicator method avoided by introducing a damage function approach. The damage function presents the relation between the impact and the damage to human health or to the ecosystem.

Contributions of many LCA experts have been merged in this 99 method. I would particularly acknowledge the contributions from several Swiss Experts and of the National Institute of Public Health and the Environment [RIVM].

The Eco-indicator 99 does reflect the present state of the art in LCA methodology and application. This of course does not mean that all problems are solved. Further developments in environmental science, material technology and LCA methodology will take place and should result in future improvements of the Eco-indicator. But we are convinced that the revised Eco-indicator methodology is sufficiently robust to play an important role in eco-design for the next years.

I hope the Eco-indicator 99 method and the resulting Eco-indicators shall contribute to the incorporation of environment in product development decisions.

Director Industry- and Consumer Policy

Jan Suurland

Acknowledgements

For the purposes of the project a consultative and collaborative structure was established, containing environmental experts from the RIVM (National Institute of Public Health and the Environment) and LCA experts from different organisations in the Netherlands and Switzerland. Especially the contribution of the Swiss scientists Ruedi Müller-Wenk and Patrick Hofstetter have proven to be extremely valuable. The work of the Swiss team for the development of the Eco-indicator methodology was sponsored by the Swiss National Science Foundation and by the Swiss Agency for the Environment, Forests and Landscape; BUWAL. The work of the Dutch team, including the RIVM, was financed by the Dutch Ministry of Housing, Spatial Planning and the Environment.

Name	Employer	Main contribution to this project
Ruedi Müller-Wenk	IWÖ-HSG St. Gallen	General methodology, Resources, Land use
Patrick Hofstetter	UNS-ETH, Zurich	General methodology, DALY, Cultural Theory, Carcinogenic and Respiratory effects, Climate change; Ozone layer depletion, Ionising radiation
Thomas Köllner	IWÖ-HSG St. Gallen	Land use
Thomas Mettier	UNS-ETH, Zurich	Panel procedure.
Arthur Braunschweig	IWÖ-HSG St. Gallen	General methodology, Ionising radiation
Rolf Frischknecht	ESU-services	Ionising radiation
Dik van de Meent	RIVM ECO	PAF
Mathieu Rikken	RIVM CSR	EUSES
Ton Breure	RIVM ECO	PAF
Reinout Heijungs	CML Leiden	General methodology
Erwin Lindeijer	IVAM Amsterdam	General methodology
Hein Sas	CE Delft	General methodology
Mark Goedkoop	PRé Consultants	Project leader, General methodology, Resources, Land use, Acidification/Eutrophication, Panel procedure
Renilde Spriensma	PRé Consultants	General Methodology, Fate analysis, DALY, PAF, Normalisation
Suzanne Effting	PRé Consultants	Ozone layer depletion

Table 1 Overview of the most actively contributing experts involved in this project

The project was steered by the Eco-indicator-Platform, containing the following members:

Name	Representing
Henk Wijnen	Ministry of Housing, Spatial Planning and the Environment (VROM)
Berno Ram	Philips, ECC
Pim Tamis	Océ, product development division
Tini Peters	NedCar. Materials specialist
Chris Dutilh	VNO-NCW and Unilever
Helias Udo de Haes	CML; Centre for Environmental Studies, University of Leiden
Sietze Kalisvaart	TNO-industry
Han Remmerswaal	Delft University of Technology, industrial design department
Erwin Lindeijer	IVAM-er
Bas van Esch	RIVM
Ton Breure	RIVM-ECO

Table 2 Overview of the active members of the Eco-indicator platform.

The persons mentioned here do not necessarily endorse the entire content of this report.

Summary

The Dutch Ministry of Housing, Spatial Planning and the Environment (VROM) has commissioned this project as part of the Integrated Product Policy. The aim is to develop an update of the Eco-indicator 95 scores. These indicator scores are widely in use by many designers in many companies. In order to calculate such single scores a methodology is needed. This report describes the new methodology. A separate "Manual for Designers" describes the application of the indicators. From this it follows that the method presented here is intended to be used for product development applications.

The most critical and controversial step in Life Cycle Impact Assessment (LCIA) is the weighting step. With this in mind the Eco-indicator methodology has been developed top down. The weighting step was our starting point. From there we have developed damage models for the most important impact categories.

There are two important requirements for the weighting step, if a panel is used:

1. The number of subjects to be weighted should be as small as possible. In most LCIA methods a panel is asked to weigh ten or more subjects; this number is clearly too high.
2. The subjects to be weighted should be easy to explain to a panel. In most LCIA methods the panel is asked to weigh rather abstract impact categories. It is very difficult to give a meaningful assessment.

From these requirements it was concluded that the panel should only weigh the following three types of environmental damages (endpoints):

- Human Health
- Ecosystem Quality
- Resources

The following damage models have been established to link these damage categories with the inventory result.

- Damages to Human Health are expressed as DALY (Disability Adjusted Life Years). Models have been developed for respiratory and carcinogenic effects, the effects of climate change, ozone layer depletion and ionising radiation. In these models for Human Health four sub steps are used:
  - Fate analysis, linking an emission (expressed as mass) to a temporary change in concentration.
  - Exposure analysis, linking this temporary concentration to a dose.
  - Effect analysis, linking the dose to a number of health effects, like the number and types of cancers.
  - Damage analysis, links health effects to DALYs, using estimates of the number of Years Lived Disabled (YLD) and Years of Life Lost (YLL).
- Damages to Ecosystem Quality are expressed as the percentage of species that have disappeared in a certain area due to the environmental load. This definition is not as homogeneous as the definition of Human Health:
  - Ecotoxicity is expressed as the percentage of all species present in the environment living under toxic stress (PAF). As this is not an observable damage, a rather crude conversion factor is used to translate toxic stress into real observable damage.
  - Acidification and eutrophication are treated as a single impact category. Here the damage to target species (vascular plants) in natural areas is modelled.
  - Land-use and land transformation is based on empirical data of the occurrence of vascular plants as a function of the land-use type and the area size. Both the local damage on the occupied or transformed area as well as the regional damage on ecosystems is taken into account.
- Resource extraction is related to a parameter that indicates the quality of the remaining mineral and fossil resources. In both cases the extraction of these resources will result in higher energy requirements for future extraction.

In figure 1 the different procedures and (intermediate) results are shown. A clear distinction is made between intermediate results (grey boxes) and the procedures (white boxes) to go from one intermediate result to the other.

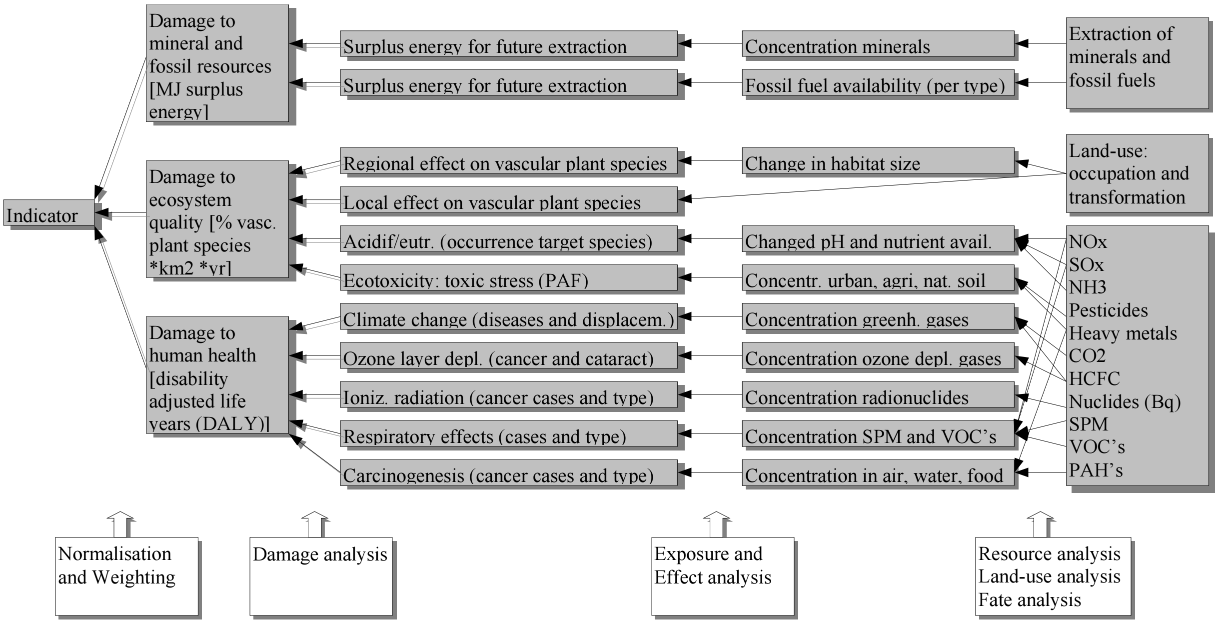

Figure 1: General representation of the methodology. The white boxes below refer to procedures; the other boxes refer to intermediate results.

A limiting assumption is that in principle all emissions and land uses are occurring in Europe and that all subsequent damages occur in Europe. Except for the damages to Resources and the damages created by climate change, ozone layer depletion, air emissions of persistent carcinogenic substances, inorganic air pollutants that have long-range dispersion, and some radioactive substances.

Two types of uncertainties are separated:

1. Data uncertainties; this refers to technical problems of measuring and assessing factors
2. Model uncertainties; this refers to the uncertainty if the model is configured correctly

Data uncertainties are presented as the squared geometric standard deviation. In some cases, especially for toxicity, these uncertainties are substantial.
Modelling uncertainties cannot be expressed as a range; a model assumption is correct or not. In order to cope with these uncertainties a system, referred to as Cultural Theory has been used to separate three versions of the damage model. A simplified characterisation of these versions is:
E (Egalitarian): Long time perspective: even a minimum of scientific proof justifies inclusion
I (Individualist): Short time perspective: only proven effects are included
H (Hierarchist): Balanced time perspective: consensus among scientist determines inclusion of effects.
The H (Hierarchist) version is chosen as default, while the other version can be used in a robustness analysis.

The weighting procedure is executed with a written panel procedure among a Swiss LCA interest group. The results can be used as a default, but should not be considered to be representative for the average European. For those who do not want to use the weighting step, a new alternative approach is suggested using a weighting triangle (see box 7.1 in chapter 7).

Stay updated via the Eco-indicator 99 e-mail user group

In order to receive feedback from you and to be able to send updates we have established a free e-mail user group. The discussion will be monitored and controlled by the development team. This team will encourage factual and concise discussions focussed especially on the Eco-indicator 99 applications and the methodology.
To join the Eco-indicator Internet User Group simply send an empty e-mail to: join-eco-indicator@lists.lyris.net
After confirmation you receive a welcome message with simple instructions and some "house" rules. From that moment, until you unsubscribe you will receive all e-mail send to the user group and of course, you can send e-mail yourself.

1. Introduction

1.1. Background

According to ISO 14040 and 14042, Life Cycle Impact Assessment (LCIA) is essentially meant to improve the understanding of the results of the inventory phase. Until now many methodologies have been suggested and described. These methodologies can be divided into theme oriented methods and damage oriented methods. The first category converts the inventory results into a number of themes [HEIJUNGS ET AL 1992; HAUSCHILD ET AL 1998], the second type models the damages caused by the inventory results. The predecessor of this project, the Eco-indicator 95 methodology was an example of a method based on a damage approach.

This report describes the Eco-indicator 99 impact assessment methodology. A project which is carried out under the authority of the Dutch Ministry of Housing, Spatial Planning and the Environment, under the label of the Integrated Product Policy. The authors collaborated intensively with a number of Swiss and Dutch LCA experts and the Dutch RIVM (National Institute of Public Health and the Environment). The Swiss experts were supported by the Priority Programme Environment from the Swiss National Science Foundation.

The report is structured into 7 chapters:
1 **Introduction**, aiming at positioning the project among other methodologies
2 **General framework**, with a brief overview of the methodology
3 **Fate analysis**, describing the procedures to calculate environmental concentrations
4 **Human Health**, describing the models that are used to calculate the damage to Human Health
5 **Ecosystem Quality**, describing the models used to calculate the damages to Ecosystem Quality
6 **Resources**, describing the models used to calculate the damages to Resources
7 **Damage assessment**, describing the normalisation and weighting procedure
The report is intended for LCA practitioners who want to understand the principles of the methodology. The report is not very specific on data. The data used and the treatment of the data is described in the annex report.
Some subjects described in this report are added as text boxes. In general these text boxes contain descriptions of methodological choices and problems. One can generally skip the boxes without missing essential aspects.

Box 1.1. Life Cycle Impact Assessment; top down and bottom up

In order to determine the interaction between a product and the environment it is necessary to understand the environmental aspects of products throughout the product life cycle. The method for environmentally oriented life cycle assessment (LCA) of products was developed to provide this understanding.
An LCA starts with a systematic inventory of all emissions and the resource consumption during a product's entire life cycle. The result of this inventory is a list of emissions, consumed resources and non -material impacts like land use. This table is termed the inventory result. Since usually inventory tables are very long and hard to interpret, it is common practice to sort the impacts by the impact category, and calculate a score for impact categories such as greenhouse effect, ozone layer depletion, acidification, etc. How these impact categories are to be weighted is much less clear. For this reason it is frequently the case that the results of an LCA can not be unambiguously interpreted. To solve this problem a more complete impact assessment methodology, followed by a weighting step, is needed. There are several ways to do this.

This way of thinking, starting with the inventory result, and then trying to interpret this is sometimes referred to as the bottom-up approach. The bottom-up approach is also reflected in the ISO standards 14040 and 14042, as they define LCIA as a way to improve the understanding of the inventory results.
In the Eco-indicator 95 and 99 projects[1] a top-down approach has been used. This term was first mentioned in [Braunschweig et al 1996]. The top-down approach starts by defining the required result of the assessment. This involves the definition of the term “environment” (the “Eco” we indicate) and the way different environmental problems are to be weighted.
The weighting of environmental problems is usually seen as the most controversial and difficult step in an assessment, as it is usually very difficult to give meaningful values to environmental problems. With this in mind, the top-down approach is designed around the weighting procedure. The rest of the procedure is set up to accommodate the best weighting procedure.

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[1] Other methods that are based on the top-down approach are typically methods based on external costs, like the [EXTERNE 1997] and the EPS [STEEN 1992]

This starting point has some important consequences for the methodology. For instance the number of environmental problems that are to be weighted is limited to just three. Furthermore the environmental problems are defined at their endpoint level, in terms of damages to Human Health, Ecosystem Quality and Resources. Definitions at this level are much easier to comprehend than the rather abstract definitions of greenhouse effect and acidification.
As a consequence the definition of the impact categories is no longer a free choice, as is the case in the bottom-up approach. This means additional requirements have to be fulfilled by the modelling of effects. In the Eco-indicator 99 project this has complicated the development of appropriate models.

In spite of this disadvantage, we firmly believe that the top-down approach is the only useful way of thinking. In the bottom up approach the modelling of impact categories is certainly simpler, but the weighting problem is virtually impossible to solve.

An important advantage of the top-down approach is the ability to separate the really important issues from the not so important issues. In the bottom up approach, one can develop methodologies for problems that are quite insignificant in most LCAs, while one can overlook very important problems. In the top down approach it becomes very clear which effects are really an issue of concern. As will be shown in the section on normalisation, some environmental problems turn out to be rather insignificant, while others dominate in the "European total" LCA. Eventually, the top down approach will steer the development of LCIA.

1.2. Aim of this project

The primary aim set by the Dutch Ministry is to calculate a large number of standard Eco-indicator values for commonly used materials and processes. These standard indicators can be used by designers to perform their own LCA analysis in a matter of minutes (see box 1.3). This aim is similar to the aim of the predecessor, the Eco-indicator 95 project (see box 1.2), which is applied by probably thousands of designers.

An important requirement is the improvement of the reliability and the validity of the calculated indicators. To improve this, the Eco-indicator 95 needed a thorough revision.
The main focus points have been:

- Better preparation of a panel to perform the valuation
- Better definition of the damage categories themselves.
- Inclusion of land-use as an impact or impact category.
- Inclusion of resource depletion.
- Better modelling of the damage functions.
- Inclusion of Cultural Theory, as a tool to manage subjectivity.
- Better and more comprehensive definitions of the effects, and inclusion of fate analysis.

The order in which the intended improvements are listed above is not coincidental. One of the overall, and perhaps most fundamental improvements is the top down approach we followed (see also box 1.1.).

Box 1.2. The predecessor: the Eco-indicator 95

The Eco-indicator 99 methodology will be based upon the predecessor, the Eco-indicator 95. [GOEDKOOP 1995]. This method has been summarised with the following figure.

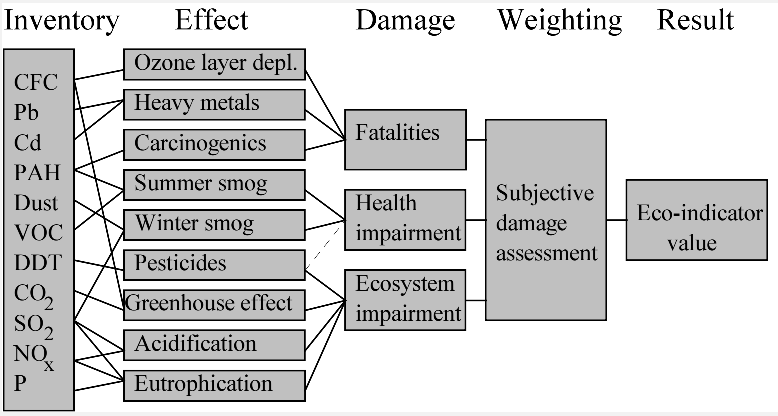

Figure 1.1: Graphical presentation of the Eco-indicator 95 methodology

In the Eco-indicator 95 method the seriousness of an inventory result was represented by the damage it causes to Human Health and to Ecosystem Quality on a European scale.

In the 99 project this principle is reviewed and improved. The 95 method had a number of clear limitations:
- The definition of environment was restricted to the effect of emissions to air and water on Human Health and Ecosystem Quality.
- The damage categories Human Health and Ecosystem Quality were not very well defined.
- The valuation between Human Health and Ecosystem Quality was not very explicitly done.
- Some of the newly developed characterisation sets were limited in scope.
- Fate analysis was not included.

The Eco-indicator 95 was set out to be based on the Distance to Target approach, as it is used in the Ecopoints method [AHBE ET AL 1992]. However, during the development of the project we found distance to target is not adequately expressing all aspects of the seriousness of an environmental problem and that an additional subjective weighting factor was needed. Mainly through the contributions from Heijungs and from studying the EPS system [STEEN 1992] we developed the damage function approach. The result of this development was that the Eco-indicator 95 methodology is based on a combination of distance to target and damage function approach.

The damage functions are used to establish the relation between an impact and the damage to human or Ecosystem Quality. The principle is simple: the slope or first derivative of the damage function expresses the incremental damage at an incremental change of the current impact level. The first derivative is thus a weighting factor that expresses the relation between an impact and the amount of damage to either Human Health or Ecosystem Quality. In the Eco-indicator 95 project all damage curves were assumed to be linear and to go through the origin. This is of course a major simplification of reality.

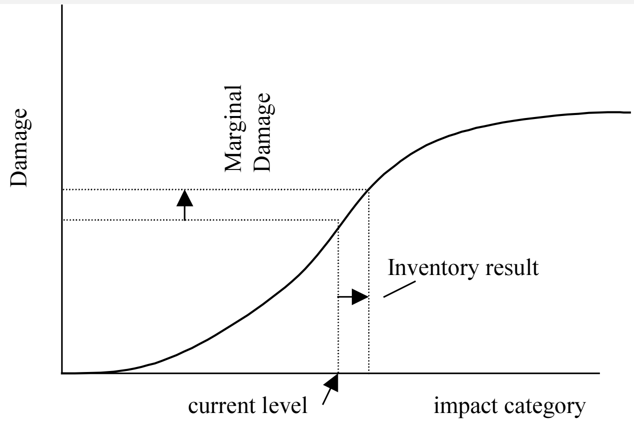

Figure 1.2: Simple correlation between damage and the impact category level such as greenhouse effect or acidification. If the current level changes, the slope of the damage function determines the resulting damage.

In the conclusions of the Eco-indicator 95 report [GOEDKOOP1995] we already stated that in a further development we should concentrate more on the damage function approach. This view was shared by a group of four LCA experts in Switzerland: Prof. Dr. R. Müller-Wenk, Dr. A. Braunschweig, Dr. P. Hofstetter and R. Förster. In their report; "Developments in LCA Valuation" [BRAUNSCHWEIG ET AL 1996], this view is further developed. The Eco-indicator 99 project is inspired by these reports.

Box 1.3 The application of Eco-indicators in a design process

Designers are not environmental specialists and they never will be. Still designers make many decisions that influence the properties of products and thus the environmental load that will occur during the product life cycle. We cannot expect that they will consult an environmental expert in every case, and therefore they need a simple, yet reliable tool to measure the environmental consequences of their design decisions.

So far two major obstacles hamper the practical use of LCA in design projects:
- The results of an LCA are often hard to interpret, as they usually contain data on numerous environmental flows.
- The execution of an LCA takes too much time to be useful to designers.

The Eco-indicator methodology solves these problems in two ways:
- An impact assessment methodology that transforms the data of the inventory table into damage scores which can be aggregated, depending on the needs and the choice of the user, to damage scores per each of 3 comprehensive damage categories, or even to one single score.
- The methodology is used to calculate standard indicator values for a large number of frequently used materials and processes. These figures are available to designers off the shelf.

In the Eco-indicator 95 project we have calculated and published over 100 such standard indicator values. Companies such as Philips have calculated several hundred of additional scores for specific materials, such as electronics and batteries. With these scores designers are able to analyse their design in just a few minutes.

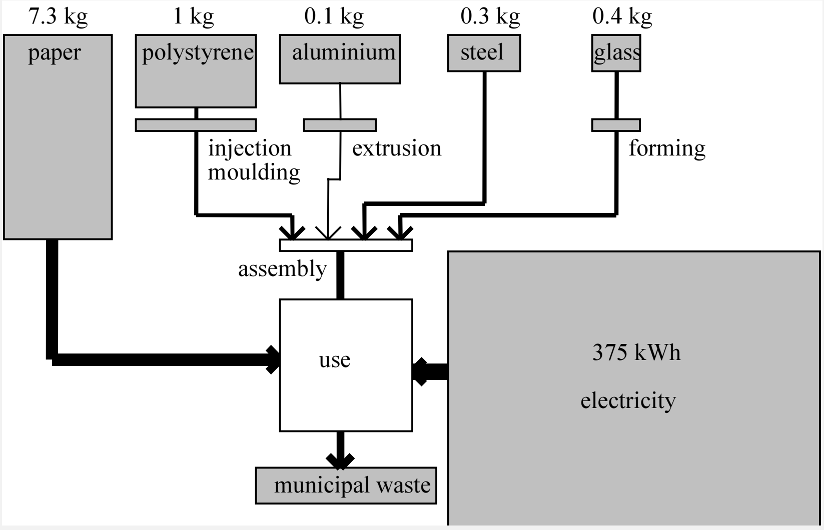

Figure 1.3: A simple analysis of a coffee machine with Eco-indicators shows the designer what the priorities should be; minimise the use of electricity and paper filters. This analysis is made by simple multiplication of the material and energy amounts with the available indicator values.

Of course Eco-indicators do not solve all problems of designers. In fact Eco-indicators can only be used in the creative and conceptual phases of the design process. In the problem analysis phase full LCAs of reference products can be applied.

Phase	**Goal**	**Environmental tools**
The search for products	Development and selection of new product-market combinations	Product policy analysis, assessment of technological and market developments
The analysis phase	Description of the exact goals, and the primary requirements of the product	LCAs of reference products, Short what if analysis
The idea generation phase	Generation of alternative product (service) solutions	Rules of thumb, earlier experiences, guidelines and ***Eco-indicators***
The concept development phase	Selection of the best alternatives and development of concepts	Rules of thumb, earlier experiences, guidelines and ***Eco-indicators***
The detailed design phase	Detailed technical design drawings, calculation of costs etc.	Very specific information, on materials, coatings etc.; Design for recycling tools

Box 1.4. Potential, actual and marginal effects

Traditionally, LCA is based on the "less is better" approach. This means impacts are assessed on their potential effect and not on their possible actual effect. Calculating actual effects is possible when more information is available, for instance, on background concentration or exposure. Several studies have shown that the site dependent factors are very important if we want to base our assessment on real damages [POTTING ET AL 1998].
The problem is that temporal and spatial information is lacking in almost all LCAs. This means that if we want to assess the real damage caused by a product, we should assess every emission (i.e. from every power plant of a power grid), taking into account the spatial and temporal characteristics. This would increase the data amount beyond practicable limits.

In the Eco-indicator 99 spatial differentiation is very limited, while temporal definition is absent. We distinguish damages that are assumed to occur in Europe, and damages that are assumed to occur on a global scale. For the first category the characteristics of the average European Environment and population are used. For the second the global characteristics are used.
The damage that is calculated is not the actual damage, but the marginal damage.

Although the methodology is developed especially for Europe, there are no fundamental reasons that would restrict the applicability to other regions. Preliminary studies have been made on adapting the method on Japan and Colombia [GOMEZ, 1998]. If models for a dozen regions in the world would be developed, it would be possible to assess each industrial process in its own region, taking into account the specific sensitivities of the environment on the continental scale. Later, the method could be refined and adapted to smaller scales. Ultimately, it must be possible to combine the LCA approach with very site-specific conditions.

1.3. Matching goal and scope with the inventory

An important aspect in any LCA is the goal and scope definition. Part of the goal and scope definition is dealing with the impact assessment. The most important requirement is that the impact assessment method suits the goal of the study.

1. The methodology can be used as an impact assessment tool in any LCA study. This tool enables the user to determine scores for environmental damages and to aggregate them up to a single score if desired. When deciding on the degree of aggregation of damage scores to be included in his study, the user will have to take into account to what extent he wants to comply with the recommendations of ISO 14042 or other recommendations. He should also be aware of the underlying assumptions that were used in the methodology for each step of damage aggregation.
2. The pre-calculated set of standard indicator values, consisting of a single score damage indicator per unit of material or process, can be used as a quick tool for product development or ecological benchmarking of production processes. This is essentially for internal use in companies.

In all applications the following characteristics and constraints should be kept in mind when the Eco-indicator 99 method is applied in an LCA:

- All emissions and all forms of land-use are assumed to occur within Europe. The damages for most impact categories are also assumed to occur in Europe, with the following exceptions:
  - The damages from ozone layer depletion and greenhouse effects are occurring on a global scale, as European emissions are influencing the global problem and not just the European.
  - The damages from some radioactive substances are also occurring on a global scale.
  - The damages to Resources are occurring on a global scale.
  - The damages from some persistent carcinogenic substances are also modelled in regions adjoining Europe.
- The method models emissions as if they are emitted at the present time.
- The method is based on a specific definition of the environment (see section 2.2.). If other definitions of the environment are intended, for instance definitions that include human welfare or the preservation of cultural heritages, the methodology is not complete or valid.
- There are special rules for modelling the effect of land use, pesticides and fertilisers (see below).
- The results of the damage models must be seen as marginal results, that is they reflect the increase of the damage when one functional unit is added to the current damage level. Also the normalisation levels are based on the marginal model (see box 7.1 in chapter 7).
- If compatibility with ISO 14042 is required, the last step, the weighting, is not allowed when the results are to be used as comparative assertions disclosed to the public.

As will be explained in the next chapter, the methodology is developed in three versions, all based on a different set of modelling assumptions. Ideally one of these perspectives is also applied in the modelling choices in the inventory stage.

The Eco-indicator 99 methodology has some special requirements for the definition of the inventory table. The most important are summarised here:

- New emissions are taken into account. For most substances we take the emission to air, water and soil into account. In the Eco-indicator 95, the emphasis was on airborne emissions.
  - Fine dust will often be an important parameter. It is important to specify SPM_{10} and $SPM_{2.5}$ carefully.
  - The effects of the fertiliser which is deliberately applied on agricultural land and remains on this land must not be listed separately as emission to soil, since the effect of the fertiliser on the agricultural land is already taken into account in the land-use data. However, the fertilisers that leach out to the water compartment or are emitted to the air must be listed as emissions to water or air.
  - The amount of pesticides, fungicides and herbicides that are directly applied on the soil, should be listed as emission to agricultural soil. In the fate calculations we have modelled, to what extent these substances migrate to compartments outside the agricultural area (air, water, natural soil, and industrial soil). The effects in these areas are taken into account. The effect of

  the pesticides that will remain inside the agricultural area are not modelled, as these effects are already included in the type of land use. The pesticides that are expected to evaporate during the spraying process and the amounts that are accidentally emitted into adjoining service water should be listed as emissions to air and water.
  - Radioactive substances should be listed per isotope, with the unit becquerel. A sum parameter of all radioactive substances has very limited meaning.
- A number of minerals and fossil fuels are taken into account. These should be listed as the minerals or the fossil fuels and not as the ores.
- Land-use is taken into account.
  - It is not always necessary to add land-use data, as a default list is supplied with land-use per raw material or transport mode. Unfortunately the land-use data supplied in the ESU-energy database [ESU 1996] is not directly usable, as this data assumes all land-use incidents are transformations, with long restoration times. However, some hints are given how this data can be used.
  - The land-use should preferably be specified as the product of area and occupation time and an indication of the type of land use. Until now a limited list of 10 different land-use types is available.
  - In the case the land-use includes transformation of land the original and target quality must be specified.

1.4. How to use a single score

The practical goal in this method is the calculation of single scores. Such scores should always be used for internal purposes and are not suitable to use in public comparisons, marketing and ecolabelling, as they lack the necessary transparency[2].

We would like to emphasise the need to document clearly the limitations, uncertainties and assumptions when the single scores are presented. This is especially important if the single scores are used in software products for non-experts. We have proposed a code of practice to some of the software suppliers. This code is available on request from the authors.

In a separate report "the Manual for Designers" [see also GOEDKOOP 1995] we have given an example on how to present the features and the limitations of the method to designers.

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[2] The situation may be different if the single scores are presented with a full documentation of the underlying data and intermediate results.

2. General framework

In this chapter the general framework of the methodology is summarised. In the subsequent chapters the different "building blocks" of the method will be described in more detail. The consequence of this set up is that some parts of the general framework are repeated in the subsequent chapters.

2.1. Technosphere, Ecosphere, Valuesphere

Suppose we can calculate that during the lifecycle of product A, a number of toxic emissions are released that will have significant impact on Human Health, while in the life cycle of product B significant amounts of SO_2 are emitted that cause significant acidification which will damage ecosystems. A nature lover may prefer product A, while another person will perhaps prefer product B.

This simple example shows that we are dealing with three problems:

- First we must analyse the life cycles of the products and determine how much SO_2 and toxic substances are emitted.
- Secondly we must analyse to what extent the SO_2 emission causes acidification and secondly, to what extent this acidification causes damage to ecosystems. Alternatively, we must analyse to what extent the emission of toxic substances indeed causes damage to Human Health.
- Thirdly we must assess how serious damage to ecosystems is compared to damage to Human Health. This is clearly a valuation.

In Life Cycle Impact Assessment we have to deal with three fields of scientific knowledge and reasoning. We refer to these fields as “spheres” [HOFSTETTER 1999]:

- Technosphere, the description of the life cycle, the emissions from processes, the allocation procedures as far as they are based on causal relations.
- Ecosphere, the modelling of changes (damages) that are inflicted on the “environment”.
- Valuesphere: the modelling of the perceived seriousness of such changes (damages), as well as the management of modelling choices that are made in Techno- and Ecosphere.

The first two spheres can be considered to be in the technical and natural science paradigms, the third sphere is clearly in the social science world, as natural science cannot deal with a term like “seriousness”.
Although the spheres are partially overlapping, they have very different characters. For instance in Technosphere we are used to relatively low uncertainties (in the order of a few percents), and we assume measurements can be verified and reproduced. In the Ecosphere we are used to rather uncertain models, which are hard to verify and data that can have uncertainties of several orders of magnitude. An important problem is that the models are partially based on value choices, such as the selection of the time perspective. In the Valuesphere there is a clear understanding that a single truth does not exist.

The understanding that we are dealing with three spheres is crucial when a methodology like the Eco-indicator 99 is being developed[3]. Often the emphasis in the LCA community is in the Technosphere modelling. Projects like the EPS method and the Eco-indicator 95 method have raised the awareness that modelling of the Ecosphere is necessary. Only a few authors in the LCA literature [a.o. HOFSTETTER 1998, TUKKER 1998] have shown that it is at least equally important to carefully model the Valuesphere. In this project we follow Hofstetter’s approach, which is based on the so-called Cultural Theory. This concept will be explained at the end of this chapter.

With these spheres in mind we can construct the basic three-stage approach of the Eco-indicator method:

- The life cycle model is constructed in Technosphere. The result is the inventory table.

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[3] This understanding is not common in the current efforts to standardise LCIA in the ISO 14042 standard. In the papers produced by the technical committee, a distinction is being made between scientific and non scientific parts in the methodology. The latter terms is used to refer to subjective parts in the methodology. It should however not be hard to see that dealing with values is the social science area and therefore not “unscientific”.

- Ecosphere modelling is used to link the inventory table to three damage categories or “endpoints”.
- Valuesphere modelling is used to weight the three endpoints to a single indicator, and to model the value choices in the Ecosphere.

This three-stage method is represented in the graph below.

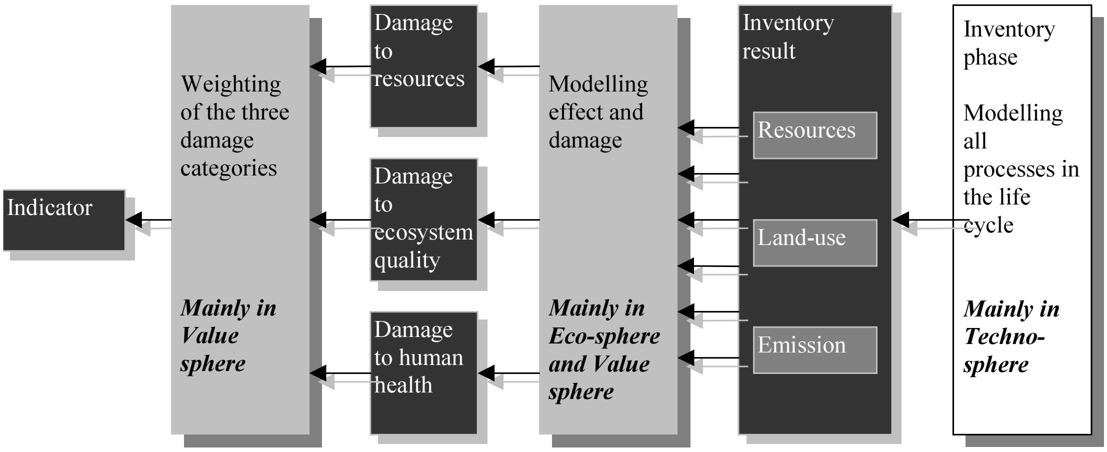

Figure 2.1: The core concept of the Eco-indicator 99 methodology. Apart from the inventory phase, there are two new parts. The development process was performed in a top down approach: starting from the definition of the indicator and the weighting process (see also box 1.1)

Although it is simple to characterise the three phases as if they belong to a single sphere, in reality the distinction is not so clear. Both in Technosphere and Ecosphere we are confronted with normative modelling assumptions and simplifications. This means that Valuesphere is also present in these phases.

As the modelling of the life cycle is not part of this project, the Eco-indicator methodology thus consists of two parts:

- In Ecosphere we use mainly natural science to calculate changes in the environment caused by the environmental flows from a product life cycle.
- A valuation procedure is used to establish the seriousness of these changes. To make our description less abstract we refer to these changes as damages, although strictly speaking changes is a better term, as a change is not a damage before it has been valued. In the ISO 14042 standard the term Endpoints is used. We do not use this term in this context.

This project has not paid any attention to the development of the Life Cycle Inventory phase. However, we realise that every time the Eco-indicator methodology is used, an inventory has to be carried out. The quality of the resulting Eco-indicator values can be no higher than the quality of the inventory phase. In particular it is very important that the goal and scope definition used in the inventory phase matches the application of the calculated indicators.

2.2. The “Eco” we indicate

In daily life the confusion over what people consider to be an environmental problem is big. The term environment is by nature so general that almost everything can be included. Before we can make any sense we have to define what we mean by environment, or what the "Eco" is we want to indicate.

We have found a wide range of different concepts describing the role of the environment in society. On the most basic level the environment is regarded as a basis for our existence and damage to the environment is considered a direct threat to mankind. This kind of thinking is often referred to as the sustainability concept. The basic idea is that, if we damage our environment too much it will no longer be able to support mankind. Basically the sustainability is aimed at avoiding disastrous developments that would endanger humanity.

On a much different level a healthy environment is seen as one of the elements influencing the broad concept of well being, welfare or even human happiness [SLOOFF ET AL 1996]. In this concept threats to the environment are not to be seen as problems that can totally disrupt society. On this level, also subtle changes in the environment are taken into account. A small increase in a certain disease or a small decrease in biodiversity in a certain area, are seen as damages, although such threats are not really a threat to mankind as a whole.

We propose to take the latter approach. That is, we base ourselves on very small changes in the environment, and not so much on the possibility of disasters.

Apart from environmental considerations, landscapes in which the ecosystems are functioning have also other values, like the cultural heritage embodied in changes in the landscape made by earlier civilisations and populations. In fact earlier damages to ecosystems (old aqueducts, castles, canals, dams etc.) have a positive cultural value. It is not our intention to express these cultural values in the Eco-indicator.

We have chosen the following definition of the term environment:
A set of biological, physical and chemical parameters influenced by man, that are conditions to the functioning of man and nature. These conditions include Human Health, Ecosystem Quality and sufficient supply of Resources.

In the Eco-indicator 99 we only look at environmental problems as they occur in Europe. In the Dobris Assessment [DOBRIS 1996], a description of the most serious environmental problems is given.

From this definition it follows that there are basically three damage categories:
- Human Health
- Ecosystem Quality
- Resources

The three terms are not sufficiently self-explaining; a description of what is included in each of the three terms is necessary for building up the methodology, and will be supplied later in this report.
- "Human Health" contains the idea that all human beings, in present and future, should be free from environmentally transmitted illnesses, disabilities or premature deaths.
- "Ecosystem Quality" contains the idea that non-human species should not suffer from disruptive changes of their populations and geographical distribution,
- "Resources" contains the idea that the nature's supply of non-living goods, which are essential to the human society, should be available also for future generations.

Please note that it is also possible to select other damage categories, such as material welfare, happiness, equality, safety etc… We have chosen not to include these aspects, partially because it is too complex to define or model such damage categories and partially because in general products can have both an intended positive effect as well as a negative (environmental) effect. This would for instance lead to the strange conclusion that pesticides have a strong positive effect on the human welfare, while at the same time Human Health could be threatened.

In order to calculate the damage categories we will use a number of intermediate results. We will define and describe these intermediate results and we will make it possible to explicitly calculate and publish such results. This will enhance the transparency of the method and make it easier to compare the results in this methodology with other impact assessment methods

2.3. From inventory results to damage categories

2.3.1. Procedures

The method uses four different procedures to establish the link between the inventory table and the potential damages:

1) In the model for Human Health four sub-steps are used:
   a) Fate analysis, linking an emission (expressed as mass) to a temporary change in concentration.
   b) Exposure analysis, linking this temporary concentration to a dose.
   c) Effect analysis, linking the dose to a number of health effects, like the number and types of cancers, and respiratory effects.
   d) Damage analysis, links health effects to the number of years lived disabled (YLD) and Years of Life Lost (YLL).
2) For ecosystem health two different approaches are used:
   a) Toxic emissions and emissions that change acidity and nutrients levels go through the procedure of:
      i) Fate analysis, linking emissions to concentrations
      ii) Effect analysis, linking concentrations to toxic stress or increased nutrient or acidity levels.
      iii) Damage analysis. Linking these effects to the increased potentially disappeared fraction for plants.
   b) Land-use and land transformation is modelled on the basis of empirical data on the quality of ecosystems, as a function of the land-use type and the area size.
3) Resource extraction is modelled in two steps:
   a) Resource analysis, which can be regarded as a similar step as the fate analysis, as it links an extraction of a resource to a decrease of the resource concentration.
   b) Damage analysis, linking lower concentration to the increased efforts to extract the resource in the future.

In figure 2.2 the different procedures and (intermediate) results are shown. A clear distinction is made between intermediate results (light boxes) and the procedures (black boxes) to go from one intermediate result to the other.

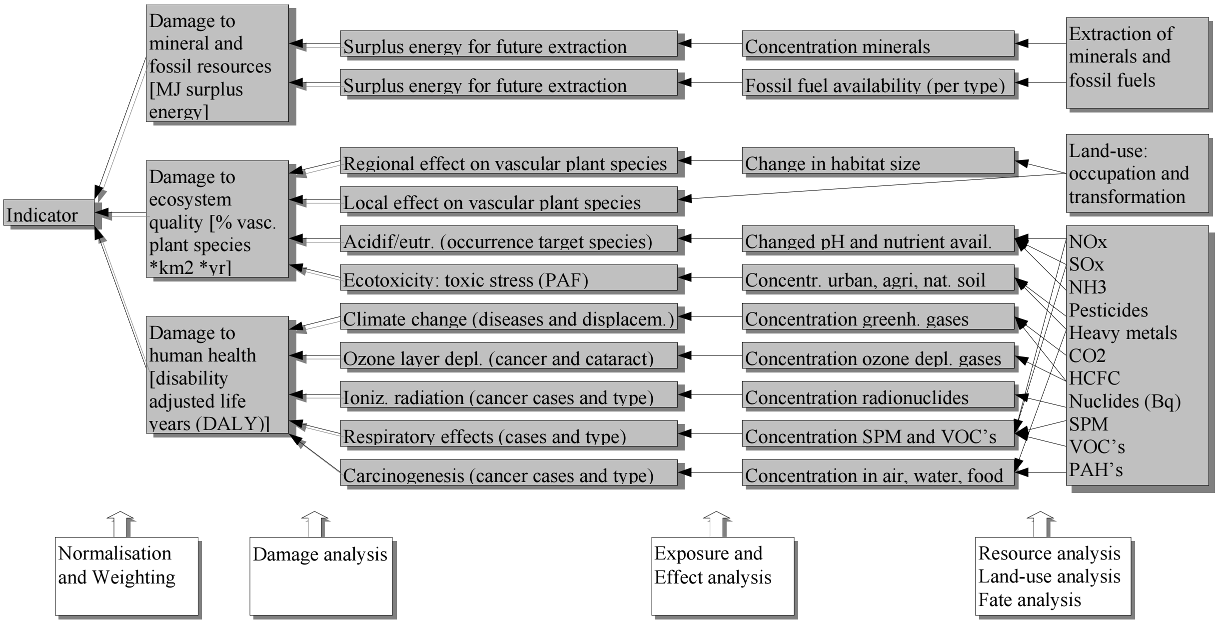

Figure 2.2: General representation of the methodology. The white boxes below refer to procedures, the other boxes refer to (intermediate) results.

2.3.2. The damage category Human Health

The health of any human individual, being a member of the present or a future generation, may be damaged either by reducing its duration of life by a premature death, or by causing a temporary or permanent reduction of body functions (disabilities). According to current knowledge, the environmental sources for such damages are mainly the following:

- Infectious diseases, cardiovascular and respiratory diseases, as well as forced displacement due to the climate change.
- Cancer as a result of ionising radiation.

- Cancer and eye damages due to ozone layer depletion.
- Respiratory diseases and cancer due to toxic chemicals in air, drinking water and food.

These damages represent the most important damages to Human Health caused by emissions from product systems. The damage category is not complete. For instance, damage from emissions of Cd and Pb, endocrine disrupters etc. cannot yet be modelled. Furthermore health damages from allergic reactions, noise and odour cannot yet be modelled.

To aggregate different types of damages to Human Health (which is highly desirable in view of the large number of different types of sickness), a tool for comparative weighting of disabilities is needed. We have chosen to use the DALY (Disability Adjusted Life Years) scale, which has been developed by [MURRAY ET AL 1996] for the WHO and World Bank. The original purpose of the DALY concept was to have a tool to analyse the rationale of national health budgets.

The core of the DALY system is a disability weighting scale. This scale has been developed in a number of panel sessions. The scale lists many different disabilities on a scale between 0 and 1 (0 meaning being perfectly healthy and 1 meaning death).

Box 2.1. Example of a calculation with DALYs

Carcinogenic substances cause a number of deaths each year. In the DALY health scale, death has a disability rating of 1. If a type of cancer is (on average) fatal ten years prior to the normal life expectancy, we would count 10 lost life years for each case. This means that each case has a value of 10 DALYs.
During a summer smog period, many people have to be treated in hospital for a number of days. This type of treatment in a hospital has a rating of 0.392 on the DALY scale. If the hospital treatment lasts 0.01 years on average (3.65 days), each case would be weighted 0.004 DALYs.

With this system, we can calculate the number of Disability Adjusted Life Years if we know how many people in Europe are exposed to a certain background concentration of toxic substances in air, drinking water and food.

[HOFSTETTER 1998], who has studied the use of DALYs in LCA, supplied most data for respiratory and carcinogenic effects due to chemical releases. Hofstetter also performed the calculations for climate change.
Next to this data, we use the proposal of Frischknecht, Braunschweig, Hofstetter and Suter [FRISCHKNECHT ET AL 1999], to include the effect of ionising radiation.

The unit for the damage category Human Health is DALY. This can easily be explained. A flow of toxic substances in tons per year will result in a number of DALY per year. If we leave out the "per year" we find a mass loading is equivalent to a number of DALYs (see also section 3.11)

2.3.3. The damage category Ecosystem Quality

Ecosystems are very complex, and it is very difficult to determine all damages inflicted upon them. An important difference with Human Health is that even if we could, we are not really concerned with the individual organism, plant or animal. The species diversity is used as an indicator for Ecosystem Quality. We express the ecosystem damage as a percentage of species that are threatened or that disappear from a given area during a certain time.

Ecotoxicity

For ecotoxicity, we use a method recently developed by RIVM for the Dutch Environmental Outlook [MEENT AND KLEPPER1997]. This method determines the Potentially Affected Fraction (PAF) of species in relation to the concentration of toxic substances. The PAFs are determined on the basis of toxicity data for terrestrial and aquatic organisms like micro-organisms, plants, worms, algae, amphibians, molluscs, crustaceans and fish . The PAF expresses the percentage of species that is exposed to a concentration above the No Observed Effect Concentration (NOEC). The higher the concentration, the larger the number of species that is affected. The PAF damage function has a typical shape as shown in figure 2.3.

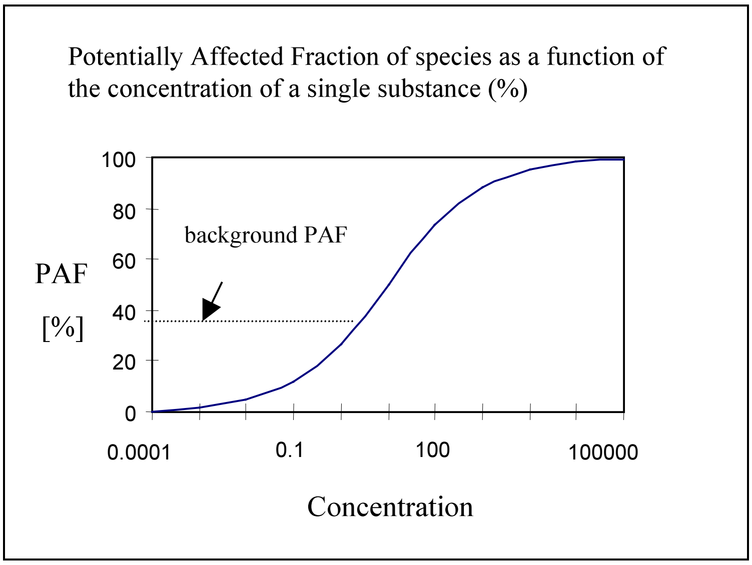

Figure 2.3: A logistic PAF-curve expressing the potential affected fraction of species at different concentrations of a substance

When an emission (mass) is released, the concentration in an area will be increased temporarily. In chapter 3.11, we will argue how we deal with the temporal aspects. This change in concentration will cause a change in the PAF value. The damage caused by the emission of this substance depends on the slope of the curve in a suitably chosen working point.

In [Meent et al 1999] it is postulated that the marginal damage to Ecosystem Quality from a specific emission depends on the present level of damage caused by the present mixture of substances in the environment. This means that we cannot use the background concentrations of single substances. Instead we have to use the combined toxic stress resulting from the present mixture of substances in the environment, the so-called combi-PAF, to find the right working point and slope. In section 5.2 we will further demonstrate how the damage to Ecosystem Quality resulting from toxic stress can be quantified.

Being based on NOEC, a PAF does not necessarily produce observable damage. Therefore, even a high PAF value of 50% or even 90% does not have to result in a really observable effect. PAF should be interpreted as toxic stress and not as a measure to model disappearance or extinction of species.

Acidification and eutrophication

For acidification and eutrophication, we cannot use the PAF concept directly, since damage from acidification and eutrophication is caused by an entirely different and complex biochemical mechanism. Instead, we will have to look at observed effects from acidification and eutrophication on plants. From these observations the probability that a plant species still occurs in an area can be determined. This is called the Probability Of Occurrence or POO [Wiertz et al 1992], which is translated for this project into Potentially Disappeared Fraction (PDF): PDF=1-POO

The computer model “Natuur Planner” developed by RIVM is used for both the fate modelling and the damage modelling for NO_x, SO_x and NH_3 depositions. A particular problem is the fact that acidification and eutrophication do not necessarily reduce the number of species. In fact very often the number of plant species are increased. The solution used by the RIVM is the use of target species. These are the species that should occur on a specific type of ecosystem if there would have been no man-made changes in the nutrient level or the acidity [Bal et al 1995]. The “Natuurplanner” contains a very detailed grid with an exact description of the type of ecosystem and the associated set of target species . The same grid is also used for a site specific fate analysis.
The damage model calculates to what extent the number of target species increases or decreases if an additional deposition is added to the background. Interestingly, it is not possible to determine whether a

damage is caused by changes in the nutrient level or the acidity. For this reason the impact categories have been combined.
Although the "Natuurplanner" is a very sophisticated instrument it is still only available for the Netherlands. The crude assumption was made that the Dutch situation is representative for Europe. Another problem of this impact category is that only damages to natural systems can be modelled and only if these damages occur through airborne depositions. So far we have been unable to include the effect of phosphate and other eutrophying emissions to water.

Land use

For land use, we also use the Potentially Disappeared Fraction (PDF) as indicator. In this case however, we do not consider target species but all species. The damage model is rather complex, as we need four different models:

1. The local effect of land occupation
2. The local effect of land conversion
3. The regional effect of land occupation
4. The regional effect of land conversion

The local effect refers to the change in species numbers occurring on the occupied or converted land itself, while the regional effect refers to the changes on the natural areas outside the occupied or converted area. The regional effect was first described by [MÜLLER-WENK 1998-2]. The data for the species numbers per type of land-use and some of the concepts used for the local effect are based on [KÖLLNER 1999].
The data on the species numbers are based on observations, and not on models. The problem with this type of data is that it is not possible to separate the influence of the type of land-use from the influence of emissions. For this reason some special care must be taken to avoid double counting of effects which are included in land-use and which could be included also in other damage models.

The Ecosystem Quality damage category is the most problematic of the three, as it is not completely homogeneous. A temporary solution is proposed to combine PAF and PDF

The unit for the damages to Ecosystem Quality is the PDF times area times year [m^2.yr]. For land-use this unit is easy to explain: the damage increases with an increase in area size, an increase in occupation time or an increase in restoration time for a formerly converted area..
For ecotoxicity and for acidification/eutrophication some additional explanation is needed (see also box 3.1). 4 logical steps are needed:

1. Let us consider a steady state flow of x kg per year per m^2. This flow will result in a steady state concentration y on a m^2.
2. Now in LCA, we do not know the flow, but only the mass. A mass can be interpreted as a flow during a certain time t.
3. This means, a mass can only be responsible for concentration y on a m^2, during that certain time.
4. As the damage can be linked to the concentration, the flow can only be linked to a certain damage in a certain area, during a certain time.

2.3.4. The damage category Resources

In the Eco-indicator 99 methodology we only model mineral resources and fossil fuels. The use of agricultural and silvicultural biotic resources and the mining of resources such as sand or gravel, are considered to be adequately covered by the effects on land use. Biotic resources which are extracted directly from nature, like fish and game or wild plants, are not modelled in Eco-indicator 99 so far.

In the case of non-renewable resources (minerals and fossil fuels), it is obvious that there is a limit on the human use of these resources, but it is rather arbitrary to give figures on the total quantity per resource existing in the accessible part of the earth crust. If we sum up only the known and easily exploitable deposits, the quantities are quite small in comparison to current yearly extractions. If we include occurrences of very low concentrations or with very difficult access, the resource figures become huge. It is difficult to fix convincing boundaries for including or not-including occurrences between the two extremes, as quantity and quality are directly linked.

Because of this problem, the Eco-indicator 99 methodology does not consider the quantity of resources as such, but rather the qualitative structure of resources. We have chosen to take the concentration of a resource as the main element of resource quality.

Market forces assure that the deposits with the highest concentrations of a given resource are depleted first, leaving future generations to deal with lower concentrations. Thus in theory, the average ore grade available for future generations will be reduced with the extraction of every kilo. This decreasing concentration is the basis for the resource analysis.

The resource analysis is very comparable to the fate analysis, instead of modelling the increase of the concentration of pollutants, we model the decrease of the concentration of mineral resources.

[CHAPMAN and ROBERTS 1983] developed an assessment procedure for the seriousness of resource depletion, based on the energy needed to extract a mineral in relation to the concentration. As more minerals are extracted, the energy requirements for future mining will increase. The damage is the energy needed to extract a kg of a mineral in the future

For fossil fuels we also use the concept of surplus energy. Much of the data is supplied by [MÜLLER-WENK 98-1]

The unit of the Resources damage category is the “surplus energy” in MJ per kg extracted material, this is the expected increase of extraction energy per kg extracted material, when mankind has extracted an amount that is N times the cumulative extracted materials since the beginning of extraction until 1990. A value of 5 is chosen for N. As the surplus energy is dependent on the choice of N, the absolute value of the surplus energy has no real meaning. Surplus energy is used to add the damages from extracting different resources.

2.4. Normalisation and damage assessment

2.4.1. Normalisation

The three damage categories all have different units. In order to use a set of dimensionless weighting factors from the panel we must make these damage categories dimensionless. The obvious way to do this is to use a normalisation step. As the Eco-indicator is developed for Europe, we will use the European normalisation values.

It should be noted that normally in LCA the normalisation takes place after characterisation, as usually the normalised effect scores are presented to the panel. In our model we extend the modelling to the damage categories and we present the damage categories to the panel for weighting. It is thus not more than logical that the normalisation also is moved to this stage. The ISO 14042 standard allows for this change.

2.4.2. Damage assessment

In the previous paragraph we have shown how we can calculate the damage to the three damage categories Human Health, Ecosystem Quality and Resources. In this process we can use the best available scientific knowledge. However, as we indicated in the introduction, we cannot use natural science to determine how serious this damage is perceived.

There are basically two methods to determine values in society:

- Observation of actual behaviour; in this context often referred to as revealed preference method. The core of this method is to analyse how decisions on comparable issues are taken. For instance in the EPS method [STEEN 1992] the value of a human life is based on life insurance, and the value of biodiversity is based on governmental expenditure on this issue.
- Questioning representatives of society (a panel) on the specific issue.

[METTIER 1999] performed a carefully conducted panel procedure among 365 members of a Swiss LCA interest group. The procedure contained a ranking and a weighting procedure. The results cannot be considered to be representative for the views of European population, but they generate a useful first default weighting-set. Next to this default weighting-set the concept of the weighting triangle is described. The weighting triangle can be used to clarify discussions if alternatives for the default weighting-set are to be used. (see box 7.1)

2.5. Sources of uncertainty

In the development of a scientific methodology there are several types of uncertainty to consider, namely fundamental uncertainty and operational uncertainty. The fundamental uncertainty is the reflection of the doubt on the correctness of choices made in the development of the method. The choice of a concept implies that the assumptions that are the basis of this concept are fixed. This uncertainty can not be quantified in an easy way. Operational uncertainty is the variation in the result of the calculations, caused by the variation of the parameters involved. This uncertainty can be quantified. [BAKKER EN VAN DE MEENT 1997].
In the Eco-indicator 99 methodology a third type of uncertainty must be added. This is the uncertainty whether the model includes all important damages that fall under our definition of the term “Eco”. We have found some impact categories that are probably relevant, but for which we have not been able to find an adequate damage model or sufficient data. Also within some impact categories we have found that there are more damage types than we are able to describe. For instance in climate change we can only model a limited set of all the health problems that can probably be related to this impact category.

This means we have three fundamentally different types of uncertainty:

1. Operational, or data uncertainty, which deals with technical uncertainties in the data. Such uncertainties are relatively simple to document by adding the information on the statistical distribution (e.g. standard deviation).
2. Fundamental, or model uncertainties are caused by unavoidable ethical and thus value based choices. Adding a standard deviation or a range on the calculated figures cannot cover this type of uncertainty.
3. Uncertainty on the completeness cannot be documented at all, except for providing a specification of possibly important, but not included damages.

2.5.1. Operational or data uncertainties

In the Eco-indicator 99 it is intended to give quantitative uncertainty estimates for operational uncertainty whenever they are relevant. Some of the sources used include uncertainty analysis in their results. For example in the work of [HOFSTETTER 1998] for each step in the calculation from fate analysis to the amount of DALYs resulting from an emission a quantitative uncertainty estimate is given as the squared geometric standard deviation (σ_g^2), assuming a log-normal distribution.
The squared geometric standard deviation expresses the variation between the best estimate and the upper and lower confidence limits (97.5% and 2.5%).

Lower limit of the 95% confidence interval = best guess/σ^2_g
Upper limit of the 95% confidence interval = best guess*σ^2_g

The uncertainties are intended for use in software tools that apply Monte Carlo type analysis.
For the resources damage category no uncertainties are given.

2.5.2. Fundamental or model uncertainties

During the development of this methodology many modelling choices must be made on issues like:

- What shall be included and excluded in the model.
- What level of scientific proof do we require to accept a theory or hypothesis.
- What time frame do we take into account.
- Are health problems among young people as serious as health problems among elderly people
- Are future damages just as serious as damages that occur today or in other words should there be discounting.
- Are potential damages, that could be avoided if proper management is applied, less serious, or should manageable problems be disregarded (see also [HOFSTETTER 1998] on “manageability index”)

The basis for making such choices is often rather subjective. This can create serious problems, as in most cases such choices have significant effects on the result.

For instance, if we want to assess the damage caused by carcinogenic substances, we will see that there are 3 groups in the IARC classification. Group 1 contains the substances for which the carcinogenic effects to humans are proven, while group 3 contains only substances which are "not classifiable as to their carcinogenicity to humans" [WHO 1987]. The question of course is, which groups do we take into account. If we follow the precautionary principle, we would include all groups. If we were very pragmatic and we were to believe only in "hard scientific facts", we would probably only take group 1.

This example can be complemented by several other problems we have run into during the development work. We have to make subjective choices and we must be aware of this. Our solution is not to stop thinking when we encounter such a choice but to set up a management system for the subjectivity in each step.

THE CONCEPT OF CULTURAL THEORY

Hofstetter [HOFSTETTER, 1998] has analysed the problem of modelling subjectivity thoroughly and he proposes to use the Cultural Theory [THOMPSON et al 1990] to distinguish five basic value systems. Thompson derives these value systems by looking at the strength of the relation people have with their group and the degree an individual's life is circumscribed by externally imposed prescriptions (their "grid"). The viable combinations of the position of each individual in this group-grid typology and their cultural bias are called way of life. The assumption is that these viable combinations have a large influence on the value system of individuals and their groups.

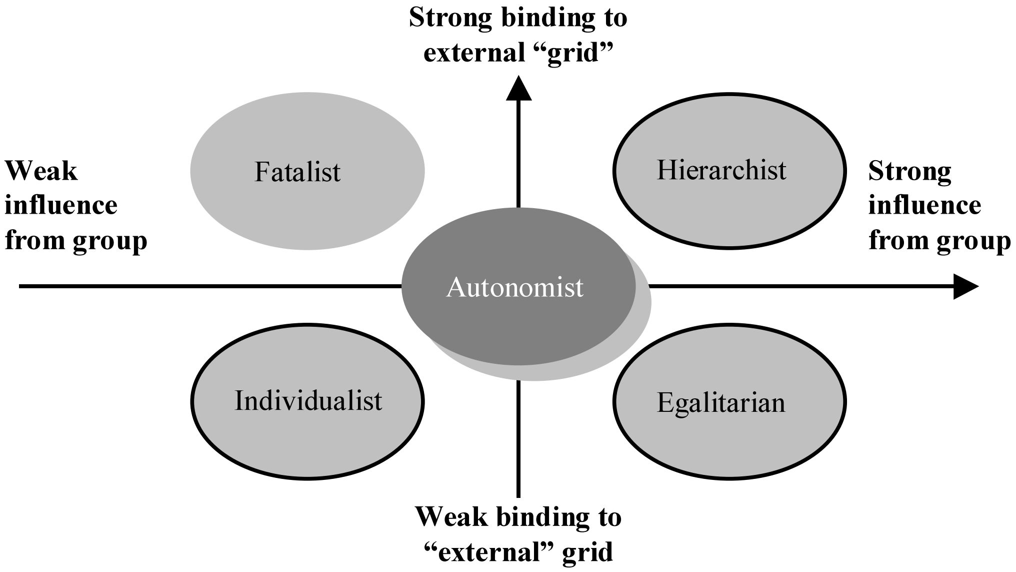

Figure 2.4: The grid-group dependency of the five archetypes distinguished in Cultural Theory. The Autonomist has no fixed position in this figure, because the Autonomist does not have social relations, and should be seen as floating over the other archetypes. Only the hierarchist,, egalitarian and individualist perspectives will be used.

These value systems have been used by several authors in risk perception studies. Experiences show that this distinction is very valuable in explaining people's attitudes. It is important to stress that this theory does not imply there are only five types of people. Almost nobody really conforms to the viewpoints of a single group in a consistent way. People can switch between different attitudes dependent on their context.

We can summarise an interpretation of the most important characteristics of the five archetypes:

1. Individualists, who are both free from strong links to group and grid. In this environment all limits are provisional and subject to negotiation. Although they are relatively free of control by others, they are often engaged in controlling others.
2. Egalitarians, who have a strong link to the group, but a weak link to their grid. In this environment there is no internal role differentiation, and relations between group members are often ambiguous, and conflicts can occur easily.

3. Hierarchists, who have both a strong link to group and grid. In this environment people are both controlling others and are subject of control by others. This hierarchy creates a high degree of stability in the group.
4. Fatalists, who have a strong link to grid, but not to a group. These people act individually, and are usually controlled by others.
5. Autonomists are assumed to be the relatively small group that escapes the manipulative forces of groups and grids.

There is sufficient evidence to assume that the representatives of the first three archetypes have distinctly different preferences as to modelling choices that have to be made.

The last two cannot be used. The Fatalist tends to have no opinion on such preferences, as he is guided by what others say. The Autonomist cannot be captured in any way, as he thinks completely independent.

The real value of the Cultural Theory is that we can predict a wide range of basic attitudes and assumptions for the three remaining archetypes. From these predictions the perspectives can be used to provide a basis for important modelling choices per archetype. As a result, three different set of value choices, and thus three versions of the damage models can be developed.

The basic attitudes related to the value systems that can be used in Eco-indicator 99 are summarised in Table 2.1.

Archetypes: ***Predictions:***	**Egalitarian**	**Individualist**	**Hierarchist**
Criteria	Argument	Experience	Evidence
Management style	Preventive	Adaptive	Control
Distribution	Parity	Priority	Proportionality
Perception of time	Long term dominates short term	Short term dominates long term	Balanced distinction between short and long term
Intergeneration responsibility	Present < future	Present > Future	Present = future
View of resources	Depleting	Abundant	Scarce
Perception of needs and resources	Can manage needs, but not resources	Can manage needs and resources	Can manage resources, but not needs
Energy future	Low growth (radical change now)	Business as usual	Middle of the road (technical fix)
Attitude to nature	Attentive	Laissez-faire	Regulatory
Attitude towards humans	Construct Egalitarian society	Channel rather than change	Restrict behaviour
Attitude towards resources	Need reducing strategy	Manage needs and resources	Increase resources
Perception (myth) of nature	Nature ephemeral	Nature benign	Nature perverse/tolerant
Perception of human nature	Born good, malleable	Self-seeking	Sinful
Attitude towards risk	Risk-aversive	Risk-seeking	Risk-accepting

Table 2.1: Typical values in the three different perspectives [THOMPSON et al., 1990, taken from [Hofstetter 1998]

The use of Cultural Theory has wide implications for the methodology. The most visible effect is that we will not have one model but three versions of the model [HOFSTETTER, 1998]:

1. In the *individualist version*, we choose to include only proven cause effect relations, when we have the choice we will use the short-term perspective. The preference for proven relationships is the attitude of individualists to consider each limit as negotiable. Of course also an individualist will accept that a limit is not negotiable if sufficient proof is given. Similarly it is not proven that long term effects cannot be corrected by the progress of science and societal developments. For human health issues age-weighting is used, since in the Individualist perspective a person is valued higher at the age between 20 and 40 years.

2. In the *hierarchical version* we choose to include facts that are backed up by scientific and political bodies with sufficient recognition. The hierarchical attitude is rather common in the scientific community, and among policy makers. A typical example is the wide acceptance of the IPCC guidelines for climate change.
3. In the *egalitarian version*, we consistently use a precautionary principle. We try not to leave anything out and if in doubt we include it, as egalitarians do not accept guidance from internationally accepted scientific or political organisations. Similarly we use the very long time perspective, as egalitarians do not accept that future problems can be avoided. It will be clear that this version is the most comprehensive version, but it also has the largest data uncertainties, as we sometimes have to include data on which consensus is lacking.

As a consequence, we will not have a single score for a material or process, but we will have three scores depending on the perspective. This may seem like a nuisance to the user, but it actually correctly reflects the fact that the judgement of environmental problems is not objective. By presenting three choices to the user, the user can choose the most adequate perspective. Of course it is possible to make an average of the three perspectives. It is even possible to make a weighted average, as there is data on the distribution of people with a certain perspective for many European countries. However, one must be aware that the average has a rather limited meaning. It is the average of fundamentally different views. As a result the uncertainties in the result will become very large, and it will become very difficult to explain the meaning of the result.

We recommend using the hierarchical version as the default method. The reason for this is that most models work according to consensus building processes, and a balance view of long and short-term perspectives. This means most models implicitly or explicitly based on the hierarchical perspective. To put it bluntly, if we would not have differentiated the perspectives, the Eco-indicator 99 would have been based on the hierarchical perspective.

The other two perspectives can be used as a robustness or sensitivity analysis. If the conclusion drawn from an LCA remains the same, independently of the perspective, we can conclude the result is independent of the perspective and thus on assumptions of time frame, the required level of proof, age weighting etc. If the conclusions change, we can conclude that the answer depends on the perspective. This is also very important information.

CULTURAL THEORY IN THE FINAL WEIGHTING

So far the use of Cultural Theory in the damage models was discussed. We used the concept of Cultural Theory also on the damage assessment phase. Earlier experiences [KORTMAN et al 1994] and [HOFSTETTER in chapter 8 in BRAUNSCHWEIG et al 1996] show that the behaviour of each panel is very much determined by the basic value system a person is using. In order to analyse the influence of the perspectives, we included a number of standard questions that should reveal the perspective each respondent adhered to while answering the questions. Although the sample size was rather small we found statistical significant differences between the weights given by respondents and the perspective they seemed to adhere to [METTIER 1999]. However, due to the small sample sizes we recommend to use the averaged weighting factors. Only in the sensitivity analysis as described above we recommend to use the weights per perspective.

2.5.3. Uncertainty due to incompleteness of the model

In our ambition to develop the damage models for all relevant effects that can contribute to the three damage categories, we took on a task that can never be completely finished. This means we had to leave out mechanisms we could no (yet) model. The problem is that, as we cannot model them, it is difficult to state how important these omissions are, although it is possible to get an impression sometimes. Of course the importance depends also on the application. For instance the lack of a damage model for phosphate in water is much more important for an LCA of detergents than for an LCA on transport systems.

As the three perspectives have different criteria for inclusion of the damage mechanisms, not all perspectives would include all models, even if they were available. The table below gives some examples of damage models that are missing or yet incomplete. With relevance we mean the probability that exclusion of a damage mechanism can lead to distorted results in an average LCA. A plus indicates whether the exclusion of a specific effect can cause a distortion of the results. Again this list is just intended to give some examples.

	Relevance
Human Health	
Effect of Noise	+
Other toxic effects from heavy metals and persistent organics.	+
Effects on hormonal systems (Endocrine disrupters)	?
Effects on the immune system from increased UV	?
Other diseases from climate change	?
Ecosystem Quality	
Phosphate in aquatic systems	+
Effect of climate change and increased UV on plants	+
Effect of increased ozone levels on plants	+
Effects on higher animals	?
Effects on other ecosystem functions besides the species diversity	?
Resources	
Silvicultural resources	?
Other minerals	?
Effect on other resource parameters besides surplus energy	?

Table 2.2: Some examples of missing damage models and the expected relevance. Not all perspectives would consider inclusion of these models relevant. Egalitarians would not exclude anything. Individualists would only include models for which sufficient proof is available that there is indeed a significant effect of damage. Hierarchists would exclude damages that can easily be avoided if proper management is applied.

Box 2.2 Compatibility with ISO 14042

While the development team has closely followed the development of the ISO 14042 on Life Cycle Impact Assessment, it was never a goal in itself to adhere to this standard. Scientific progress and coherency was regarded far more important than standardisation. However in the end it appears that the differences between the Eco-indicator 99 methodology and the requirements put forward in the ISO 14042 text are quite small.
ISO distinguishes between mandatory and optional elements (§4.3). The most relevant mandatory element is the selection of the impact categories and category indicators. The ISO standard allows for selection of the category indicator at the level of the endpoints, but in one of the recommendation (§5.3) it states that the modelling must be performed in a scientifically and valid manner. Of course it is a matter of debate whether the models used here are considered to be sufficiently scientifically valid.

The standard describes three optional elements (§6); normalisation, grouping and weighting. In the Eco-indicator we do all three, more or less conform the standard. The most important deviation is that we do not follow the recommendation (§6.2) to define the normalisation value on different temporal and spatial scales. An important provision (§9) is that weighting is not allowed in the case of comparative assertions disclosed to the public. As the use of the single scores is intended for internal product improvement applications, this provision is no real problem. For comparative assertions disclosed to the public we recommend to apply the weighting triangle presented in box 7.1.

As a conclusion, we can state that there is no real conflict with the ISO standards, while it is on the other hand not completely clear if the method satisfies all requirements. Much depends on how the ISO standard will be interpreted in real test cases.

Box 2.3 Relations with the Dutch CML2 project

While this project has been running, in the Netherlands the so called LCA-2 project was started. The latter project aims at revising and updating the well-known theme-oriented CML 1992 Guide to LCA [HEIJUNGS et al. 1992]. The project is carried out by CML in Leiden. In this project all new developments in the LCA methodology are included, and the method is updated. Although the project is completely independent from the Eco-indicator project, close contacts and an open information exchange have been maintained all the time.
Although the results are not yet available, the main result regarding impact assessment is an updated theme oriented approach, and not a damage oriented approach. This means that the projects do not compete, but supplement each other.

3. Fate analysis for mass loads

3.1. Description of the problem

An important feature of the inventory result is the lack of spatial and temporal information. The inventory result of an LCA only specifies a mass, without specifying where, and with what rate this mass is released[4].
Lead emissions that occur in the raw material extraction phase are added to lead emissions that occur from electricity generation (in a usually large number of power plants). Next they are added to emissions from leachates from future decomposition of the products in many landfills over hundred or more years. This means we only know the total quantity of an emission, without knowing when and where the emissions occur and what the concentrations in the environment are.

This lack of information on concentrations is a major obstacle in damage modelling, as damages are not caused by releases (expressed as a mass unit), but by concentrations, or in a marginal model, changes in concentrations. A procedure is needed to convert discrete releases (in mass units) into concentrations.

3.1.1 Fate models

There are numerous models available to establish a relation between a flow of a substance and a concentration. These models are called Fate models, as they model the fate of a substance. This is important as substances degrade, or are transferred to areas that can be regarded as sinks. A sink is a place that can be seen as a final destination.
As all substances ultimately disappear from the environment, a steady state concentration can only be related to a flow, and not to a discrete mass release. The problem with most fate models is that it is difficult to model dynamic situations. Basically they are designed as steady state models that can be used to calculate the steady state concentration as a result of a steady state emission.

Even with fate models that calculate dynamic changes, there is still be a problem, as the inventory result does not contain information on the shape of a pulsed release. Suppose a factory makes 1000 electric razors per hour, and has an emission rate of 1 kg per hour of a certain substance. If the effects of a single razor are to be modelled, it is permitted to say that one razor produces an emission of 1 kg during 3,6 seconds, but one could also state it produces a flow of 1 gram per hour, or 1/24 grams per day. The problem becomes even more complicated if the emissions from the other life cycle stages are added, such as the intermittent electricity use every morning, or the steady flow of emissions from a land-fill after disposal.
Clearly, a solution is needed for this fundamental so-called flux-pulse problem.

3.1.2. The flux-pulse problem

The flux-pulse problem can be solved in several ways [GUINÉE ET AL 1996]. In this project a time allocation procedure has been used, that can be summarised as follows (see also box 3.1).

- In the fate models a steady state flow is used to calculate a concentration. This flow rate is the assumed pulse height.
- The discrete emission from the inventory table is divided by the flow rate, the result is a certain time span. (the length of the pulse)
- As a result a discrete emission can be related to a steady state flow during a certain time span.

As a consequence all damages are expressed as temporary damages as a time span is added[6]. This is not strange, as the increase in concentration created by a product will by definition be temporary.
The selection of the pulse height is of course proportional to the time span found. In the damage models developed in this project it is permitted to use any pulse height, as all models are linear.

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[4] For ionising radiation, the unit of becquerel is meant here, for land-use the unit is area times time, but in order to keep the text readable, the case of mass is described.

[6] This is independent of the question if the damage is reversible or not. If a mass load contributes to an irreversible damage, it can only be held responsible for that irreversible damage during a certain time period.

The following fate models are used in this project:

- Substances with carcinogenic effects and substances with ecotoxicological effects, are modelled with EUSES [EUSES 1996].
- Substances that cause respiratory effects are modelled with atmospheric deposition models and empirical observations, as described by [HOFSTETTER 1998].
- For ozone depletion and climate change, the fate models from several studies are used.
- For acidification and eutrophication the SMART model developed by the RIVM [KROS 1995] is used. This is a GIS based multi-media model.
- For radioactive substances a number of box models representing different sites from the French nuclear fuel cycle are used [DREICER et al 1995].

In this chapter only the fate model for human carcinogens and ecotoxic substances is described, as this model calculations have been carried out by the development team, and must be documented consequently. The other fate models are taken from existing literature and are described in the impact category paragraphs.

Box 3.1 Explanation of the flux-pulse problem

When we make an LCA of an electric razor that is to be produced next year, we can assume that the emissions from the mining process have already occurred, while the emissions during use will occur in the next ten to twenty years. The emissions from disposal, especially if the razor is put in a landfill, will probably occur in the next century.

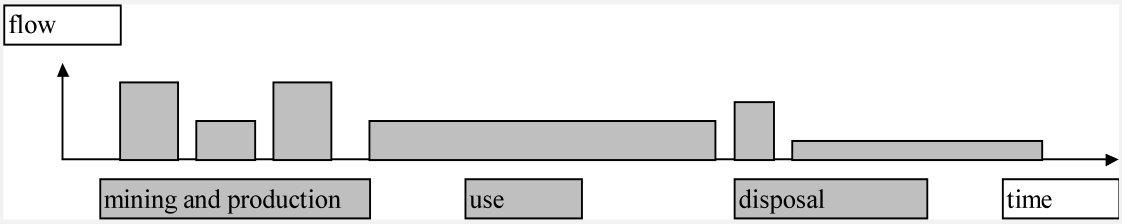

Fig. 3.1 General representation of the flows from a life cycle

It is meaningless to determine the actual flows to the environment from this razor, as it is impossible to know where and when they take place. For instance we can assume that the razor creates a pulse as large as the total emission from the factory during a second, or that the production of a razor results in an emission pulse that is one thousands of the total emission during 1000 seconds. Both assumptions are correct, but would lead to completely different pulses.

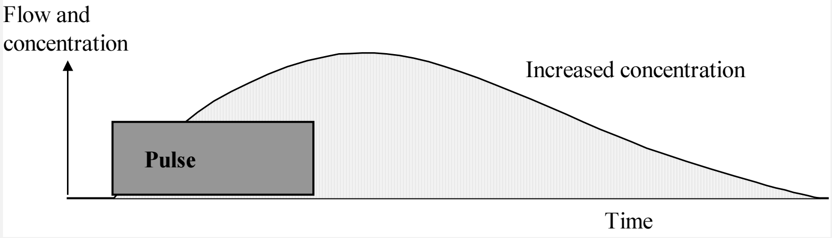

Figure 3.2 Transient concentration from a pulse

A mass loading will only cause a temporary increase of the concentration, as all substances in the environment will be degraded, immobilised permanently or diluted to natural background concentrations sooner or later. Although we cannot model the real flow into the environment, we can assume the mass specified in the inventory table will be released in the form of a pulse. The product of flow and time period is equal to the mass. In figure 3.2 a possible transient response of the pulse is plotted.

The fate models used here can only calculate steady state concentrations from a steady state flow. This means we cannot calculate the transient increase. Instead we have to find another solution[7]. This solution can be described in both a practical as a more mathematical way. We shall use both explanations.

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[7] This problem is also described in [Guinée et al 96]. This report describes 8 possible solutions and selects one, that uses a reference substance. The idea is that the time constant does not need to be determined, as long as it is the same for all substances. As a consequence the concentrations can only be determined relative to each other. For the purpose of the study mentioned this is an appropriate solution as they aim only to calculate relative differences between substances.

It is estimated that there are at least many millions of different mass-produced product designs in the world. Many of these designs are mass produced. This means there must be billions to trillions of individual product life cycles taking place now. All these product life cycles contribute to a more or less steady state flow of substances into the environment.
Let us simplify this situation and split up the world production into three types of product groups, each type representing 2 million product designs. Suppose that for each of the three main product groups, one generic life cycle is operational. The three product groups emit flows at different times and at different magnitudes. Since those three product life cycles represent the total world production, the combined effect is in reality more or less a steady state flow. This can be illustrated by the figure below, which shows the steady state concentration that is the result of the steady state flow of all the products in the three product groups. Furthermore it shows which product is responsible for which part of the concentration.

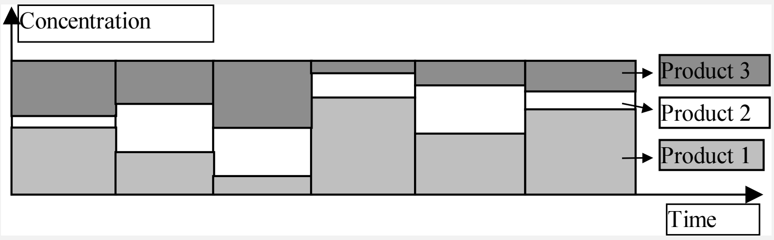

Figure 3.3 A constant flow caused by three fictional product life cycles

As in LCA we disregard the information on when an emission occurs, this figure can also be presented in an alternative way, as in figure 3.4.

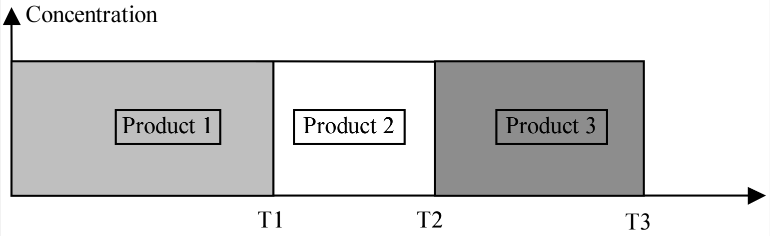

Figure 3.4 Alternative representation of the flows from three product life cycles.

In this presentation we have the same steady state concentration, but it is ALLOCATED in a different way to the three product groups. We may do this, as all the fate models and the following damage models are in principle linear. It is important to state that we do not assume the concentration “pulse” caused by product 1 has the actual shape as presented in figure 3.4. The “pulse” depicted here is the result of rearranging all the small pulses a product produces over its lifetime. The “pulse” is the result of an allocation procedure.

The advantage of working with this allocation procedure is that now we can say that a mass load will be held responsible for a certain steady state during a certain time. This can be done for the large mass loadings resulting from one of three main product groups, but also for an individual product, which is a small part of this total group of product. The fate model links the mass to the product of pulse height and pulse length. Again as all models are linear, it is not necessary to know the duration or the pulse height. Only the product of pulse height and duration is important.

Having developed this principle for the concentration, we can also develop the same concept for the damages. All damage models link a steady state concentration to a steady state damage. As a mass loading can only be held responsible for a temporary concentration, it can also be held responsible for a certain amount of damage during a certain period of time.

In algebraic terms we can describe the model with the simple equation:

$F = M/T = a*A*PEC$

with:

F	flow [kg/yr]
M	mass [kg]
T	time [yr]
a	model constant, determined through EUSES [m/yr]
A	Area [m^2]
PEC	predicted environmental concentration [kg/m^3]

The equation can easily be adapted by multiplying the left and right hand side with T:
$M = a*PEC*A*T$
The mass is causing a concentration over an area over a certain time. As the damage is proportional to concentration, we can say a mass is causing a damage over a certain area, during a certain time.

3.2. Fate analysis with EUSES

3.2.1. Introduction and calculation principles

For the fate analysis of carcinogenic substances causing damage to Human Health and ecotoxic substances causing damage to Ecosystem Quality, the European Uniform System for the Evaluation of Substances [EUSES 1996] is used. EUSES is designed for quantitative assessment of the risks posed by new and existing substances to man and environment in the European Union. The model is based on the EU Technical Guidance Documents (TGDs) and the parameter settings are accepted by the European Commission to give a proper representation of the European Environment.

The EUSES program includes a multi-media environmental fate model, which is the follow up of USES 1.0 that was applied by [GUINÉE ET AL 1996]. In these models, different environmental media (air, water, sediment, and soil) are modelled as homogeneous, well-mixed compartments or boxes. Inter-media transport and transformation are represented by simple, first-order processes. The distribution of a substance over the different compartments is determined by the physical/chemical properties of the substance and the characteristics of the environmental compartments which induce transfer processes between compartments and degradation of the substance [EUSES 1996].

In assessing the fate of emissions we suppose that the emissions are released within Europe. Furthermore we assume that Europe is closed and emissions are distributed evenly over the continent, resulting in an average European concentration in air, water and soil. This means that local, acute effects from high concentrations of substances close to the source of the substance are not accounted for in the method.

An emission from Technosphere forms the input of substances into the environment. Substances disappear out of the environmental compartments by degradation and by deposition into a sink. In the EUSES model there are two sinks. Via vertical transport of substances (leaching) through the soil, the substances will eventually end up in the deeper groundwater or substances associated with suspended solids can be buried into sediments permanently. Steady state concentrations are reached when the disappearance of the substance balances the emission input [EUSES 1996].

For the fate analysis a steady state concentration (Predicted Environmental Concentration or PEC) of a substance in different compartments is calculated using a standard flow. From these concentrations, the damage to ecosystems for different toxic substances can be calculated. For human exposure the concentration of substances in drinking water, crops, meat, milk and air is calculated from the PEC. With information on human intake of air, drinking water and food, the total human exposure to a substance is calculated by the model.

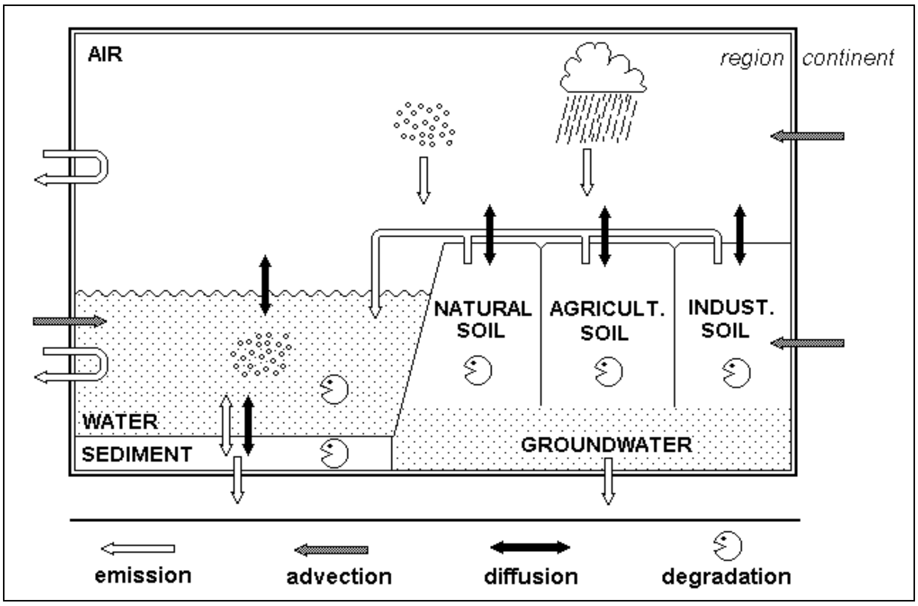

Figure 3.5: Transfer of substances in EUSES [EUSES 1996]

Box 3.2 Fate models for other parts of the world

Fate models, such as EUSES are also available for other parts in the world. In Japan an adapted version of USES is being developed. In the US several systems are available or under development. Recently a comparison was made between our work and the work of Hertwich using the US model CALTOX by [JOLLIET ET AL 1998]. Aside from some input errors, this comparison showed a reasonable fit for most organic substances, given the fact that the models were tuned to different regions. It also showed the difficulties when modelling the behaviour of heavy metals in air in such models. [GOMEZ 1998] adjusted the fate model for Latin America.

3.2.2 The EUSES model, input data and specific model settings

The EUSES model is designed for organic substances. For organic substances general principles describing the relation of substance properties and the behaviour of the substance in the environment can be used as calculation rules within the model. For metals there are no such general principles and fate analysis for metals is more complicated and requires specific model settings. Fate analysis for metals that cause cancer in humans or have serious ecotoxicological effects is described separately in box 3.3.

The range of substances included in the impact category human carcinogenicity is determined by the extensive work of Hofstetter, who performed damage analysis for carcinogenic substances [HOFSTETTER 1998]. For carcinogenicity fate analysis is performed with EUSES on the best known 52 substances included in IARC groups 1, 2A, 2B and 3. The total amount of substances included in these IARC groups is about 500. However, the inclusion of substances in the calculations depended upon the availability of data on physico-chemical characteristics and dose-response relations.

The range of substances included in the impact category of ecotoxicity is limited by the availability of damage information (PAF-curves). RIVM has constructed damage curves for a selection of 43 priority substances. It is assumed that these substances are the most important contributors to the damage to ecosystems in the Netherlands.

The substance data[8] used for calculations in EUSES are mainly derived from recent work from [HOFSTETTER 1998] and RIVM [BAKKER AND VAN DE MEENT 1997]. In both studies an extensive literature search on substance data has been carried out in order to provide the most up-to-date data set.

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[8] The first results of the EUSES calculations have been reviewed by comparing EUSES calculations with results from another Mackay model (Caltox) [JOLLIET 1998]. This comparison revealed some input data errors. These errors have been corrected and the results have been recalculated.

Only in a few cases data from the Handbook of environmental data [VERSCHUEREN 1996] or the CML data set which was used in [GUINÉE ET AL 1996] have been used.

The data for area size and total number of inhabitants are derived from the OECD environmental data 1995. In the European continental system the number of inhabitants and the total surface area are based on the countries from the EU and the EFTA without Iceland. Iceland is excluded because it is too different from the rest of the countries and the continental system is supposed to be a homogenous system, with only the surface water included.

For the fate analysis calculations in EUSES the regional scale is used. The regional and continental models are in fact identical, with equal parameter settings and emission input, the results of the model calculations will be exactly the same. However, the advantage of the regional model is the possibility of directly calculating the exposure of humans through food and drinking water. The standard EUSES settings, based on the consensus of international expert groups for the European continental model have been the basis for the calculations. However, some of these standard defaults have been adjusted to meet the requirements of the Eco-indicator methodology.

3.2.3. Simulating a closed Europe

EUSES has not been designed for LCA purposes and must be modified in order to get the desired results. To be able to determine the total effect of an emission the effects of transboundary pollution should be excluded, since the leaching of emissions to areas outside Europe will cause an underestimation of the concentrations. Consequently, flows of emissions into the system must also be excluded. Some of the model parameters are adjusted to isolate the system from the surrounding compartment. Normally the regional scale is open. By minimising the wind speed and run-off transfer from substance to outside areas is minimised. However, these artificial settings influence the distribution of the substance within the regional scale of the model. Especially the accumulation of heavy metals in soil and the water concentrations are influenced.

The net influence of these modifications on the calculation results has been checked in a calculation with the Simple Box 2.0 model, which can be closed entirely [JAGER 1998]. The conclusion from this check is that by closing the model less than 1% of the substance is lost to areas outside Europe, whereas with an open model the losses can be up to 25%, depending on the type of substance. The relatively small decrease in water concentration is acceptable, since this compartment comprises only 3% of the total area of Europe and this effect is amply compensated by higher concentrations in the soil. This means that the adjustment of the parameters does not influence the stability of the model and the reliability of the resulting concentrations in a serious way.

3.2.4. Fate analysis for heavy metals

Fate analysis for heavy metals that cause carcinogenic effects in humans and heavy metals that cause ecotoxicological effects is carried out combining EUSES with other information sources and adjusting the calculation results accordingly. For human exposure through air model calculations are replaced by estimates based on measured or experimental data [A.O. HOFSTETTER 1998]. Exposure to air emissions through drinking water is recalculated with the corrected air concentrations. For human exposure to heavy metals through food data from [HUIJBREGTS 1999-1] on transfer coefficients are used to recalculate concentrations in food. In box 3.3 the fate analysis is described in more detail.

Box 3.3 Fate analysis for heavy metals

Specific model settings

EUSES is originally designed for organic substances. The fate of metals in the environment is much less predictable and the fate modelling is very complex. It is possible to use EUSES for fate modelling, but different input data are required [EUSES 1996]:

- For water solubility estimates have been made using average circumstances in natural waters (pH, concentrations of suspended solids etc.).
- The octanol-water partition coefficient is not relevant for metals. Measured partition coefficients have been used for partitioning of metals between water-soil, water-sediment and water-suspended matter.

- Most of the metals present in the atmosphere will be associated with aerosols. Therefore an extremely low value for vapour pressure has been used to make the model estimate the fraction bound to aerosol close to 1 (except for Mercury).
- Volatilisation can be ignored for metals, except for mercury. The Henry-constant has been set to a very low value.

Degradation and residence time for metals
The influence of degradation time of substances on the concentrations is linear. This means that increasing the degradation with a factor of 10 in a specific compartment will lower the concentration with the same factor if degradation is the main influencing factor. In the EUSES model it is possible to choose a degradation of 0. Within an almost entirely closed system and a degradation of 0 the retention time of persistent substances like metals depends on slow processes e.g. the transport to deep groundwater and the burial of the material in sediments. Therefore the retention times of metals and other non-degradable substances are very long and the resulting steady state concentrations are high. Since it is normal for these substances to accumulate in the environment, this scenario is not unrealistic. However, the time necessary to reach steady state for this persistent substances under this circumstances is a few hundred until a few thousand years, depending on the emission compartment and the substance. This is not very different from a "normal" situation for the European continental system, where the ocean sediments are used as an additional sink[9].

A time period of several hundreds to several thousands years to reach steady state means that the present emission level has to be maintained for the same period of time in order to reach this steady state concentration. It also means that the total effect of a present emission will occur in the far future. Choosing to calculate concentrations with 0 degradation thus implies that all the future generations over the next few hundreds to thousands years are considered equally important as the present population (Egalitarian principle).

Another option, which was used in the CML project "Impact assessment of toxic releases" [Guinée et al 1996] is to choose a time horizon. In the CML project a degradation half life of 10,000 days was chosen, resulting in a total degradation time between 100 and 200 years. The real residence time of metals in the system, according to the model calculations is less than a 100 years, because the degradation is not the only influencing factor. The practical reason for this was the fact that the 10,000 days was the maximum for the USES model, used in this project. In EUSES the concentrations in the environment using a half life of 10,000 days are between 1 and 2 orders of magnitude lower than the long term steady state concentration.

For metals the short term and long term results have both been calculated. The choice of a time horizon is subjective and depends on cultural perspective (see section 3.3).

Calculation procedures for metals
For human exposure through air fate factors for metals calculated by EUSES have been rejected. The model results do not concur with literature data and give an underestimate according to [HOFSTETTER 1998] and [JOLLIET 1998]. Since respiratory intake is an important intake route for many substances in case of air emissions, fate factors derived from measured data give more reliable results. The work of [HOFSTETTER 1998] provides these fate factors, which are between 1 and 2 orders of magnitude higher than the values calculated by EUSES.

For human intake through drinking water results have been recalculated with EUSES using the resulting higher fate factors for air. However, the change in the air concentration has only a very small effect on the concentration in drinking water.

For food intake the model can not produce reliable results for metals, because there is no way to adjust the model for properties of metals. The uptake through leaf crops, root crops, meat, milk and fish is different and very specific for each metal. Therefore the food intake is based on calculations using experimental data from integrated criteria documents on specific metals [JANUS AND KRAJNC 1990] [ROS AND SLOOFF 1987] [SLOOFF ET AL 1990A] [SLOOFF ET AL 1990B] and data for transfer coefficients from [HUIJBREGTS 1999-1]. In the integrated criteria reports data on average environmental concentrations and concentrations in food in the Netherlands are specified. From these data the transfer coefficients for transfer from environmental compartments to food products are estimated, in case they were not available in [HUIJBREGTS 1999-1]. These transfer factors are combined with the intake of food specified in EUSES and the environmental steady state concentrations calculated from standard flows in EUSES to calculate the human exposure through food. The resulting fate factors demonstrate that for metals the intake pathway through food is very important as is stated before by [HOFSTETTER 1998].

For ecotoxicity the damage to Ecosystem Quality is caused by exposure to environmental concentrations in soil and water. The resulting air concentration is therefore of lesser importance. For ecotoxicity the environmental concentrations for metals have been calculated using the specific model settings adjusted for the metals as described above.

3.3. The role of cultural perspectives

In fate analysis with EUSES for substances that have carcinogenic effects on human health or toxic effects on ecosystems the role of cultural perspectives is important in two aspects:

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[9] See also the Greenpeace (1999) report "the tip of the iceberg" on global distillation and the concentration of persistent pollutants in the Arctic and Antarctic regions, available via the web.

- For metals there is a choice in the length of the time horizon. In the long term perspective the steady state concentration is calculated for all substances, irrespective of the time that passes before steady state is reached. This means that for some metals it may take thousands of years before this concentration actually occurs. In the egalitarian point of view future generations are equally important so the long term perspective is used. Since the Hierarchist does not favour the present over the future, the Hierarchist also chooses the long term perspective. In the hierarchist perspective there is no scientific reason to choose a specific time horizon. In the short term perspective that is only used by the Individualists, only the near future is relevant. In the individualist perspective the maximum concentration in the environment with a half life of 10^4 days as was used in [GUINÉE ET AL 1996]. Depending on the type of substance and the initial emission compartment, with this half life, the steady state is reached within a 100 years.
- The use of a closed system for Europe could in fact be considered an egalitarian point of view. An artificial situation is created to make sure that no underestimation of the effects occurs. Individualists could argue that the effects of emissions that blow into the oceans is not proven and adjusting the model in a way that all emissions stay in Europe is not justified. However, within this project it was not possible to calculate fate analysis for over 100 substances in two different ways, because it would take too much time. Therefore only the closed Europe, the Egalitarian perspective is used. This causes a discrepancy in the results of the Individualist perspective. The difference between an open and a closed system in the concentrations is probably substantial for air emissions.

3.4. Sources of uncertainty

In fate analysis there is a lot of uncertainty in the results. The use of a multi media fate model for calculation of environmental concentrations based on substance properties also introduces uncertainty. The main reasons are [HOFSTETTER 1998]:

- The linear and proportional nature of the exchange rate between compartments and accumulation within compartments does not reflect reality.
- The assumption of uniform concentrations within compartments is relevant for human exposure but does not hold in reality.
- The substance specific physico-chemical and degradation data are themselves subject to uncertainty due to weakness in the analytical techniques and to their variability.
- The Mackay type models are inappropriate for some of the substances (see section 3.2)

The key parameters that influence the predicted environmental concentrations are:

1. Partitioning. For substances that show minor partitioning between compartments, the model results can be quite reliable. This is logical, because the uncertain relations describing the transfer of substances from one compartment to the other are not used in the calculations.
2. The properties of the substance are uncertain. Calculation based on these uncertain properties give results with high uncertainty.
3. The substance is not well suited for the model. The calculation rules do not describe the transfer and degradation processes for this substance in a proper way.

A quantitative estimate (geometric standard deviation) of the uncertainty for fate factors calculated with multi media fate models like the EUSES model has been made based on expert opinions [SETAC 1995], experience with the models and literature reviews [HOFSTETTER 1998]. The geometric standard deviation (σ_g^2), based on the 3 key parameters, varies between 2 and 80.

For a substance that shows minor partitioning, is suited for the model and has properties that are well known the σ_g^2 for the fate factor is 2 , when the emission compartment is also the receiving compartment (emission to air, concentration in air) and a factor of 4 to 8 in case these are different (emission to air, concentration in water or food). This means the difference between the average and the 97.5% and 2.5% is a factor of 2 to 8. For a substance that shows major partitioning, is not suited for the model and has properties that are not very well known, the σ_g^2 for the fate factor is 20, when the emission compartment is also the receiving compartment and a factor of 40 to 80 in case these are different.[HOFSTETTER 1998]

The use of a risk assessment model for fate analysis in LCA based on "average Europe" model settings causes a mismatch in the human exposure. The exposure pathways are maximised, which means that the human intake of harmful substances is not average but based on a worst case approach. In this approach the consumption of fish, crops, meat and milk are maximised. Because there are no average European values for human exposure that are generally accepted the worst case approach had to be used for calculation of fate factors for intake through food. For exposure through inhalation and drinking water the damage assessment is based on concentrations in air and drinking water, and the worst case intake for these exposure pathways is not used for calculations.
The worst case approach incorporated in the EUSES diet derived from risk for human intake of food causes a deviation from the average effect. From EUSES calculations with an average diet based on literature data on eating habits, the deviation caused by a worst case diet can up to a factor of 10.

The change of the model simulating a closed Europe may also add to the uncertainty, because the transfer between compartments is influenced.

4. Human Health

4.1. Description of the damage category

The World Health Organisation defines health in a very comprehensive way: Hhuman health is "a state of complete physical, mental and social well being, and not merely the absence of disease or infirmity".

Further WHO states that environmental health (of humans) "includes both the direct pathochemical effects of chemicals, radiation and some biological agents, and the effects (often indirect) on health and well-being of the broad physical, psychological, social and aesthetic environment, which includes housing, urban development, land-use and transport [WHO 1995, p 24].

In LCA we can only deal with some of these aspects:

- LCA only accounts for anthropogenic emissions in air, water and soil, thus excluding unhealthy conditions at workplaces and homes, traffic accidents, drinking or smoking.
- LCA also does not deal with health problems caused by natural disasters, climate, micro-organisms or volcanic eruptions.
- LCA in general does not address economical aspects [ISO14040, p.4]; thus the consequences of low income are not taken into account.
- All other aspects of welfare apart from absence of disease.

These limitations in the scope of an LCA require us to define the damage category Human Health much more restrictive than the WHO does. Human health means the absence of premature death, sickness or irritations caused by emissions from industrial and agricultural processes to air, water and soil.

4.2. The concept of DALYs

If we want to quantify the damage category Human Health, it is important to find a scale to measure health of a population. Preferably we want to use an existing scale.

There are several factors that should be included:

- The number of individuals affected by the problem.
- The time humans suffer from the limitation, or the lifetime lost by premature death.
- The severity of the health problem, ranging from premature death to irritation.

Many attempts have been made to scale the severity of health problems. For instance in most Western countries such a scale is used to determine the welfare or ability to work for ill and (partially) disabled persons. In these systems the relative degree of disability is the measure in this scale.

On the international level, a single indicator to quantify the total burden of disease was developed by Murray for the Global Burden of Disease study, collaboratively undertaken by the World Bank and the World Health Organisation. It is intended to be used in health economics as an objective tool to allocate money to health care [World Bank 1993].

This health-indicator, expressed as the number of Disability-Adjusted Life Years (DALYs), measures the total amount of ill health, due to disability and premature death, attributable to specific diseases and injuries. The DALY concept thus compares time lived with disability (YLD: Years Lived Disabled) and time lost due to premature mortality (YLL: Years of Life Lost). Health is simply added across individuals. That is, two people each losing 10 years of disability-free life are treated as the same loss as one person losing 20 years [Murray et al 1996].

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[10] These excluded conditions cause a much greater damage to health than the emissions covered by LCA. For instance environmental conditions are estimated [Scientific American 1996] to contribute only 2 percent of all cancer cases; this is similar to the risk of food additives, like salt. For certain products, unhealthy conditions in homes should be included, but this is the choice of the LCA practitioner.

4.3. Discussion on DALYs in the Eco-indicator 99

Not everyone appreciates the ethical dimension of health status indicators. However, most choices in allocating resources in health care deal with a number of variables and are the implicit choices of decision makers. The intention of the designers of this indicator is to provide an integrative, comprehensive methodology to make explicit choices that are open for debate and public scrutiny.

There has been little formal empirical work on measuring individual preference for age weights. In fact, most of the choices made in this method have not been extensively verified against the general public opinion, but are based on a revealed preference of a group of specialists. However, sensitivity analysis has shown that the quantitative results of the Global Burden of Disease study are remarkably insensitive to the particular social preferences incorporated into the calculation of the DALYs.

Using the concept of the DALY method to quantify the damage to the damage category Human Health has several advantages. All important factors that should be included in LCA, like the number of individuals affected by the problem, the time humans suffer from disabilities and the lifetime lost by premature death, as well as the severity of the health problem, ranging from premature death to irritation are included in the health indicator and can be quantified theoretically. Another advantage is that the method is very transparent and the social preferences that are included could be adjusted if necessary.

In the Eco-indicator '99 methodology this quantification method for the damage category Human Health is used to value the different disabilities caused by environmental impacts. The health scale is used to estimate DALYs due to disease caused by environmental conditions.

Some adjustments of the complete method have been made by [Hofstetter 1998] who revised the DALY method in the light of cultural theory for use in LCA. According to [Hofstetter 1998] the lack of time information in LCA *sensu stricto* compels us to refrain from discounting future health states. Age weighting is included only in the perspective of the Individualist. It is assumed that Individualists assign greater importance to being healthy at younger years than the group-bounded Egalitarians and Hierarchists. In table 4.1 the characteristics for each perspective are demonstrated.

	Discounting	**Age-weighting**
Egalitarian	No	No
Hierarchist	No	No
Individualist	No	Yes

Table 4.1: Characteristics of the DALY-concept according to the three cultural perspectives.

Box 4.1 The technical basis for DALYs

The following text is derived from [Murray 1994] and [Murray et al 1996]

The DALY concept is based on several principles :
To the extent possible, any health outcome that represents a loss of welfare should be included in an indicator of health status. This concurs with the WHO definition of health.
Individual characteristics affected by a health outcome that should be considered in calculating the associated burden of disease should be restricted to age and sex. This means that only distinctions between individuals are made based on these two variables.
Treating like health outcome alike. This means that, for example, the premature death of a 40 year old woman expressed in DALYs should always be the same regardless of her social position and living conditions.
The unit of measure is time. The use of the general measure of time provides a simple and intuitive method to combine the time lived with disability, with the time lost due to premature mortality.
According to Murray [Murray 1994] four key social preferences or values must be incorporated into the indicator of burden of disease "DALY".

Duration of time lost due to premature death
Duration of time lost due to a death at each age, which is used to measure years of life lost due to premature mortality (or the number of years of life gained by averting death). This measurement requires defining the potential limit of life; in the case of DALYs, standard years of life lost are used. The potential limit of life expresses an ideal situation and therefore the standard has been chosen to match the highest national life expectancy observed, which is that of Japanese women (82 years).

For a specific standard, the expectations are based on a model life-table (West Level 26), which has a life expectancy at birth for females of 82.5. The potential life expectancy at birth for males has been set at 80. According to [MURRAY ET AL 1996] a sex distinction is made because of the, scientifically established, biological difference between men and women. A sex distinction can only be applied if the statistics are specific enough about this.

Non-fatal health outcomes
Disability weights or degrees of incapacity or suffering associated with different non-fatal conditions are necessary to make comparisons across diseases and for comparing time lived with a disability with time lost due to premature mortality.

Disability Class	Severity weights	Indicator conditions
1	0.00-0.02	Vitiligo on face, weight-for-height less than 2 SDs
2	0.02-0.12	Watery diarrhoea, severe sore throat, severe anemia
3	0.12-0.24	Radius fracture in a stiff cast, infertility, erectile dysfunction, rheumatoid arthritis, angina.
4	0.24-0.36	Below-the-knee amputation, deafness
5	0.36-0.50	Rectovaginal fistula, mild mental retardation, Down syndrome
6	0.50-0.70	Unipolar major depression, blindness, paraplegia
7	0.70-1.00	Active psychosis, dementia, severe migraine, quadriplegia

Table 4.2: Revised disability classes for the Global Burden of Disease Study [MURRAY ET AL 1996]

Disability weighting was done for 22 indicator conditions by estimating the extent of loss of physical functioning associated with a certain indicator condition. Based on these weights seven disability classes were formed. Subsequently, a group of independent experts established weights, ranging from 0 (perfect health) to 1 (death), for 100 indicator conditions. The disability classes were adjusted according to this detailed estimate (see table 4.2). Sensitivity analysis showed that changes up to 0.1 in the specific weights of class 3 through 6 have only a minor effect on the estimated total DALYs by cause. Class 1 and 2 however are more sensitive to changes.

Time preference
Time preference is the value of health gains today compared to the value attached to health gains in the future (in standard economic theory, the latter is assumed to be lower than the former). It is standard practice in economic appraisal of projects to use the discount rate to discount benefits in the future. The process of discounting future benefits converts them into net present-value terms. The discount rate used in the DALY formula is 3 percent. Time discounting is not used, as there is very little information on the time an impact occurs in the LCA framework. in the Eco-indicator 99 calculations the 0 discounting is indicated by the first zero in DALYs (0,0).

Social value of the time lived at different ages
In all societies social roles vary with age. The young, and often the elderly, depend on the rest of society for support. Therefore the DALY concept includes age-weights, which indicate the relative importance of healthy life at different ages. The age weights used in the World Bank report rise from birth until age 25 and decline slowly thereafter (see figure 4.2).

Figure 4.2: The age-weight function

Age weighting is only applied in the individualist perspective. Age weighting is indicated in the Eco-indicator 99 calculations by the second number in DALYs (0,1), whereas calculations without age-weighting are referred to as DALYs (0,0).

Results of the Global Burden of Disease study
In the Global Burden of Disease study the WHO [MURRAY 1994] has calculated all DALYs for over 100 different causes in 1990 by region, sex and age. The leading causes of all DALYs (disability and premature death) world wide (1990) are presented in the table below.

	DALY (million years)	%
All causes	1362	100
Acute lower respiratory infections	114	8.4
Diarrhoeal diseases	100	7.3
Childhood cluster	68	5.0
Tuberculosis	46	3.4
Cerebrovascular disease	44	3.2
Ischaemic heart disease	42	3.1
Motor vehicle accidents	31	2.3
HIV infection	30	2.2
Sexually Transmitted diseases (excl. HIV)	22	1.6
Major affective disorder	19	1.4
Chronic obstructive lung disease	19	1.4

Table 4.4 Leading causes of DALYs in the world

4.4. Damage to Human Health caused by carcinogenic substances

Based on [HOFSTETTER 98]

4.4.1. Description of the problem

Providing evidence of a causal link between agent and tumour incidence is a complex task and needs the consideration of all kinds of experimental results and epidemiological studies. The International Agency for Research on Cancer (IARC) is one scientific body performing such qualitative risk assessments. IARC uses a classification system to group substances according to the evidence available on human or animal carcinogenicity.

Group	Description
1	The agent, mixture or exposure circumstance is carcinogenic to humans
2A	The agent, mixture or exposure circumstance is probably carcinogenic to humans (a positive association has been observed between exposure and human cancer for which a causal interpretation is credible. But change, bias or confounding could not be ruled out with reasonable confidence; there is also sufficient evidence of carcinogenicity in experimental animals)
2B	The agent, mixture or exposure circumstance is possibly carcinogenic to humans (there is sufficient evidence of carcinogenicity in experimental animals, but no adequate data on cancer in exposed humans)
3	The agent, mixture or exposure circumstance is not classifiable as to its carcinogenicity to humans (this group applies if no other category is used)
4	The agent, mixture or exposure circumstance is probably not carcinogenic to humans

Table 4.5: IARC classification of substances

For all substances classified in IARC groups 1, 2A, 2B and 3 for which sufficient information on physico-chemical characteristics and carcinogenesis is available, the damage to human health resulting from an emission is calculated. For group 1 there is epidemiological evidence that the substances are carcinogenic to humans. The other groups have proven or expected cancer potency in animals (see table 4.5).

4.4.2. Fate analysis

In order to create a consistent fate analysis for different damage categories within the Eco-indicator 99, the fate factors calculated by [HOFSTETTER 1998] are not used, but replaced by new fate factors calculated with EUSES. In chapter 3 the fate analysis is described in more detail.

The result of the fate analysis is a fate factor that provides the link between an emission in Europe (kg/yr) and the steady state concentration in air, drinking water and food resulting from this emission.

For the emission in the form of a mass load (1 kg) the resulting concentration can be allocated to the emission during a certain period of time (see also box 3.1).

4.4.3. Effect analysis: estimation of cancer incidence

The cancer incidence is estimated using the Unit-Risk concept:
The unit risk factor for inhalation is an estimate of the probability that an average individual will develop cancer when exposed to a pollution at an ambient concentration of one microgram per cubic meter for the individual's life (70 years) [UR in cases per µg/m3]
The Unit-Risk concept [WHO 1987] is used for estimation of the dose response relationship. Unit risk factors are mostly derived from IRIS (US EPA), WHO Europe and a study for Germany [LAI 1992]. In case more unit risk factors are available the most recent factors are preferred.

In case the unit risk factor for only one exposure pathway is known, the unit risk factors for the other pathways have been extrapolated by calculating an equal intake, using data on inhalation rate, consumption of drinking water and food. This procedure is common for US-EPA and the data are derived from risk assessment and can be considered worst case, meaning the extrapolated UR factors are probably overestimating the effects. The differences in bio-availability of substances for different uptake routes are not taken into account.

The population density determines the number of people exposed. Since pollutants with long atmospheric residence times are blown to rural areas, the effect of these substances is mostly on lower population densities. Long range distribution of substances is mostly via air, regardless where the emission takes place. Therefore the population density is modelled per substance dependent on their atmospheric residence times. In the calculation of substance specific residence time the following assumptions based on the average wind speed for large scale modelling are used:

- The population density equals the average population density of Western Europe (160 P/km2) if the residence time is one day (assuming full dispersion over Europe in one day).
- The population density equals the average population density of the World if residence time is one year (assuming full global dispersion).
- The maximum population density is set to 300 P/km^2

With the information on population density, unit risk and the fate factors, the cancer incidence, expressed as the number of cancer cases per kg substance emitted can be determined for the different pathways and emissions. This is named the effect factor.

4.4.4. Damage analysis

Estimation of the Years of Life Lost (YLL) and Years Lived Disabled per cancer incidence resulting from the effect factors are estimated from:

1. The type of cancer that is expected.
2. Which share of tumour patients will die.
3. How many potential life years are lost (depending on age).
4. How long is the illness.
5. What is the severity of the disability.

TYPE OF CANCER AND SURVIVAL RATE

It is assumed that the first mentioned site of tumour incidence is the most important one (shares are unknown). Only those sites distinguished in [Murray et al 1996] and registered in cancer statistics are considered. The category unspecified is modelled by the average over all cancer sites. The consequences for a wrong categorisation of cancer types are limited because of slight differences in results for the different cancer types.
The survival rate and the YLL are estimated from cancer statistics using the standard life table from Murray. The disability weights and the Years Lived Disabled for the cancer types are provided by work of [MURRAY ET AL 1996].

4.4.5. The role of cultural perspectives

Many of the steps in the calculation of the potential damage to Human Health caused by carcinogenesis of environmental pollutants could be changed by "value choices", due to differences in cultural perspective.

- Discounting future damages is not in accordance with LCA because no information on time is included in LCA at all. Therefore none of the cultural perspectives includes discounting.
- Individualists only include "proven damage", which means for carcinogenesis only IARC class 1 is included. Furthermore the Individualist will use age-weighting because they assign greater importance to being healthy at younger years. Another argument to include age-weighting for Individualists is the "human capital approach".
- Egalitarians include all IARC groups that may cause cancer (1,2,3) because they are risk aversive and want to prevent damage. They will not use age-weighting because they reject inequity among persons.
- Hierarchists include IARC groups 1 and 2, choosing a balance between probability and evidence, following the guidelines of governmental bodies or respected international organisations (WHO). By law all humans are equal thus no age-weighting is permitted.

4.4.6. Sources of uncertainty

The difference between the EI approach of "best estimate" and the use of risk analysis tools like EUSES and WHO unit risk causes a bias. These risk analysis tools, which are used to set standards, incorporate worst case scenario's, which means that the effects will be overestimated.

Uncertainty[11] in the estimation of cancer incidence at specific environmental concentrations in cases/kg emission [HOFSTETTER 1998] is caused by:

- Assumed non-threshold behaviour in the dose response relations.
- Modelling of the population density: uncertainty on the distribution of the population.
- The high variability between risk estimates from different sources for the same substance.
- Extrapolation of missing effect factors.

In the estimation of quantitative uncertainty estimates these aspects have been taken into account.

Not all Unit Risk factors used in the study have the same basis because different extrapolation methods are used by different sources. Part of the factors have been mathematically extrapolated by [HOFSTETTER 1998] assuming the inhalation intake causes the same effect as oral intake. Also no differences in bio-availability for oral intake and inhalation are considered in this extrapolation.

The overall uncertainty in the damage: DALYs caused by a certain emission in Europe, including all steps from fate analysis until the actual DALYs per kg emission, is quantified in [HOFSTETTER 1998]. The uncertainty expressed as σ_g^2 is an accumulation of the uncertainties in the separate steps.
The uncertainty is related to the perspective. In the Egalitarian perspective, a lot of very uncertain effects are included. The damage resulting from substances from the different substance groups has a higher uncertainty as the evidence becomes less strong.
However, the strongest influence on the overall uncertainty comes from the fate analysis, especially for the exposure through food. For DALYs from emissions to air caused through inhalation in substance group 1 the σ_g^2 varies from 5 to 8. For DALYs from emissions to water caused through food intake in substance group 1 the σ_g^2 varies from 22 to 236. This difference is mainly caused by the uncertainty in the fate factors.

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[11] Since so many assumptions and extrapolations have been made, the uncertainty in the effects is high. Some human health experts are opposed to attributing cancer to all types of substances emitted to the environment based on too many assumptions. However, this does not make the effect analysis less valuable for use in LCA. A decision on which classes of substances to include in the method is a subjective choice, which is influenced by cultural perspective.

4.5. Damage to Human Health caused by respiratory effects

Based entirely on [Hofstetter 98]

4.5.1 Description of the problem

In epidemiological studies it has been shown that several non organic substances and dust are related to respiratory effects in humans. In a recent literature review by [PILKINGTON ET AL1997] epidemiological data on respiratory effects from environmental pollution are summarised. According to Pilkington the following substances cause respiratory effects:

- Particulate matter PM_{10} and $PM_{2.5}$
- Nitrate and sulphate
- SO_3
- O_3
- CO, and probably
- NO_x

The primary emissions that cause exposure to these substances are PM_{10}, $PM_{2.5}$, TSP, NO_x, NH_3, CO, VOCs, and SO_x. These substances are considered as primary pollutants in the fate analysis.

4.5.2 Fate analysis

[HOFSTETTER 1998] calculated fate factors for particles with a simple model using assumptions on residence time and dilution height. These fate factors are compared to literature data. For all primary emissions the results of several statistical methods using empirical and modelled data (a.o. the European Monitoring and Evaluation Programme and ExternE) are compared.

Best estimates have been made for the fate factors based on three main principles:

1. The fate factors should be appropriate for European conditions.
2. Preference is given to fate factors for which the underlying assumptions are known.
3. Average fate factors that assume a proportional relation between emissions and concentration have been used for all but ozone creation. In this case the marginal factor was considered most appropriate in order to reflect the non-linear atmospheric chemistry of ozone formation.

THE UMBRELLA PRINCIPLE FOR INDIVIDUAL VOCS: POCP

One fate factor has been estimated [HOFSTETTER 1998] for total NMVOC, but the reactivity of single hydrocarbons varies widely and should be considered when the mixture of VOC emissions is known or if, e.g., one wants to support the decision on the choice between solvents.

A differentiation between single hydrocarbons can be undertaken by applying the concept of Photochemical Ozone Creation Potential (POCP). The POCP expresses the incremental ozone concentration per incremental emission for a specific VOC specie normalised by the ratio for ethylene. Ethylene serves as the reference substance and is one of the most reactive VOCs. The unit is without dimension and the POCPs are normally given as a percentage of the one for ethylene. Based on the most recent data on POCP and fate factors for organic substances reported by [JENKIN ET AL 1997] fate factors have been calculated by [HOFSTETTER 1998] for about 120 single VOCs. They are representative for climatic conditions typically found in North-western Europe.

4.5.3. Estimation of dose-response relations

For the damage analysis of emissions that cause respiratory effects the epidemiological approach is used.

Toxicological experiments do not produce effects at ambient concentrations, because test populations used are not representative and the substance is not equal to the substances in the environment. For this reason and the reason that the slope factor for the dose-response curve is not constant, toxicological data are difficult to extrapolate to effects at ambient concentrations.

On the other hand the epidemiological approach suffers from limited possibilities to prove causalities, to show correlation with several dozens of substances and to attribute health risks among pollutants with identical emissions sources. On the other hand the epidemiological approach can profit from a large body of literature and has already been used in several externality studies. Therefore this approach will be used to estimate DALYs for respiratory diseases caused by environmental pollutants.

Criteria for causation are used to assess the validity of dose-response relations derived from epidemiological studies. In selecting valid dose-response relations the following criteria apply:
1. The effects come after the exposure.
2. Biological plausibility (depending on the level of knowledge).
3. Consistency of results from large set of studies with similar objectives but different approaches
4. Results should be coherent.

The epidemiological information that is used by [HOFSTETTER 1998] is mostly derived from extensive and detailed reviews by [DONNAN ET AL 1997] and [PILKINGTON ET AL 1997]. The assumptions which are necessary to calculate exposure-effect slopes are taken from [PILKINGTON ET AL 1997].

- Particles: epidemiological evidence on adverse acute health effects of air polluted with particles is very substantial. There is strong but much less widespread epidemiological evidence on chronic health effects. Sulphates are assumed to result in very small particles ($PM_{2,5}$), whereas nitrates result in PM_{10}. A conversion factor is used to calculate the relationship for $PM_{2,5}$.
- Ozone: The overall evidence strongly supports the view that the acute effects of ozone can be quantified and that they should be added to those of particles.
- Sulphur dioxide: Some study results establish a clear association between SO2 and mortality, but causality is not proven yet.
- CO: there is little epidemiological evidence concerning CO. The relationship between CO and acute mortality is used in the calculations.
- Nitrous dioxide: a positive association between NO2 and mortality and respiratory hospital admissions is reported.

In the calculations it is assumed that the slopes are independent from ambient concentrations [PILKINGTON 1997]. The slope at the ambient concentration in the study region is taken as a proxy. However, such a linear relation is questionable when it is applied to vulnerable risk groups. It is the % increase in effect, which is found to be constant at all levels of concentration. In the study of [HOFSTETTER 1998] this simplified assumption is justified because the background concentrations of the study region and of the release assessed in the LCA may be similar and -more important- because the results are dominated by mortality which shows little dependency on the background concentration. Exposure-response functions are determined using the information on ambient concentrations, population density in the study area, daily hospital admissions for respiratory causes and the relative risk. The calculated exposure-response slopes have a background similar to the one of the Unit Risk factors for carcinogenicity (see section 4.4). However, no generally accepted list of exposure-response slopes for respiratory diseases has been published yet. The non-terminal endpoints are not yet defined sharply enough to allow for an international compilation [HOFSTETTER 1998]. This means the calculation of damage to human health from respiratory effects can be considered more experimental than the calculations for damage from carcinogenic substances.

4.5.4. Damage analysis

To calculate final results in DALYs the seriousness and duration of the diseases had to be estimated. The disability weights are estimated using the weights from Murray as a starting point, because no disability weights have been established for all endpoints. Also the duration of illness has been estimated. Literature values sometimes have a large variation. Poor data for life years lost due to premature death caused by respiratory effects had to be used.[HOFSTETTER 1998]

4.5.5. The role of cultural perspectives

Some of the steps in the calculation of the potential damage to Human Health caused by respiratory effects of environmental pollutants could be changed by "value choices", due to differences in cultural perspective.

- Discounting future damages is not in accordance with LCA because no information on time is included in LCA at all. Therefore non of the cultural perspectives includes discounting.
- Individualists only includes "proven damage". In case of respiratory effects the criteria for causation have been used to evaluate the validity of the evidence. Strong evidence, for the exposure-effect relations for fine particles, sulphates and ozone is supported by all groups, whereas weak evidence is only supported by Egalitarians.
- The Individualist will use age-weighting because they assign greater importance to being healthy at younger years. Another argument to include age-weighting for Individualists is the "human capital approach".

4.5.6. Sources of uncertainty

For fate factors for substances that cause respiratory effects, geometric standard deviations have been estimated from the variations between literature sources for the fate factors. The σ_g^2 for these fate factors varies between 2 and 4. [HOFSTETTER 1998]

Since there is no generally accepted list of UR available, the list of ER-slopes for respiratory diseases is a compilation derived mostly from [ExternE 1997] and is less strong than the internationally accepted list of UR for carcinogenesis.
Disability weights for morbidity due to respiratory illnesses and the duration of morbidity have been estimated and are not validated. The resulting uncertainty in the DALYs for respiratory diseases is high. The overall uncertainty depends mostly on the uncertainty in the effects, and in the case of respiratory effects, uncertainty in the fate analysis is less important compared to the uncertainty in the effects. Also for respiratory effects the uncertainty is related to the perspective. The Egalitarian perspective includes more uncertain effects. The overall uncertainty for this perspective ranges from a σ_g^2 of 16 to a σ_g^2 of 64. For Hierarchists and Individualists the σ_g^2 varies between 16 and 36.

4.6. Damage to Human Health caused by climate change

4.6.1. Description of the problem

There are a few particular difficulties in the modelling of health effects from the greenhouse effect:

1. Climate change does not create much direct damage at present, but the predictions are that the current emissions will create considerable damages in the coming decades and thereafter. This means we have to resort to scenarios and models that could not be validated by experimental data.
2. The vulnerability of systems at risk is dependent on the development of the economy and society, as some effects can in principle be averted, if proper care is taken.
3. Temperature change has many important positive health effects, next to the negative effects.
4. The greenhouse emissions in Europe will cause damage all over the world.

Due to these special characteristics there is wide disagreement about the consequences of the greenhouse effect. We are confronted with an effect that combines possibly very high damages with very high uncertainties whether the damages will ever occur or not.

Damage to Human Health can occur via several impact pathways to multiple endpoints. Table 4.6 gives an overview and indicates which impact pathways can be modelled quantitatively given the present knowledge.

Mediating Process	**Health outcomes**	**Change in mean temperature and weather**	**Extreme events**	**Rate of change of climate variable**	**day-night difference**
Direct					
Exposure to thermal extremes	Altered rates of heat- and cold-related illnesses and death		+++		+
Altered frequency and/or intensity of other extreme weather events	Death, injuries, psychological disorders, damage to public health infrastructure		+++		
Indirect					
Effects on range and activity of vectors and infective parasites	Changes in geographic ranges and incidence of vector-borne diseases	+++	++	+	++
Altered local ecology of water-borne and foodborne infective agents	Changes incidence of diarrhoea and other infectious diseases	+	+		
Altered food (especially crop) productivity, due to changes in climate, weather events, and associated pests and diseases	Malnutrition and hunger, and consequent impairment of child growth and development	++	+	++	
Sea-level rise, with population displacement and damage to infrastructure	Increased risk of infectious disease, psychological disorders	++	++	+	
Levels and biological impacts of air pollution (A), including pollens and spores (P)	Asthma and allergic disorders; other acute and chronic respiratory disorders and death	A:+ P:++	P:++		P:+
Social economic and demographic dislocations due to effects on economy, infrastructure, and resource supply	Wide change of public health consequences: mental health and nutritional impairment, infectious diseases, civil strife	++	+	+	

Table 4.6: Possible major types of impact and probable relative magnitude of impacts of climate change on Human Health (after [MCMICHAEL ET AL 1996]) (+++=great effect; +=small effect; empty cells indicate no known relationships) The grey fields are actually modelled here.

We cannot model all effects, as many effects are too uncertain to quantify. Therefore in the Eco-indicator 99 we could only model the grey areas.

Box 4.2 Compilation of study results/effect analysis

Most studies look at the damages of an equilibrium climate change associated with a doubling of the pre-industrial CO_2-equivalent concentration of all greenhouse gases. Some of these results are shortly reviewed in the table below and the following text. The table and citations are taken from [PEARCE ET AL 1996]:

	Cline (1992a)	Fankhauser (1995)(a)	Titus (1992)	Tol (1995)(b)
Human life	5.8	11.4	9.4	37.4
Migration	0.5	0.6	-	1
Water pollution			32.6	
Tropospheric ozone	3.5	7.3	27.2	
Total HH	**9.8**	**19.3**	**69.2**	**38.4**
Total all risks	**61.1**	**69.5**	**139.2**	**74.2**

Table 4.7 Monetarised $2xCO_2$ damage to present US-economy, base year 1990, billion $/a [PEARCE *ET AL.* 1996]

(a) Fankhauser (1995) estimates at 2.5° warming ($2xCO_2$) a world-wide increase of 137'700 death per year due to health effects and an additional of 8'000 by hurricanes.

(b) Tol (1995) estimates at a doubling of CO_2 an additional 215'000 death per year world-wide.

Next to the studies cited in Pearce et al, we have analysed a number of other (more recent) studies:

[MOORE 1998] reports health damages due to thermal stress and amenity effects of global warming. He presents results only for benchmark level damages in the United States of America. For a warming of 4.5° in USA (CO_2-doubling) he finds a reduction of 40'000 death per year, hospital savings of about 19 to 22 billion $ and a willingness to pay of about 30 to 100 billion $, both in 1994 $ and annually. Main reasons for the negative damage is that he assumes that the winters and nights will become much more warmer than the already hot summer days do. The morbidity costs are based on crude hospital statistics, which are not very convincing to us. The willingness to pay looks at differences in wage-rates depending on the climate. Qualitatively, Moore argues that people like to live in a warm climate with large seasonal changes – and, of course – he refers to the retired Americans that move to the South (similar in Europe with Greece and Spain).

These findings are fully supported by [MARTENS 1998] and The Eurowinter Group [EWG 1997], although the latter was only looking at the adverse effects of cooling. [MARTENS 1998] estimates based on his meta-review that the cardiovascular mortality rate decreases by 50/100'000 for an increase in winter temperature while the summer increase with rising temperature is only 3/100'000. Although he mentions that for respiratory effects the results are more difficult to express and that heat wave-related mortality is not specially looked at because his review focused on long-term influences he concludes that global warming may well cause a decrease in mortality rates.

[KALKSTEIN AND GREENE 1997, KALKSTEIN AND SMOYER 1993] look to heat-waves and draw a completely different picture based on data for 44 U.S. cities. They show with three different climate models and the statistical finding that the mortality rises more steeply for temperature increases in summer than decreases in winter. This means that a careful assessment has to separate effects from average temperature change and from heat/cold waves.

[TOL 1999B] presents his latest results including the new data for thermal stress. He finds for malaria (66'000/°C), Dengue (7'400/°C), cardiovascular effects from heat-stress (94'400/°C) and heat-related respiratory effects (255'500/°C) increased mortality for a rise in temperature but for Schistosomiasis (-900/°C) and cold-related cardiovascular effects (-479'100/°C) a decrease. For the vector-borne diseases this results in a net increase of 72'500 deaths per °C and for thermal stress a decrease of 129'200 deaths per °C. This would equal a net benefit from global warming!

4.6.2. Marginal damages

The quoted studies refer to the total damage of climate change. However, we are only interested in the marginal damage; that is, we would like to know the increase in the damage per ton of CO2. Unfortunately there are only a few studies addressing this issue. The most important project in this field is the ExternE project. This project aims at calculating the societal, or external costs of energy production systems in Europe, The project used two models to make the appropriate assessments, the FUND 1.6 and the Open Framework Model. The Open Framework Model, does not address the damage to Human Health very explicitly. The FUND model is therefore the most appropriate to use here. Basically the FUND model is a benchmarking model, that, like we have seen before, calculates the total damage at a doubling of the CO2 concentration. It models: dryland loss, wetland loss, coastal protection, migration, agriculture, heat stress, cold stress, malaria, tropical cyclone, extratropical storms, river floods and unmanaged ecosystems in nine regions of the world. For the ExternE project, it was adapted to calculate marginal damages.

After experiencing many difficulties in trying to interpret this study in such a way that the result could be expressed as DALYs per ton of greenhouse gases, Tol was willing to make a number of special calculations for this project. Tol used the latest version of the FUND model, quoted above (version 2.0). The results have been presented to Hofstetter in the form of excel tables and some personal communication[12] [TOL 1999B]

The procedure can be summarised as follows:
1. Tol made a model run of the so-called IPCC IS92a scenario, and calculated damages for each year between 2000 and 2200.
2. Tol repeated this process three times and added a flow of 1Mt per year of CO_2, CH_4 and N_2O respectively.
3. The differences between these runs were interpreted as the marginal damage

The results include change in deaths due to malaria, schistosomiasis, dengue fever, cardiovascular and respiratory disorders, all due to changes of the average temperature. Next to this, the number of people which have to be displaced due to sea level rise were calculated.

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[12] Clearly there is a risk in using data from a new software model that has not been published in a reviewed article, but the alternative, interpreting a study that was written for other purposes turned out to be at least as risky.

All these parameters are calculated for nine world regions. Unlike the other impact categories, the greenhouse gases emitted in Europe contribute to damages all over the world.

The data used on health effects in the Fund model is to a large degree coming from Martens and Kalkstein et al, see above, however Tol made a number of changes. He assumes that people with an income higher than $ 3100 do not get Malaria, Dengue and Schistosomiasis, as these people can afford prevention. Furthermore he excludes information of Kalkstein on the increase of extreme hot days.

4.6.3. Dealing with negative damages

An interesting result of the calculations is, that there are also negative damages to Human Health. The calculations show that the number of cold–related cases of cardiovascular disease decreases considerably if the average temperature increases. As a result the decrease of cold related cardiovascular diseases is up to a factor 5 bigger than the increase of heat related cardiovascular diseases. However, as cardiovascular diseases contribute relatively little to the total health damage the overall effect is not so significant.

However, this result prompts a number of fundamental questions:

- In general the negative damages are left out in LCA impact assessment. The question is, if it is justified to follow this general principle here?.
- What are the ethical consequences of allowing negative effects to be compensated by positive effects in the case of Human Health. Is it justified to tell people that suffer from malaria in Africa, or that are displaced from Polynesia that their problem is outweighed by health improvements in Finland, and that in general the world is happy with the climate change?

The following compromise has been made:

1. The positive and negative damages within a world region are allowed to compensate each other. For instance a damage in Italy, can be compensated by a negative damage in Denmark.
2. The positive and negative damages are not allowed to compensate each other between the 9 world regions.

This choice has also a pragmatic background, as it is difficult to separate positive and negative effects within a region, due to some modelling restrictions.

4.6.4. Damage modelling

The calculation results make reference to no less than 6 endpoints for which DALYs have to be calculated. Tol also calculated DALYs but these have not been used, because the calculation procedure seems different from our standard approach.

4.6.5. Extrapolation to other greenhouse gases

The extrapolation of the damages from the three gases to the full range of greenhouse gasses can be done with the umbrella principle [HOFSTETTER 1998], also used in the case photochemical ozone creation. The most recent list of equivalency potentials is provided by [SCHIMEL ET AL 1996]. The list is developed for three time perspectives. The perspective for 100 years fits best for the calculations of Tol.

However, the lifetime of the gases is not only responsible for the magnitude of the radiative forcing, but also on the question whether health benefits or damages occur. Substances with a short lifetime appear to have greater benefits than substances with a longer lifetime.

According to Tol [personal communication] the equivalency factors can lead to misleading results. In fact all gases need to be treated separately through the FUND model. As this was not feasible within the budget of this project the following solution has been found:

- Gases with a lifetime below 20 years behave like methane in the damage model
- Gases with a lifetime between 20 and 110 years behave like CO2 in the damage model
- Gases with a lifetime above 110 years behave like N2O in the damage model

The damage of greenhouse gas i can now be calculated as:

$GWP_i * D_{refsub}/GWP_{refsub}$

In which D_{refsub} represents the damage factor for either CO2(as C), CH_4 or N_2O presented in table 4.8, and GWP_{refsub} represents the global warming potential for one of the three reference substances.

4.6.6. The role of cultural perspectives

The following choices are made for the three perspectives:

- Individualists use age weighting in the calculation of DALYs, and they use the short time perspective, which is set at 100 years.
- Hierarchists do not use age weighting, but would disregard the effect of displacements. They would argue that displacements are not a big problem if it is well organised.
- Egalitarians also do not use age weighting, but they do include the effects of displacements.

With these choices it is possible to calculate both the positive and negative damages (table 4.8).

		Negative damages			**Positive damages**			**Negative damage as percentage of positive**		
		Total E-attitude	Total H-attitude	Total I-attitude	Total E-attitude	Total H-attitude	Total I-attitude	E	H	I
		DALY$_{(0,0)}$	DALY$_{(0,0)}$	DALY$_{(0,1)}$	DALY$_{(0,0)}$	DALY$_{(0,0)}$	DALY$_{(0,1)}$			
CO2 up to 2100	per tC			-1.8E-5			7.4E-4			2.4%
CO2 up to 2200	per tC	-2.5E-5	-2.5E-5		7.5E-4	7.5E-4		3.3%	3.3%	
CH4 up to 2100	per tCH4			-1.3E-4			4.4E-3			3.0%
CH4 up to 2200	per tCH4	-2.0E-4	-1.9E-4		4.4E-3	4.4E-3		4.5%	4.3%	
N2O up to 2100	per tN2O			-1.5E-3			6.7E-2			2.2%
N2O up to 2200	per tN2O	-2.1E-3	-2.1E-3		6.9E-2	6.9E-2		3.0%	3.0%	

Table 4.8: Positive and negative human health damages per ton emitted substance, expressed in DALYs, for three cultural perspectives. The last three columns show the relative importance of the negative damages, which have not been incorporated

4.6.7. Sources of uncertainty

The overall uncertainties for the damage factors are represented by a σ^2_g value that ranges from between 1 for CO_2 to 10. These are only the technical uncertainties. As stated in the beginning of this section, there are many conceptual uncertainties.

The following potential health effects are not considered in this assessment (see table 4.6):

- Other diseases than vascular diseases due to heat waves; death, injuries, psychological disorders, and damage to public health infrastructure due to extreme weather events;
- Other vector-borne diseases than malaria, schistosomiasis, dengue; infectious diseases[13];
- Effects of malnutrition and hunger;
- Increased impact of pollutants at higher temperatures,
- Civil strife etc.

All these direct and indirect consequences may have significant effects on the damage assessment. It is therefore not clear whether the quantified effects are the most important ones!

Next to these data gaps [WATSON ET AL 1998:7] remind that "quantifying the projected health impacts is difficult because the extent of climate-induced health disorders depends on other factors – such as migration, provision of clean urban environments, improved nutrition, increased availability of potable water, improvements in sanitation, the extent of disease vector-control measures, changes in resistance of vector organisms to insecticides, and more widespread availability of health care."

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[13] An illustrative example to indicate the importance of such omissions: The serious African horse sickness (mortality rate 95% in susceptible animals) appears to be directly related to the El Niño events (Nature, vol. 397, 18 February 1999, p.547). 13 of the 14 major epidemics in the horse sickness since 1803 coincided with the El Niño events.

4.7. Damage to Human Health caused by ionising radiation

4.7.1. Description of the problem

This paragraph describes the damage to Human Health related to the routine releases of radioactive material to the environment. It is a summary of a paper that has been written for our project by Frischknecht, Suter, Hofstetter and Braunschweig [FRISCHKNECHT ET AL 1999] Some parts of this paragraph have been quoted directly.

Box 4.3. Ionising radiation in LCA

In the nuclear fuel cycle, in phosphate rock extraction, in coal power plants but also in oil and gas extraction, air- and waterborne radionuclides are released to the environment. Up to now, such emissions have rarely been considered in Life Cycle Assessment due to a lack of appropriate operational Impact Assessment models. However, ionising radiation has been mentioned in LCA literature for some time and in 1994 and 1996 the extensive energy systems database "Ökoinventare von Energiesystemen" [ESU 1994/1996] published emission data for a large number of radionuclides emitted within the nuclear fuel cycle and by coal power plants. In 1995, the first ExternE reports included a detailed assessment of the health effects of nuclear facilities in France [DREICER et al. 1995]. Based on these Frischknecht (1998:129ff.) created an impact category for radioactive releases compatible with the Eco-indicator 95 and the CML impact oriented characterisation method [HEIJUNGS et al. 1992a&b].

Health effects due to possibly large accidental releases are not considered because they escape the present methodological framework of LCA. Health effects due to occupational exposure are not considered.

Table 4.9 gives an overview of the entire assessment method of health effects of ionising radiation introduced in this paragraph. The model starts with the release at the point of emission, expressed as Becquerel (Bq). One Becquerel is equivalent with one decay per second.

Phase of the Model	***Stage of Pathway***	***Units***
Inventory Analysis	Radioactive releases	Becquerel, Bq; Bq/FU [1])
↓	↓	
Fate Analysis	*Transport, dispersion and deposition* Contamination in environment	Bq/kg, Bq/l, Bq/m^2, Bq/m^3
↓	↓	
Exposure Analysis	Standard characteristics of people Inhalation, consumption of food and water Absorbed Dose Effective and Average Individual Dose Collective Dose	m^3, kg, l gray, 1Gy=1J/kg Sievert, Sv man.Sievert, man.Sv
↓	↓	
Effect Analysis	Dose response relationship Fatal, non-fatal cancer, severe hereditary effects	Number of cases/man.Sv
↓	↓	
Damage Analysis	Disability weighting scale Disability adjusted life years (DALYs)	YLD, YLL, DALYs/fatal cancer
↓	↓	
Damage Assessment and Cultural Theory	Value-laden assumptions value weighted DALYs	DALYs/kBq

Table 4.9: Overview of impact pathway stages of radioactive releases in the assessment described in this paper, based on [DREICER ET AL 1995:19] and [HOFSTETTER 1998]. [1]) FU: Functional unit

4.7.2. Fate analysis

The fate model has been based on [DREICER ET AL 1995], who described the routine[14] atmospheric and liquid discharges in the French nuclear fuel cycle.

Box 4.4. Fate and dispersion for radionuclides

Data of discharges from the sites (mining and milling, conversion, enrichment, fuel fabrication, electricity production, and reprocessing) and of the surrounding conditions (population density, lifestyles of that population, meteorology, etc.) refer to the French situation. The models use a time horizon of 100'000 years in order to consider significant impacts of the different pathways[15]. For the assessment of long-term global impacts the world population is assumed to remain at a constant 10^{10} people for 100'000 years.

For dispersion of atmospheric discharges a Gaussian plume model is used. For liquid releases into rivers a simple box model is used assuming instantaneous mixing in each section and representing the radionuclide concentration in a compartment with a differential equation.

For globally dispersed radionuclides, i.e., Tritium, Carbon-14, Krypton-85, and Iodine-129, simplified models over a time horizon of 100'000 years are applied. For H-3 the global hydrological cycle is modelled dynamically based on seven compartments. For C-14 four environmental compartments are used in a dynamic model. For Kr-85 a dynamic model with two compartments (for the two hemispheres) is used. For I-129 a dynamic model with nine compartments is applied.

The confidence in the results of the global assessments for Carbon-14, Tritium, Iodine-129, and Krypton-85 is low "due to the extremely general models that are used and the propagation of very small doses over a large population for very long periods of time" [DREICER ET AL 1995:310].

The uncertainty in the fate analysis is approximately a factor 2 to 4. For the global assessment the uncertainty is probably greater than an order of magnitude, except for Carbon-14. As we will see the latter uncertainty is disturbingly high, as the global effects turn out to be very significant.

4.7.3. Exposure analysis

In the exposure analysis we calculate what dose human actually absorb, given the radiation levels that are calculated in the fate analysis. The measure for the effective dose is the Sievert (Sv), based on human body equivalence factors for the different ionising radiation types (α-, β-, γ-radiation, neutrons). 1 Sv = 1 J/kg body weight.

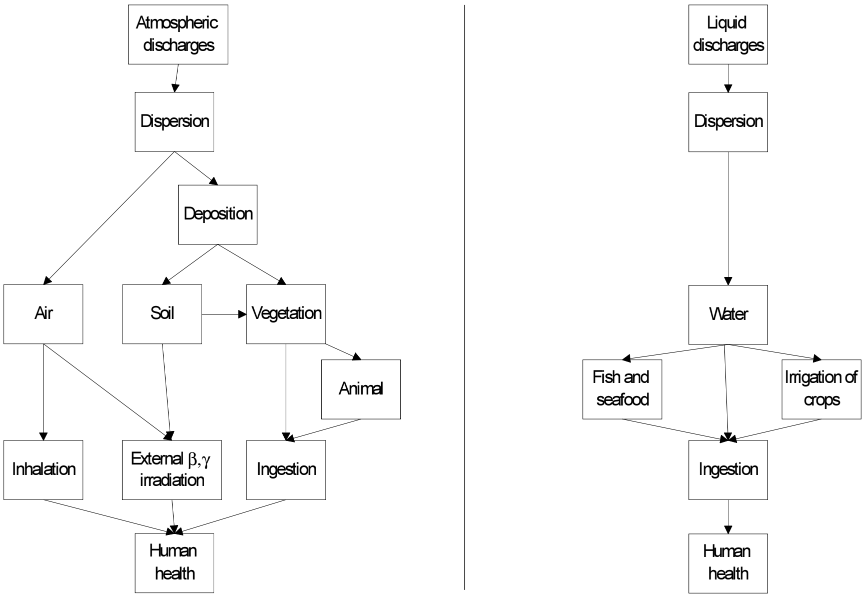

Figure 4.3: The exposure pathways that are taken into account. Also here a difference is made between exposure from atmospheric and liquid releases through the local/regional fate modelling and global exposure.

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[14] Routine emissions: Emissions due to normal operation excluding the low probability of severe accidents.

[15] With half-lifes of $1.6 \cdot 10^7$ years for Iodine-129 or $7.1 \cdot 10^8$ for Uranium-235 additional impacts are to be expected beyond 100'000 years. According to [DREICER 1995:52] only about 15% of the collective effective dose of Iodine-129 occur during the first 100'000 years compared to an assessment until infinity.

Data expressed in Sievert contain physical data on energy doses and biological data on the sensitivities of different body tissues. An intermediate stage in the calculations of doses is often expressed as Gray (Gy). This is the measure of absorbed dose without considering the different reaction types of body tissues. In order to link the emissions (Bq) to immissions (Sv), we need to draw up the exposure routes.

The global exposure of Tritium, Carbon-14, Krypton-85, and Iodine-129 have been calculated for a time horizon of 100.000 and 100 years. The uncertainties for the global exposure is considerable. A σ_g^2 of 10 to 50 has to be assumed.

4.7.4. Effect and damage analysis

In the damage analysis we concentrate on carcinogenic and hereditary effects, as these appear to be the most significant [DOBRIS 1996].
Two issues are important:

- Establishing the number of cases that occur as a result of the calculated exposure.
- Establishing the number of DALYs per case.

An important discussion is whether and how epidemiological findings at medium and high exposure may be extrapolated to low doses[16]. Linear, supralinear, sublinear, threshold models and even beneficial effects of low radiation levels thanks to a hormetic[17] effect have been suggested.

Most international advisory boards assume a linear no threshold (LNT) behaviour for low doses of ionising radiation. The slope including high dose-rates can be best described as S-shaped and the section where no acute effects are observed is supposed to follow a linear-quadratic function.
Most of the epidemiological information is available from the quadratic intersection.

Box 4.5. Extrapolation of low dose damages

In order to correct for the slope for low doses a so-called 'dose and dose-rate effectiveness factor' (DDREF) is used which was found to be between less than 2 and 10 [ICRP 1999]. for example, A DDREF of 5 means that the risk increase per man.Sv observed at high doses is divided by 5 to assess risks at low doses. All higher DDREF stem from animal tests. Epidemiological data on the association between exposure doses and cancer cases are available from a still ongoing study with the survivors of the atomic bomb attacks in Hiroshima and Nagasaki. This study includes survivors with a large range of exposure up to low doses as well. A dose and dose-rate effectiveness factor of 2 is the best estimate for the extrapolation to low doses although the ICRP "recognises that the choice of this value is somewhat arbitrary and may be conservative" [ICRP 1999:19]. This factor corrects for the epidemiological and toxicological findings that effects are lower at lower dose-rates.

[FRISCHKNECHT ET AL 1999] list the DALYs for the same types of cancers we have used in previous calculations. In total they found 0.5 fatal and 0.12 non-fatal cases per Man.Sv. The σ_g^2 is a value of 3. [FRISCHKNECHT ET AL 1999] also assume that the radiation induced cancer cases occur at the same age pattern as for other cancer causes. Frischknecht et al also present the cases with and without age weighting. The method of calculation of the number of DALY per cancer case is identical to the one used for carcinogenic effects.

The number of severe hereditary effects is assumed to be 0.01 case per Man.Sv [ICRP 1999] This number is very uncertain (σ_g^2=5), as it was derived from animal tests.
For hereditary effects it is much less clear how one case should be expressed in DALYs. Frischknecht et al quote [DREICER 1995], who assumes severe hereditary effects either result in immediate death or severely impaired life. [MURRAY ET AL 1996] suggest disability weights of about 0.2 to 0.6 for serious disabilities, including genetic defects. Frischknecht et al assume that 50% of the cases result in immediate death, while the rest lives with a disability weighted as an average of 0.4. This results in 57 DALY per case with age weighting and 61 DALYs per case without age weighting.

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[16] Low doses are equivalent doses resulting from absorbed doses below 0.2 Gray (ICRP 1999:19).

[17] Hormetic effects are effects stimulating the immune system.

4.7.5. The role of cultural perspectives

For the calculation of the whole cause and effect chain, we run into a number of value-laden choices, which are dealt with using the cultural perspectives. Frischknecht et al mention the following aspects:

1. The time horizon for the integration of exposure to people.
2. The area to be considered in the fate and exposure analysis.
3. The necessary evidence for an association between low-level radiation and cancer cases.
4. The extrapolation model to be used for estimating health effects at very low doses.
5. The dose and dose-rate effectiveness factor that should be applied if linear no-threshold extrapolation methods are used.
6. The assumptions in the concept of disability adjusted life years (DALYs).

Frischknecht at al argue that the decades long debate has resulted in sufficient agreement on fate and exposure analysis, the necessary evidence, and the dose-rate effectiveness.

The remaining disagreement may therefore be seen as technical uncertainties. The choices on the time horizon and the choices within the DALY system, especially the age weighting are dealt with through cultural perspectives:

- The *egalitarian* and the *hierarchist* perspectives use the longest time horizon (100,000 years) and do not use age weighting
- The *individualist* perspective integrates the exposure over 100 years, and applies age weighting.

With these choices Frischknecht et al calculate the DALYs per emission (Bq) for 31 nuclides, which are supposed to be the most important in nuclear power plant operations.

4.7.6. Sources of uncertainty

The uncertainties are already described above.

4.8. Damage to Human Health caused by ozone layer depletion

4.8.1 Description of the problem

Stratospheric ozone levels are near their lowest point since measurements began in 1970. The most important reason is the increase of the chlorine and bromine levels, due to the release of substances such as CFCs with a long atmospheric residence time. This low level has resulted in increased UV radiation levels:

- about 4 to 7 % at mid latitudes in the Northern and Southern hemisphere
- about 130% in the Antarctic Spring
- about 22% in the Arctic Spring

Since the Montreal Protocol and the Copenhagen and London Amendments have been accepted by many industrial and developing countries, the production and release of chlorine and bromine compounds with a long atmospheric residence time such as CFCs have been significantly reduced, and are still decreasing. Only in some, mainly developing countries, production and releases are continuing. Figure 4.4 shows the sharp reduction in ODP (ozone depletion potentials) weighted releases from the most important producers of ozone-depleting substances the AFEAS [www.afeas.com]. According to AFEAS their industries represent about 40% of global CFC production and 90% of HCFCs. The remaining 60% of CFC production takes place in China, Russia and Argentina.

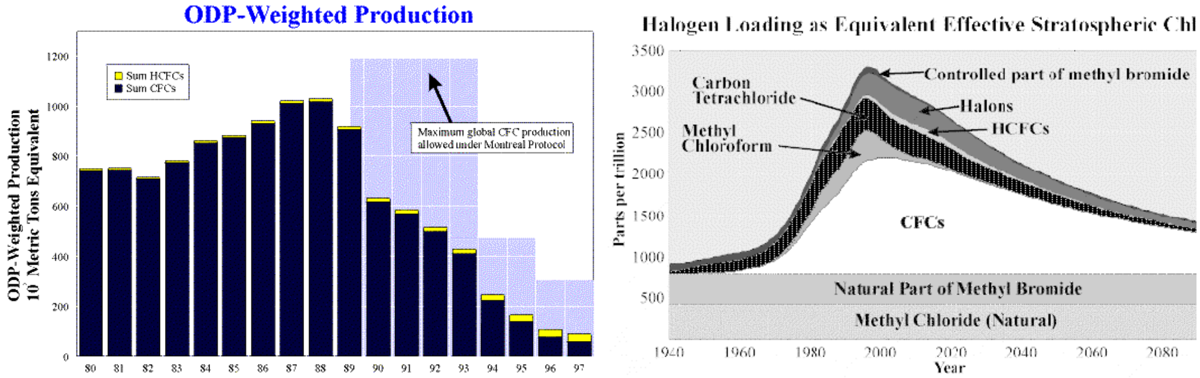

Figure 4.4: Decrease of ozone depleting substances (left) and contribution of substances to the stratospheric chlorine concentration.

As the figure shows, the protection of the ozone layer is clearly a good example of how international co-operation averted a potentially big problem. Without the Montreal Protocol the number of excess death because of skin cancer would have increased rapidly to 10 or more deaths per 100.000 inhabitants in mid-latitude areas, this would have made the ozone layer depletion problem by far the most important environmental problem [UNEP 1998].
With the apparent success of the abatement, one could argue that ozone layer depletion is no longer an issue. However, this is not true; any emission of CFCs or to a lesser extent HCFCs is still contributing to the damages to human health.

The modelling of damages has some specific difficulties:

- Most studies are analysing the effect of the Montreal, Copenhagen and London Protocol. No studies have been found analysing the marginal effect of releasing an additional kg of CFCs.
- Knowledge on a number of endpoints is not sufficient. It is unclear to what extent the ozone layer depletion contributes to the damage to the human immune system and the damage to ecosystems.
- It is relatively easy for humans to avoid exposure to higher UV levels by behavioural changes.
- The UV increases are dependent on the latitude. In the tropics the increases are negligible, near the poles they are at a maximum.
- Only fair skinned persons are sensitive for skin damages due to increased UV levels, while cataract can occur with every skin type.

4.8.2. Fate and exposure model

Chlorine containing substances are diluted in the troposphere. In an average of 4 years they also drift into the stratosphere, where they contribute to chemical processes that result in the depletion of the ozone layer. Clearly the atmospheric residence time (which spans a range of about 1 to 1000 years) is an important factor. Substances that have a significantly lower residence time than 4 years do not reach the stratosphere in substantial amounts. This means the damage created by a substance is depending on the time horizon. If the time horizon is just 100 years, part of the damage created by substances with a residence time of more than 100 year will be neglected. (see also under “Equivalency factors”).

No real fate factors or fate models have been found in this research. However, [SLAPER ET AL 1992] present some useful data for CFC11. Unfortunately they do not provide a fate factor. However, for CFC11 Slaper presents two graphs, displaying the CFC11 production rates under the London amendments, and the expected CFC11 concentration for this scenario. As a temporary solution the fate for CFC11 will be calculated. For other substances equivalency factors will be used.

As a first estimate the surface area under these graphs was assessed. The left-hand picture of figure ... displays the flow (Mkg/yr) as a function of time. The surface under these graphs is a mass. For the London amendments, the expected cumulative emission is 9.8 Megatonnes. In the right hand picture the expected concentration is plotted. The surface under this curve has the dimension of concentration times years. The London amendments will result an average concentration of 27.5 ppb_v * yr.

Apparently the relation between an emission and a change in concentration for CFC11 is 2.8 ppb_v*yr/Mton.

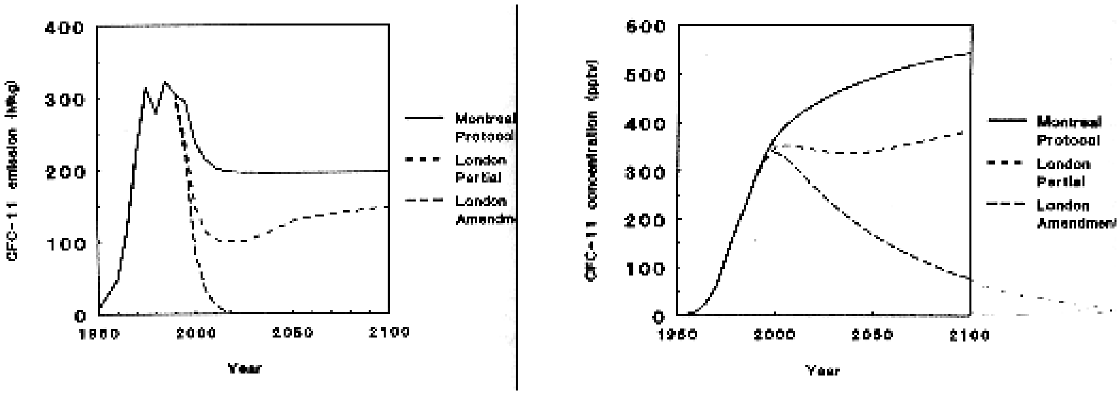

Figure 4.5: Relation between the predicted emission pulse according to the London protocol and the predicted CFC concentration. The fate factor can be derived by comparing the surface areas under the curves for the London Amendment protocol.

The concentration of CFC11 is not equivalent with the ultimate concentration of chlorine, as each CFC molecule has three chlorine atoms. Per ppb CFC, there will be 3 ppbs chlorine.
Finally, the relation between chlorine concentration and ozone depletion must be established. This relation is dependent on the geographic latitude.

$$\Delta O(\phi,t) = k(\phi) * (Cl(t) - Cl(0))$$

With delta O representing the annually averaged change in Ozone column at the 10° latitude band at time t [%/year]. Cl(0) is the chlorine concentration at which no damage to the ozone layer would occur. This is estimated to be 1.9 ppb [Martens 1997]. As we are interested in the marginal effect, this threshold is irrelevant; only the factor k is important. Martens gives four values for k (expressed as % per ppb). These figures are based on the observed Ozone trend by the TOMS (total Ozone Mapping Spectrometer) data. The uncertainty in k is (twice the standard deviation) is 1,3%

Latitude	**Measured ozone trend (TOMS)**	**k factor**
55° North	-0.35%/yr	-3.2%/ppb
15° South	-0.02%/yr	-0.2%/ppb
25° South	-0.14%/yr	-1.0%/ppb
35° South	-0.29%/yr	-2.6%/ppb

Table 4.10: Some values for the k factor for different latitudes (source:[MARTENS 1998])

From this data it is clear that the ozone depletion will occur mostly at the higher latitudes. [SLAPER ET AL 1992] gives higher k factors (4,9% per ppb at 50° N and 4.3%/ppb at 40°N) but this is based on older measurements from the TOMS system.
We have not been able to perform a population density weighted average calculation, but it is clear that most people live between 30° South and 55° North. As an average value we take a value for k as 2%/ppb, with a σ^2_g= 2

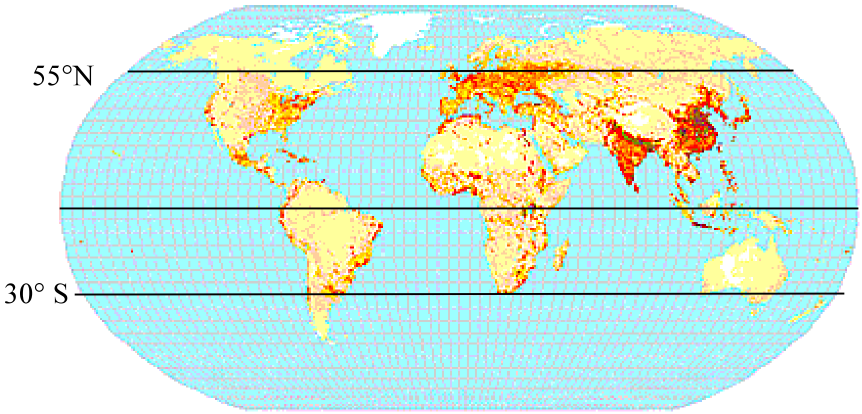

Figure 4.6: Population density distribution over the world. (source www.ESRI.com)

The relation between the release of a mass load M_{CFC11} and a temporal decrease of ozone layer thickness can be calculated, with:

$$\Delta O = f * n * k \ \ * M_{CFC11}$$

In which n =3 (the number of chlorine atoms), f = the fate factor (= $2.8E^{-9}$ ppb*yr/kg), and k = the relation between chlorine concentration and ozone depletion (= 2%)
With this simple model, it is possible to establish the relation between a CFC11 emission and the change on ozone layer thickness, which is directly proportional with an increase in UV.

In most analysis, so called ozone depletion factors are used, that express the relative harmfulness compared to CFC11. These factors are based on a time frame of 500 years or more. For the individualist perspective, we are also interested in the damage when the timeframe is only 100 years. [SOLOMON AND ALBRITTON 1992] calculate the polar ozone depletion potentials for some substances and a function of the time horizon. This calculation shows that the difference between a time horizon of 100 and 500 years time perspective is not very big. The ODP value for a long living species, such as CFC113 is 28% lower, while the ODP for HCFCs can be approximately 50% higher. As Solomon makes her calculation only for a few substances, and as her calculations are only valid for polar regions, it is difficult to use this data. For the time being this difference will be ignored and we will use the standard ODP values.

4.8.3. Effect analysis

UV radiation can cause both beneficial and adverse effects on humans. A direct beneficial effect of exposure is the formation of vitamin D. Adverse effects are among others: sunburn, “ageing” of the skin, and snow blindness. Health risks associated with ozone depletion are increased damage to skin, eyes, and immune system [UNEP 1994].
In light-skinned populations, exposure to solar UVR appears to be the most important environmental risk factor for skin cancer (basal and squamous cell carcinomas and cutaneous melanoma). From experimental data and epidemiology, it can be inferred that chronic accumulation of UV exposures is important throughout the development of SCC [AUTIER AND DORÉ, 1998][UNEP 1998]. In the cases of both basal cell carcinoma (BCC) and melanoma (MSC), increases in risk are tied to early exposures (before about age 15), particularly those leading to severe sunburns. There is reasonably good evidence that such immuno-suppression plays a role in human carcinogenesis. However, the implications of such immuno-suppression for human infectious diseases are still unknown [UNEP 1998].
Ocular damage from UV exposures includes chronic eye conditions like cataract [UNEP 1998]. Cataracts may be a more widespread health effect than skin cancers, because all populations are affected [UNEP 1994].

Quantitative risk estimates are available for some of the UV-B-associated effects, e.g., cataract and skin cancer; however the data are insufficient to develop similar estimates for immuno-suppression.

The impact of increases in ambient UV-B on these diseases has been quantified in terms of the biological amplification factor (BAF): the percentage increase in incidence that would result from a 1% increase in ambient UV radiation. The other step in calculating overall increase in incidence per percent ozone depletion is represented by the radiation amplification factor, RAF: the percentage increase in effective UV per percent decrease in ozone. The overall percentage increased incidence per percentage ozone depletion is then represented by the amplification factor: AF = RAF x BAF [UNEP 1998][ARMSTRONG 1994].
The AF for SCC (Squamous Cell Carcinoma) has a greater degree of certainty than that for BCC because of uncertainties in its action spectrum. For MSC, the AF is probably even more uncertain [UNEP 1998]. Experimental studies for melanomas in fish indicate an RAF of 0.1 [SETLOW ET AL 1993] whereas for DNA damage in skin a RAF of 1.6 is more likely [KELKENS 1990]. In this assessment, the assumption is made that skin cancers depend on cumulative UV-B exposure, following the assumptions made by [MARTENS 1998]. Data for the AF of cataracts show a high degree of uncertainty and are based on [UNEP 1994].

		Skin cancer			**eye**
		SCC	**BCC**	**MSC**	**Cataract**
Biological amplification factor (BAF)	[%]	*2.5*	*1.4*	0.6	0.7
Radiation amplification factor (RAF)	[%]	*1.2*	*1.2*	1.2	0.75
Amplification factor (BAF*RAF=AF)	[%]	3	1.68	0.72	0.45
Total Incidence in 1990 [MURRAY 1996]	[cases/yr]	2.40E+05	9.60E+05	1.67E+05	8.33E+06
Lethal fraction [MURRAY 1996]	[%]	3	0.3	25	0.063
Excess cases per percentage ozone decrease per year					
Incidence at 1% ozone depl. (=total incidence*AF)	[cases/yr]	7.20E+03	1.61E+04	1.20E+03	3.75E+04
Mortality (=total incidence*lethal fraction)	[cases/yr]	2.16E+02	4.84E+01	3.01E+02	2.63E+01

Table 4.11: Percentage incidence increase per percent ozone depletion during one year.

4.8.4. Damage assessment

Based on the AF and the world-wide incidence of skin cancer and cataract in 1990 [Murray, 1996], the excess incidence as a result of 1% ozone depletion during 1 year is calculated. Mortality is calculated on base of lethal fraction of the disease and the incidence. Incidences and mortality are translated to DALYs using the approach of [HOFSTETTER 1998] and data from [MURRAY ET AL 1996] for age at onset of the disease, average duration of the disease and disability weighting.

Three-quarters of all DALYs per percentage of ozone layer decrease are caused by disabled years as a result of cataracts. Most DALYs caused by early death (years of life lost) result from increased mortality due to SCC and MSC.

		SCC	BCC	MSC	Cataract	All causes
Age at onset	[year]	55.9	55.9	55.9	62.8	
Average duration	[year]	4.2	4.2	4.2	1.9	
Disability weight	[-]	0.045	0.045	0.045	0.624	
Years Lived Disabled per case (YLD) (E+H)	[year]	0.19	0.19	0.19	1.19	
Years Lived Disabled per case (YLD) (I)	[year]	0.18	0.18	0.18	0.98	
Years of Life Lost per case (YLL) (E+H)	[year]	23	23	23	19	
Years of Life Lost per case (YLL) (I)	[year]	16	16	16	12	
DALYs due to YLD per % ozone depletion (E+H)	[DALY]	1.36E+03	3.05E+03	2.27E+02	4.45E+04	
DALYs due to YLD per % ozone depletion (I)	[DALY]	1.30E+03	2.90E+03	2.16E+02	3.67E+04	
DALYs due to YLL per % ozone depletion (E+H)	[DALY]	4.97E+03	1.11E+03	6.91E+03	4.49E+02	
DALYs due to YLL per % ozone depletion (I)	[DALY]	3.46E+03	7.74E+02	4.81E+03	2.83E+02	
Total DALYs per % ozone depletion (E+H)	[DALY]	6.33E+03	4.16E+03	7.14E+03	4.49E+04	6.25E+04
Total DALYs per % ozone depletion (I)	[DALY]	4.75E+03	3.68E+03	5.03E+03	3.70E+04	5.05E+04

Table 4.12: Calculation of the number of DALY.s per percent Ozone layer depletion. All skin cancer cases are treated equally. For the E+H perspectives The DALYs are calculated as follows:
DALY =YLD*weight*incidence+YLL*mortality.

4.8.5. The role of cultural perspectives

The ODP factors used in most publications have been calculated using a time frame of 500 years. In the individualist perspective a timeframe of 100 years would be more appropriate. [SOLOMONS 1992] shows that the ODP factors are highly time horizon dependent when the time horizon is less than several decades. On a time frame of 100 years, the difference is not so big. Substances with a considerably higher lifetime than CFC11 have a lower ODP, while substances with a short lifetime, such as HCFCs have a higher ODP.

In the damage assessment, age weighting is only applied in the individualist perspective. This leads to the following choices for the three perspectives:

- Individualists use age weighting in calculation of DALYs, and they prefer the ODP factors for a time perspective of 100 years. However, due to data availability problems, this cannot be implemented. As a result there is an additional uncertainty of σ^2_g of about 0,5
- Hierachists and Egalitarians do not use age weighting, and they use the ODPs for a time perspective of 500 years.

4.8.6. Sources of uncertainty

The damage assessment is based on percentage decrease of the ozone layer. Ozone layer decrease is depending on latitude and season, and so does the RAF. In this assessment, average data for 30S-55°N are assumed to be valid for world wide averaged ozone depletion and UV increase. This coarse assumption introduces an uncertainty of σ^2_g of approximately 2

One of the uncertainties which is already mentioned is that there is limited knowledge of the risk of ozone depletion and melanoma skin cancer and BCC. In addition UV may induce chronic effects which cannot be modelled as a result of lack of understanding [UNEP 1998].

Since skin cancer is supposed to depend on cumulative exposure, skin cancer risk increases with age. As population demographics change and people tend to become older, the incidence of skin cancer is likely to increase too [MARTENS 1998]. The same accounts for cataract. In this assessment is not accounted for an increase in life expectance since it is not common to account for changing population demographics in LCA. However, this may lead to an underestimation of the assessed damage since ageing of the population will probably lead to an increase of the excess incidence of 60% [UNEP 1998].

Since people are becoming aware of the fact that UV radiation can cause adverse effects, exposure habits tend to change [UNEP 1998], which will lead to a lower excess incidence of UV-related diseases.

Although some uncertainties are hard to quantify, the overall uncertainty on the damage factors is estimated to be $\sigma^2_g = 4$, for Hierarchist and Egalitarian perspectives, and 4,5 for the Individualist perspective.

4.9. Missing impact categories

4.9.1. Other toxic effects from heavy metals

Unlike in the predecessor, the Eco-indicator 95 method, we have only been able to model the carcinogenic effects of some heavy metals. Other human health effects of heavy metals, such as damage to nervous systems, liver etc. are not modelled here. This means the damages caused by some important non-carcinogenic heavy metals are not included.

It is difficult to say how important this omission is, as these metals have impacts on many different endpoints. Furthermore the association and causality between a metal and one or more endpoints is a difficult matter. In the Eco-indicator 95 we based the damage model on the Globe report [GLOBE 1992] from the RIVM. The Globe report mentioned high concentrations of lead in central Europe. The target level was set on the basis of this information. In Western Europe Globe considers this problem as being solved, since lead is no longer added to gasoline, and since most industries developed advanced emission treatment systems. If we would have made an update according the old methodology, we would probably have reduced the weight significantly, or even dismissed the issue.

Like in the case of Ozone-layer depletion this again shows the difference between the previous and the current method. In the previous method an issue is no longer important if the emissions are reduced; in the 99 method it makes no difference if the emissions are reduced. The only relevant matter is if there are significant health damages per kg of emission.

The [DOBRIS 1998] report gives an overview of the most important health effects that can be expected from such heavy metals, based on exposures to rather high concentrations. Such a table is rather mechanistic, as it remains unclear to what extent health problems will occur at more common, lower concentrations.

Type of disease	**Cardiovascular**	**Allergy and hypersensitivity**	**Reproduction**	**Nervous systems**	**Osteoporosis**
Sensitive group	Elderly	Children	Foetus, Young	Foetus, Children	Elderly
Contributing Metal	Arsenic*), Lead, Cadmium*), Cobalt	Nickel*) Chromium*)	Lead Mercury	Lead Manganese Aluminium	Lead Cadmium*) Aluminium Selenium

Table 4.13: Overview of possible health effects from some metals when humans are exposed to rather high concentrations. For the metals marked with) the carcinogenic effect has been modelled, but these metals have more effects than carcinogenic.*

4.9.2. Other toxic effects

The list of potentially toxic substances is sheer endless, and for many substances the toxic effect is unknown or not sufficiently documented. Examples are the so-called endocrine disrupters and persistent organic substances that build up in the food chain. For carcinogenic substances the list is deliberately limited per perspective. For other substances, a simple lack of sufficient data was the main reason for excluding these substances. Some examples are given in the Dobris Assessment [DOBRIS 1998].

It is unclear if these effects would cause significant damages in the LCAs of common products.

4.9.3. Noise

In [HOLLANDER ET AL 1999] a preliminary calculation is made of the number of DALYs related to noise under the Dutch population. Although physical stress due to noise is not easy to rank on the DALY scale, the conclusion is that noise can be an impact category that is as least as important as other impact categories discussed here. We do not know for sure, as the DALY calculation used by den Hollander differs on some important points from our approach.
More work needs to be done to establish a damage model that is compatible with our models.

5. Ecosystem Quality

5.1. Description of the damage category

Ecosystems are heterogeneous and very complex to monitor. There are a number of treaties and declarations (UNCED, UNEP, Council of Europe), that list attributes that are important to mankind, such as: biodiversity, aesthetic and cultural values, ecological functions and services, ecological resources and information functions (in genes).
One way to describe Ecosystem Quality is in terms of energy, matter and information flows. If we want to characterise Ecosystem Quality in terms of these flows, we could say that a high Ecosystem Quality is the condition in which the flows are not noticeably disrupted by anthropogenic activities. In contrast, a low Ecosystem Quality is the condition in which these flows are disrupted by anthropogenic activities. The level of disruption is thus the most important parameter to monitor ecosystem quality.
To complicate things further these flows can exist on many different levels. For instance the information flow can be described on the level of ecosystems, species and genes. The material and energy flow can be described in terms of free biomass production, as is proposed in [LINDEIJER ET AL 1998].

It is clear we cannot model all these attributes on all these levels and dimensions. For our purpose, we concentrate on the information flow, on the species level. This means we assume the diversity of species is an adequate representative for the quality of ecosystems.

5.1.1. Reversible damages

Practically all species groups can be affected by anthropogenic influence. It is impossible to monitor them all. We had to make a choice for the species groups that can be used as an appropriate representative for the total ecosystem quality. Furthermore it is important to choose between:

1. The complete and irreversible extinction of species.
2. The reversible or irreversible disappearance or stress on a species in a certain region during a certain time.

Although the first type of damage is probably the most fundamental damage to ecosystems, it is extremely difficult to model in the LCA context, since it requires information on the exact location of the last representatives of a species in relation to the location of an impact. In fact we can assume that complete extinction usually occurs as a result of many different factors. This means no single product life cycle causes the extinction, but all the product life cycles together are responsible for the full extinction.
In the second option we assume the damage caused by a product life cycle results in a temporary stress on ecosystems. This stress can be one of the factors that result in a full extinction of a species, but we do not know. The stress caused by a product life cycle is temporary as long as a functional unit is used with a limited time perspective. Such an LCA results in emissions that are expressed as a mass loading and a temporary occupation of an area. Even if habitats are destructed by land conversions we assume this damage will be restored.

The damage to Ecosystem Quality now can be expressed as:

the relative decrease of the number of species (fraction)* area * time

5.1.2. Modelling the effect on species groups

The crucial parameter in the model for Ecosystem Quality is the parameter that represents the effect on a species group. Unfortunately we have not been able to find a uniform parameter for this purpose, such as the DALY. We use two different expressions:

- For toxicity we use the PAF, the Potentially Affected Fraction of species, as proposed in [HAMERS ET AL 1996]. The PAF is used to express the effect on (mostly lower) organisms that live in water and soil, such as fish, crustaceans, algae, worms, nematodes, micro-organisms and several plant species. The PAF can be interpreted as the fraction of species that is exposed to a concentration equal to or higher than the No Observed Effect Concentration (NOEC). It is a measure for toxic stress, and in fact not a real damage, as defined here.

- For acidification, eutrophication and land-use we use the PDF of species, the Potentially Disappeared Fraction. The PDF is used to express the effects on vascular plant populations in an area. The PDF can be interpreted as the fraction of species that has a high probability of no occurrence in a region due to unfavourable conditions. The PDF is based on the POO, the Probability Of Occurrence, as used in [ALKEMADE ET AL 1996] to model the effects of acidification and eutrophication. The PDF is in fact represented by 1-POO. This means the fraction of species that does not occur can also be described as the fraction of the species that has disappeared. For this project the PDF concept is also used for land-use.

This means we do not have a uniform damage unit for the damage category ecosystem quality, as we have in the damage category Human Health. There are two problems:

1. We use different species groups as representatives for the total ecosystem: vascular plants for acidification, eutrophication and land-use and a broad range of (mostly lower) aquatic and benthic organisms for toxic effects.
2. We use different levels to determine the effects, the level at which species are affected and the level at which species disappear.

The reasons for modelling damage for different impacts on different species groups are as follows:

- The different impacts are based on separate models. For each model the relation between a specific impact and an effect on the species level is described in a different way, providing the best scientific basis for that specific dose-response relationship.
- For toxic effects in soil and water, the relation between the diversity of aquatic and benthic species and the NOEC derived from laboratory testing is at present the best scientific basis to translate from emissions to toxic effects on the ecosystem level.
- For land-use, acidification and eutrophication the observed occurrence of vascular plants derived from field monitoring is at present the best scientific basis to determine the relation between impact and damage.

Modelling the effect on higher organisms, such as birds and mammals or reptiles is even more difficult, as the species migrate, have complex food patterns, as they are usually at the end of the food chain. Perhaps most importantly, these species are all very different in their response to stresses; therefore it is hard to treat them as a group. We assume that the occurrence and health of a selection of aquatic and benthic species and vascular plants, which are usually essential providers for adequate food supply and other habitat characteristics, are a good indicator for the health and occurrence of the higher species.

The reasons for differentiating between potentially *affected* and *disappeared* fractions is partially pragmatic: the different models have different results, and partially fundamental :

- In ecotoxicity the NOEC is widely in use to determine the toxic effect. Alternative measures are the Lethal Concentrations, such as LC50 or LC5. These are concentration levels at which 50 or 5% of the population has died. A problem with the LC values is that lower species can rather easily adapt to higher toxic stress levels. This means the laboratory test used to determine LC values are difficult to translate to conditions in the field, where long-term exposure is dominant. Not enough information from field observations is available to use real observed damage that can be related to the disappearance of these species.
- It is difficult to establish a measure at which we can say a vascular plant is *affected* by a certain condition. It is much easier to determine if a species has disappeared or simply cannot exist under measurable field conditions.

5.1.3. Combining PAF and PDF

As PAF and PDF are very different, we cannot simply add damage expressed as PAF and PDF. The fact that different species pools are used is acceptable when we assume all species have equal importance. This means we assume an orchid species is just as important as a fish, algae or nematode.

The biggest problem is the difference between the level at which species become affected and at which level they disappear. In paragraph 5.9 an attempt is made to develop a conversion factor.

The damage to Ecosystem Quality will thus be expressed as PDF*area*time [m^2.yr]

One additional complication arises in the model for eutrophication and acidification. For eutrophication and acidification we will see that this assumption of equality among species is not always justified. This is because eutrophication and acidification do not always decrease the number of species, but they merely shift the species composition. In fact eutrophication will very often tend to increase the number of species. In that case we still consider eutrophication to be damaging, as very often rare and unique species will be replaced by common species. In other words, for acidification and eutrophication the reference will not be the percentage of species, but the percentage of "target" species. The target species, which represent the natural state of specific ecosystems are derived from [BAL ET AL 1995]. The table below summarises the approaches:

	Species	**Damage unit**	**Definition of "threatened"**	**Assumption on equality among species**
Ecotoxicity	Several species	PAF*m^2*yr	Toxic stress (above NOEC)	All organisms equal
Acidification	Vascular plants	PDF*m^2*yr	Probability of disappearance	Only target species included
Eutrophication	Vascular plants	PDF*m^2*yr	Probability of disappearance	Only target species included
Land use	Vascular plants	PDF*m^2*yr	Probability of disappearance	All vascular plants equal

Table 5.1. Differences between damage units of impact categories

These differences show that the damage category Ecosystem Quality is not as homogenous as the other damage categories. As a result we are introducing a considerable uncertainty when these impact categories are combined to a total indicator for Ecosystem Quality.

Until now we have not found applicable models that would express the damage caused by climate change, increased UV radiation, Photochemical Smog and Changes in the groundwater table in terms of PDF or PAF.

5.2. Damage to Ecosystem Quality caused by ecotoxic substances

Based on the report by [BAKKER AND VAN DE MEENT 1997], some parts are quoted directly.

5.2.1. Fate analysis

The fate analysis for ecotoxic substances included in the Eco-indicator 99 methodology is carried out with EUSES. A detailed description of the calculation procedure can be found in chapter 3.
The result of the fate analysis is a link between an emission to air, water, agricultural soil and industrial soil and concentrations in water, and pore water of agricultural, industrial and natural soil.

5.2.2. Effect analysis

The method used to calculate damage to Ecosystem Quality is the elaboration of the concept by [HAMERS ET AL 1996], providing an algorithm to calculate the toxic stress on ecosystems denoted as a Potentially Affected Fraction (PAF) of species. The value of PAF indicates the fraction of naturally occurring organisms exposed to concentrations higher or equal to the laboratory NOEC. The toxic stress in a multiple substances exposure situation is indicated by the indicator for toxic effect substances (Itox), which is in fact equal to the combined PAF or combi-PAF. Secondary poisoning is not incorporated into the PAF calculations. The main exposure route is assumed to be water for aquatic ecosystems and pore water for terrestrial ecosystems. The exposure route through food is considered not to be important.

A substance specific dose-effect curve, which is representative for the naturally occurring organisms has to be calculated. It is assumed that the dose-effect curve can be described by the log logistic distribution function of NOECs. The log logistic distribution function is estimated from single species toxicity data. The distribution function is based on chronic NOECs. PAF is calculated from the combination of the estimated distribution function and the calculated field concentration.
In formula the log-logistic function can be described as:

$$PAF(c) = \frac{1}{1 + e^{(\alpha - \log c)/\beta}}$$

c: concentration of the substance
α: parameter calculated from the average NOEC for a single substance for all species
β: coefficient derived from the standard deviation of the NOECs for the substance.

The distribution function is characterised by the mean value α and the standard deviation σ of log transformed NOECs. Alpha is thus the average of the log NOECs, representing the average toxicity of the substance and bèta is equal to approximately 0.5σ representing the variation in the NOECs. At least 4 NOECs must be available to determine these parameters in a reliable way.

The combined toxic stress in a multiple stress situation can be calculated according to two methods from the single species combination toxicology, respectively concentration addition and effect addition. concentration addition can only be applied to inert hydrophobic substances, the mode of action of such substances is called narcosis. Effect addition is applied to all other substances.

5.2.3. Damage analysis

For LCA purposes , a specific way to add up damages from combined emissions of a product system, which is a combination of concentration and effect addition, has been worked out by [MEENT ET AL 1999]. Since spatial and temporal information is not included in LCA, an average background concentration for all substances, equal in all areas of Europe, has to be assumed. A marginal increase of the concentration of one single substance, resulting from a product system, has only a very small influence on the average situation in Europe. It is postulated [MEENT ET AL 1999] that the many different chemicals present cause concentration additive effects. According to [MEENT ET AL 1999] the marginal damage to ecosystems from a marginal increase of the concentration of a single substance depends on the present level of damage from the mixture of substances already present in the environment. This means that the slope of the single substance PAF curve is not relevant, but the slope of the overall PAF curve, based on mixtures of substances, which are present in the European environment, must be determined to assess the marginal damage from an emission.

Such a PAF curve for mixtures can be constructed by standardising the concentrations of individual substances into units of average toxicity of the total mixture, so called Hazard Units (HU). Therefore the marginal concentration increase of a substance must be divided by the average NOEC ($=10^{\alpha}$) for that substance, creating standardised hazard units which are very similar to the well known PEC/PNEC ratios. The effects of different levels of pollution by unknown (but presumably relative invariable) environmental mixtures, standardised to Hazard Units follow the hypothetical logistic curve shown in figure 5.1. This curve yields the total toxic stress (i.e. the proportion of species for which the NOEC is exceeded = combi-PAF), as a function of the sum of hazard units in the mixture [HAMERS ET AL 1996], which can be viewed as the toxicologically standardised mixture concentration. HU=1 (PEC/PNEC=1) means that all species are exposed to a background level equal to the average NOEC (which is based on the distribution of NOECS of all species). Since 50% of the species has a NOEC below this average, this implies that these 50% of all species are affected. This explains that at HU=1 the potentially affected fraction is 50%.

Based on [ZWART AND VAN DE MEENT 1998], an appropriate standard deviation of mixture toxicity can be found in a β-value of 0.4. The working point is determined by the slope of the PAF-curve for mixtures at the present level of toxic stress in Europe. According to RIVM [MV 1997], ambient levels of combi-PAF in water and soil in the Netherlands are typically 10%-50%. We assume that the European value of combi-PAF lies within the same interval. This means the slope of the curve in the possible working points varies with almost a factor of three with a combi-PAF between 10 and 50% (see calculations in the annex report). Since no additional information is available, the geometric mean (24%) is used as average European Combi-PAF.

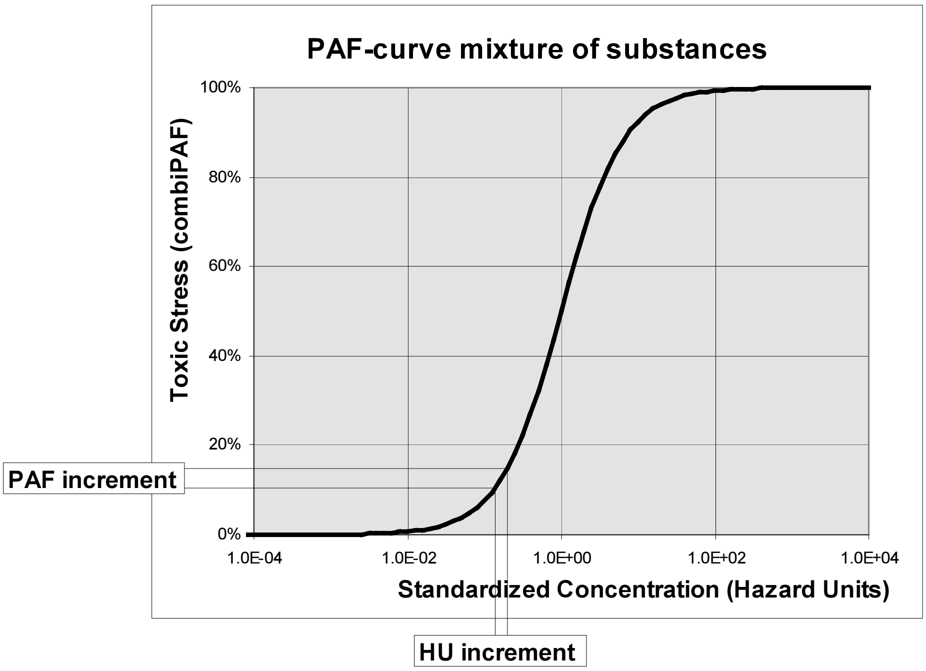

Figure 5.1. Damage to Ecosystem Quality is indicated by the Potentially Affected Fraction (PAF) of species. The dose-response relationship of the ambient mixture of substances follows a logistic curve. Concentrations of single substances are standardised to Hazard Unit (HU), representing the background mixture of substances.

From the marginal increase of the concentration the marginal increase in hazard units can be calculated. When the working point is determined the marginal damage from the marginal increase in hazard units can be calculated.

The procedure to calculate damage to Ecosystem Quality resulting from an emission can now be described as follows:

- Determine the temporary, marginal increase of the concentration in a specific environmental compartment from the fate model, for each specific substance (in section 3.1 it is explained that a mass loading, as found in inventory table, can only cause a temporary increase in the concentration).
- Determine the increase in standardised toxicity units (hazard units) from the concentration increase of the substance for each emitted substance that may cause an impact on Ecosystem Quality using the average NOEC of each substance. Add up the total increase in hazard units.
- Choose a reference value for the slope of the combi-PAF curve for substance mixtures representing the present ambient level of toxic stress (working point).
- Determine the temporary marginal damage (in the environmental compartment considered) from the total increase in hazard units using the slope of the combi-PAF function at the workpoint. Multiply the calculated increase in combi-PAF with the total area of the environmental compartment.

For one specific emission, this procedure is repeated for the concentrations in all relevant environmental receiving compartments separately (water, agricultural soil, industrial soil, natural soil). In table 5.2 an example of the calculation procedure is given for an emission to air and the resulting damage in natural soil. Finally the damages in PAFm2yr of the different compartments can be added up, resulting in the total damage in Europe.

Calculation step	Calculation procedure	Result
Emission to air in Europe	10.000 kg/d standard flow	1.0E-6 $kg/m^2/yr$
Concentration increase (ΔC) in natural soil	EUSES	6.96E-7 mg/l
No Effect Concentration (NEC terrestr.)	Geometric mean NOECs	1.04 mg/l
Hazard Units increase (ΔHU)	$\Delta HU = \Delta C/NEC$	6.69E-7
Slope factor $\Delta PAF/\Delta HU$ at Combi-PAF=24% (European average)	Slope factor = 0.593 $\Delta PAF/\Delta HU$	
PAF increase in natural soil for 10.000 kg/d in Europe	$\Delta HU*0.593 = \Delta$ PAF	4,13E-7
PAF increase in natural soil for 1 kg/yr in Europe	$\Delta PAF/(10.000*365)$	1,130E-13
$PAFm^2yr$ in natural soil (2.16E+6 km^2) for 1 kg in Europe	1,13E-13*surface area natural soil	0.244 $PAFm^2yr$

Table 5.2 Calculation procedure for emissions to air and resulting damage in natural soil

5.2.4. The role of cultural perspectives

The cultural perspectives are only applied to the fate analysis of the substances (see chapter 3). The time horizon for the fate modelling of metals is not the same for all perspectives. Egalitarians and Hierarchists use the long term perspective and Individualists use the short term perspective. No other assumptions are related to the perspectives.

5.2.5. Sources of uncertainty

The PAF concept that is presented here contains a substantial amount of uncertainty. The fundamental uncertainty lies in the assumptions and consequences of choices made in the construction of the calculation principles. This fundamental uncertainty can not be quantified easily. Fundamental uncertainties are:

- Are the NOEC data from laboratory tests representative for the total of the species?
- Is the NOEC distribution in fact log logistic?
- Is the addition of PAFs for combinations of substances valid?

The operational uncertainty is determined by the variation in the input data and can be quantified.
A source of operational uncertainty is:

- Uncertainty in the NOEC-distribution. Assuming the laboratory experiments are representative and the NOEC are distributed log-logistic, the average and standard deviations of the distribution are known. The accuracy of calculating the average and standard deviation is small but known.

In the report of [Bakker and van de Meent 1997] upper and lower limits for the operational uncertainty of the average NOECs for part of the substances are given. Depending on the number of NOECs that is available the distance between average and lower or upper confidence limit can vary a lot (see annex report for details).

5.3. Damage to Ecosystem Quality caused by acidification and eutrophication by airborne emissions

5.3.1. Description of the problem

acidification and eutrophication are caused by depositions of inorganic substances such as sulphates, nitrates and phosphates. These depositions occur mainly through air and directly into water. The primary effect is the change in nutrient level and acidity in the soil.

Airborne emissions also influence aquatic ecosystems. As we will see the model takes aquatic systems such as wetlands and swamps only into account if they can be classified as natural areas. The effect of airborne emissions on rivers, canals and lakes is not taken into account. In many parts in Europe this is not a big problem as the direct emissions into water are often much more important. However, especially in Scandinavia many lakes are reported to be heavily acidified by airborne emissions from central Europe.

In the case of ecotoxicity, it can be assumed that any increase in toxic pressure results in a damage. For changes in the nutrient and acidity levels this is not so simple. For almost all plant species there is a clearly defined optimum combination of nutrient level and acidity. Any deviation from this optimum is detrimental for that specific species. As a result, changes in nutrient levels will mainly cause shifts in the species populations. Sometimes these shifts result in an increased number of species, sometimes there is a decrease.

The problem here is that we need to find out to what extent a shift can be considered to be a damage. This means we have to differentiate between desired and less desired species populations. We cannot make general statements about desired and less desired species in general. We have to consider the desired species population per type of ecosystem. In the method described below, a list of target species that has been developed for over 40 types of ecosystems is used [Bal et al 1995]. The target species represent the natural state of a specific ecosystem. It is clear there are subjective elements in the selection of these target species. The main criterion is whether species can be considered to be typical and representative for an ecosystem or not.

With these target species, we can now monitor the effect of depositions on these target species. We can only do this per ecosystem, which means we have to incorporate a Geographic Information System (GIS) into the model, and we have to select the "projected" ecosystem into each grid-cell. Once such a system is set up, we can model the effect of depositions on the PDF[18] for the target species in that cell.

As the substances causing eutrophication and acidification are inorganic, their fate cannot be modelled with EUSES. The model we use for damage analysis has its own fate model, based on the characteristics and background levels of each grid cell.

In our model we only take into account changes in natural areas. Changes in acidity and nutrient levels in agricultural areas as a result of depositions are not really relevant, as these depositions are small compared by the application of nutrients and acidity regulating agents applied by farmers. A big problem at this moment is the lack of models for eutrophication and acidification of aquatic systems.

So far we only have found a Dutch model (the Natuurplanner, see below) that is able to translate changes in depositions into changes in the PDF of plants. This is of course a serious limitation. Our temporary solution is to assume that the Dutch natural areas have the same sensitivity as the European natural areas. We expect that this simplification results in a number of biases, due to the specific characteristics of the Dutch natural areas.
There are no rocks, let alone mountains in the Netherlands. Furthermore, the largest part of natural areas is formed by sandy dune-like landscapes near the coast, and in the centre or the north east of the country.

5.3.2. The "Nature planner"

For the Eco-indicator 99 we had access to the "Natuurplanner" [LATOUR ET AL 1997], or "Nature Planner" that has been developed by RIVM[19]. The Nature Planner uses a 250 by 250 metre grid for the Netherlands. The Nature Planner contains several databases with information on vegetation, soil conditions and fate models, which are combined directly with effect models. It aims at the combined effect assessment of eutrophication, acidification, desiccation, fragmentation, climate change and pollution by toxic substances on ecosystems and species (multi-stress). It is meant for a national or regional scale. At this moment the Nature Planner is operational for the effect assessment of eutrophication, acidification and desiccation on the vegetation and butterflies.

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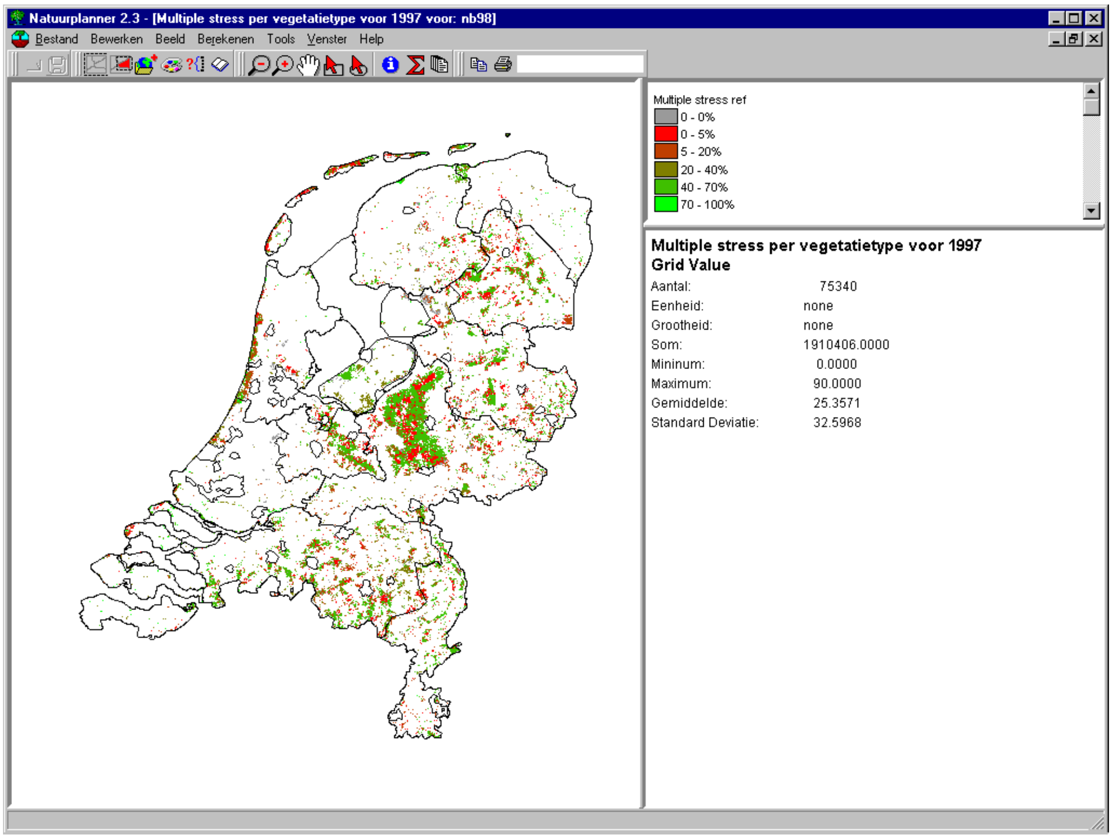
[18] In the computer models we used, the damage is expressed in POO (probability of occurence). In this report we use PDF, which can be defined as PDF=1-POO

[19] We thank Rob Alkemade for explaining us the system and allowing us to use his computer for several days. The Natuurplanner Software is only used inside RIVM, and under continuous development.



Multiple stress ref
▨ 0 - 0%
🟥 0 - 5%
🟧 5 - 20%
🟫 20 - 40%
🟩 40 - 70%
🟩 70 - 100%

Multiple stress per vegetatietype voor 1997
Grid Value

Aantal: 75340
Eenheid: none
Grootheid: none
Som: 1910406.0000
Mininum: 0.0000
Maximum: 90.0000
Gemiddelde: 25.3571
Standard Deviatie: 32.5968

Figure 5.2: Typical result of the output of the Nature Planner. Only the grey areas are taken into account. The rest is considered as non-natural areas. The grid-cell size is 250 by 250 metres.

The Nature Planner consists of two parts: a soil model (SMART) and a vegetation response model (MOVE):

- SMART can be regarded as a fate model, as it calculates the pH and the nutrient level, expressed as Nitrogen availability, and the availability of water. The desiccation part of the model is not used here. Calculation has been performed with a fixed groundwater level.
- MOVE can be considered as a damage model, as it calculates the effects of the SMART results on the PDF for each grid-cell.

5.3.3. Fate analysis with SMART

SMART [KROS 1995] is a simple one-layer dynamic soil model, which includes a N-cycle (litter fall, mowing, litter removal, mineralization, nitrification, denitrification and uptake), geochemical processes (weathering and cation exchange) and a simple hydrological cycle including solute transport through upward seepage. This model predicts the changes in soil pH and N-availability in the root zone on a year by year basis. An indicative validation shows generally a reasonable agreement with pH and N-data from other literature.

Normally SMART is used with actual or projected depositions per grid cell. The result is the total effect of the damage. For our model we are interested in the marginal effect of adding a certain flow over a certain period. For our calculation we used the latest version of the deposition data (used for [RIVM 1998]), and we added a deposition of 10 mole NO_x, SO_x and NH_3 per km^2 to each grid cell. As depositions are usually in the order of several 100 Mole's, this increase can be considered to be a marginal change.

Of course the deposition of ten moles is not the same as the emission of ten moles. We propose to use the same basic reasoning as in the EUSES model. We assume Europe to be closed, so every mole emitted in Europe is deposited there. Next we must consider that only a part of the emissions is actually deposited on the natural soils. If we assume an even distribution, we can assume that only 60% of the emission is deposited on natural soil, since 60% of the surface area of Europe consists of natural soil [EUSES 1996].

The MOVE model was used to calculate the decrease in pH and the increase of Nitrogen availability due to the increased deposition for each substance in each grid cell. As the model also calculates the average, and the distribution, we can summarise the results in the table below.

	Deposition increase mole per year per hectare	Deposition increase in kg/m2/yr	Resulting acidity pH	Change in pH	Resulting nutrient availability	Change in nutrient availability
Reference	0		**4.648**		**7077.33**	
SOx	10	6.4*E-5	**4.646**	-0.0018	**7075.61**	-1.72
NOx	10	4.6*E.5	**4.547**	-0.1010	**7089.31**	11.98
NH3	10	1.7*E-5	**4.648**	-0.0004	**7090.22**	12.89

Table 5.3: Main results for the fate modelling with SMART assuming all substances are deposited on natural soil.

From the calculation results we can make a number of observations:

- There is a strong relation between the deposition of NH3 and an increased nutrient availability, while the deposition of SOx results in an slight decrease of nutrient availability. This can be explained as nutrients become less available if the pH drops, and as Nutrient availability is expressed as nitrogen availability.
- There is a weak relation between the deposition of NH3 and acidity, while the relation between acidity and NOx deposition is strong.

5.3.4. Damage modelling

The results from SMART form the input for the vegetation model MOVE. This model consists of the response functions of more than 900 Dutch plant species. The response functions describe the relationship between the PDF and the soil acidity, nutrient condition and the moisture condition and their mutual interactions. The most important source for these response functions is [Ellenberg et al 1992]. [Alkemade et al 1996] made a verification study to compare (and adjust) the Ellenberg data with measured data.

The model MOVE can calculate the potentially disappeared fraction for these values per grid-cell. A species is considered to meet unfavourable conditions if this probability is lower than some threshold value (set to 2,5%). These species suffer from stress caused by the combined effect of acidification and eutrophication. The number of stressed target species are counted per grid-cell and the results can be aggregated for the total natural area of the Netherlands, resulting in a percentage of threatened species caused by a specific deposition.

The table below shows the main results of the damage modelling as PDF.m2.yr per kg of emitted substance.

Emission	**Deposition increase in kg/m2*yr on natural soil**	**Average PDF in the Netherlands**	**Damage to Ecosystem Quality in PDF*yr*m2 per kg emitted substance (100% deposition in natural areas)**	**Damage to Ecosystem Quality in PDF*yr*m2 per kg emitted substance[20] (60% deposition in natural areas)**
reference value		**0.746429**		
SOx	6.4*E-5	**0.74654**	**1.73**	**1.04**
NOx	4.6*E.5	**0.746867**	**9.52**	**5.71**
NH3	1.7*E-5	**0.74687**	**25.94**	**15.56**

Table 5.4: Damage caused by three emissions.

The table shows that, according to the model, the PDF for target species is three quarters of the full range of target species. This means the damage due to acidification and eutrophication is high in Dutch natural systems. The standard deviation on this figure is 0.32.

As the $PDF_{reference}$ and the $PDF_{10mol/hectare}$ are almost equal, the difference is very small. This is still permitted as the uncertainties in the PDF calculations are synchronised. This subtraction is a way to find the slope of the damage curve (coefficient of direction).

In the third column of table 5.4 the result is given for a deposition of 1 kg per square metre in a natural area of undefined size, with the natural characteristics of the Netherlands with 100% deposition in natural areas. In the last column the result is given for Europe, where 60% of the total area consists of natural areas. Therefore the figures are multiplied by 0.6.

5.3.5. Limitations of the modelling

It is still a problem that aquatic ecosystems are not covered by the nature planner and damage from phosphorous emissions is also not included. Calculation of PDF for acidification or eutrophication of the aquatic environment is therefore not possible up till now. We still have to find a solution for this.

5.3.6. The role of cultural perspectives

We have not been able to separate differences for the cultural perspectives. However, in a future development, we could consider to use the following points:

- The use of target species, and the way they are defined are probably a cause for differences of opinion (in fact they are a point of discussion among experts). Individualists would probably argue that target species should only be used to a very limited extent. They would include a much larger range of species. For instance they would not leave out target species because they have been imported from abroad, as is the case in the current lists. As far as we have been able to interpret the criteria for inclusion of target species it seems the current definition would especially be appealing to Egalitarians.
- The Nature Planner allows the use of a weighting factor for the rarity of a specie. If a specie is rare, it gets a higher factor. We have chosen not to use this weighting factor.

5.3.7. Sources of uncertainty

Although the Nature Planner is a very sophisticated model, it is only applicable to the Netherlands. We have made the very crude assumption that the average sensitivity of Dutch natural areas is representative for the average sensitivity of other natural areas.
The calculation of the damage factors was performed by subtracting the result of two figures close to 0.75, with a standard deviation of 0.32. The Nature Planner does not provide suitable data to calculate the standard deviation for the difference, but we propose to assume an estimated σ^2_g of 2 to reflect the uncertainties in the calculation, including the uncertainties that are due to the fact that only the Dutch region is used.

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[20] The resulting figures are larger than one. This is because of the chosen units. 1 kg deposition per square metre is a very large dose. If the damage would be expressed per hectare the figure would be 4 orders of magnitude lower.

5.4. Damage to Ecosystem Quality caused by land-use

5.4.1. Description of the problem

The impact of land-cover changes on ecosystems is very significant. In most parts of Europe this influence is perhaps more significant than the effects of many other impact categories [MÜLLER-WENK 1998-2]. As we will see, land-cover changes do not only have effects on a specific local area, also the surrounding region can be affected. Furthermore we have to distinguish land occupation and land transformation. This means the damage model must be developed in four different versions.

Unlike other damage models, the data required for the land-use model is based on empirical data, such as observations of species numbers in different types of land-cover, in stead of extrapolations of laboratory data and computer models. This aspects has some important consequences for the model:

- The observed number of species is the result of many different influences, such as the concentration of toxic chemicals and the nutrient and acid level, or even the influence of increased UV patterns or climate changes. This means it is impossible to separate the effect of land-use changes from other impact categories. At the end of this chapter we will propose a method to avoid the most serious aspects of this double counting.
- There are many different land-cover types and the species number on these types can vary widely in different parts of Europe. We will use the CORINE inventory to classify land-use types [CORINE 1991]
- The data availability on field observations is a big problem in two ways:
  - There is sufficient data for just a very few land-use types
  - The land-use types for which data is available are not always suited for practical application in LCA

The methodology for the regional effect was originally developed by [MÜLLER-WENK 1998-2]. [KÖLLNER 1999] developed an approach and collected data to reflect the local effect of land-use. Approximately at the same time [LINDEIJER ET AL 1998] published his proposal, containing some conceptional elements as Köllner. For the Eco-indicator 99 the data and some of the theoretical concepts are taken from [KÖLLNER 1999] and [MÜLLER-WENK 1998-2].

5.4.2. The species area relationship

A complicating factor in land-use models is the species area relationship. The number of species increases with the area size. The relation between the size of area and the species diversity was first described by Arrhenius in 1921, and is generally regarded to be adequately expressed by:

$$S = a * A^b \qquad (1)$$

with

S = species diversity

a = species richness factor, usually between 20 and 2000

A = area (in hectare)

b = species accumulation factor, usually between 0.2 and 0.5

This simple formula as well as a number of alternative expressions, has been tested by several authors on empirical data. In general a good fit is found [KÖLLNER 1999].

The species area relationship implies that if a cornfield made in a natural area, there will be two effects:

- The number of species on the cornfield will decrease. This is referred to as the *local* effect; it is the effect that occurs *on* the area that is being used or converted.
- The number of species on the untouched natural area will be decreased, as the natural area becomes slightly smaller. This is referred to as the *regional* effect; it is the effect *outside* the area that is used or converted

Both the regional and the local effect will be modelled here.

Another consequence of the species-area relationship, is that when an existing area with cornfields is expanded at the expense of a natural area the species number on the cornfield area slightly increases, while the species number on the natural area decreases. In general the increase of species in the cornfields can be neglected, for two reasons:

- Most species that occur on artificial areas, can also occur in natural areas. The increase in artificial areas does not add anything to the species diversity in the region.
- At the present condition of the European environment the decrease of species in natural areas is much more important than the increase in non-natural areas[20]

5.4.3. Land conversion and land occupation

There is a distinct difference between the following two cases:

1. Land that is being converted from one state to another.
2. Land that has been converted earlier and is occupied for a number of years.

It is useful to distinguish these two cases.

A typical example is the production of corn in an old agricultural area. In LCA, this activity cannot be held responsible for the fact that once the area was converted from a natural area long ago. However, each year a certain area remains occupied and can not return to its original natural stage. For this reason the damage due to land occupation is seen as the damage caused by preventing the occupied area from returning into its natural condition.

A typical example of land conversion is the mining activity in a pristine natural area. For each ton of extracted metal, a small additional area is converted from its natural conditions into a mining pit. After the mineral has been extracted, it will take a considerable amount of restoration time before the area returns to a situation that has the same diversity as the original situation. If the mining operation occurs in a agricultural area, the change in species numbers will be smaller, and it may take less time before the area returns to a situation that has the same diversity as the original condition.

The fact that we have impacts from two different types of processes (conversion and occupation) and impacts in two different types of areas (regional, local) to consider, results in four different versions of the damage model (see table 5.5).

5.4.4. The general principle for the damage model

The concept of PDF can be rather easily applied to model the regional and local damage caused by land occupation and conversion. The potentially disappeared fraction of vascular plant species is expressed as the relative difference between the number of species S on the reference conditions and the conditions created by the conversion, or maintained by the occupation. Thus the PDF can be generalised as:

$$PDF = \frac{S_{reference} - S_{use}}{S_{reference}} \qquad (2)$$

With:

$S_{reference}$ = Species diversity on the reference area type (see below for the choice of the reference)
S_{use} = Species diversity on the converted or occupied area

The damage to Ecosystem Quality can be calculated when the PDF is multiplied with the appropriate area and time span. As we can see below the area size and time span is different for the four versions of the damage model.

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[20] An interesting discussion can be found in the Dobris assessment [Dobris 1996]. Without man's influence most parts of Europe would be covered with a single large forest. Although this forest must have had open spots, in general the flora and fauna must have been relatively monotonous. When man started to construct open spaces for agricultural purposes, initially this must have lead to a larger species diversity in Europe. This increase in species diversity has probably continued until around 1850. Since that time the species diversity has been on a steady decline. It is suggested that this decrease has been the result of a rapid transformation of large natural areas into agricultural and urban land, in combination with a sharp increase of emissions.

$$EQ = PDF * area * time = \frac{S_{reference} - S_{use}}{S_{reference}} * A * t \qquad (3)$$

This general formula can be used for both occupation and conversion and for regional and local effects. Table 5.4 specifies how the factors in equation (3) should be applied.

Type	**$S_{reference}$**	**S_{use}**	**Area size A**	**Time period t**
Conversion Local	Original state	S on new land-use type	Converted area	Restoration time
Occupation Local	Natural state	S on new land-use type	Occupied area	Occupation time
Conversion regional	Original state	Smaller natural area*)	Natural area**)	Restoration time
Occupation regional	Natural state	Smaller natural area*)	Natural area**)	Occupation time

*Table 5.5: Overview of the parameters determining the 4 types of land use. These parameters can be combined using equation (3). *) With smaller natural area we mean the species number that occurs when the natural area is reduced by conversion or is kept small due to occupation. **) The species reduction occurs in the natural area outside the converted or maintained land.*

The restoration time must be estimated depending on the type of land before and after the transformation. An important problem is that most land-use types are never restored into the exactly original condition. For instance a drained swamp will never become a swamp again. The criteria for selecting the restoration time should not be the estimated time before the area is exactly the same as it was, but the time it takes to form a comparable quality as before. We propose the following defaults, if no other information is available:

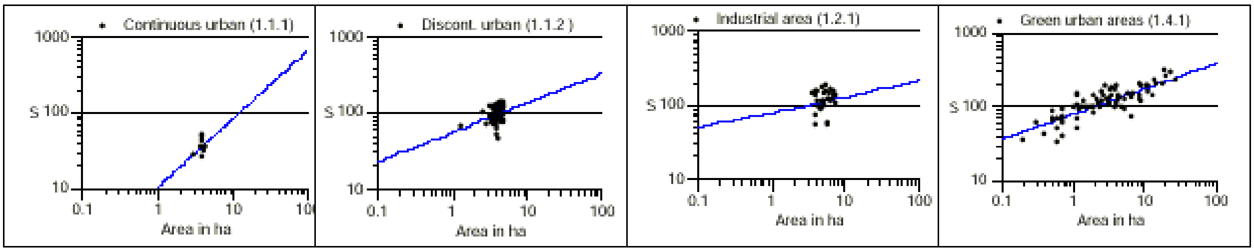
- For conversions from agricultural to urban areas and vice versa we choose a restoration time of 5 years
- For conversions between natural areas to urban or agricultural areas we choose a restoration time of 30 years, unless it is obvious that the restoration will take much more time.

These defaults are more or less compatible with the defaults used in the [ESU 1996] database. The main difference is that in this database the assumption is made that restoration of a natural area always takes 100,000 years to return to it's original state. Since we do not take the original state with the original combination of species of an area as a reference, but a natural state with an equal state of biodiversity as the reference natural state, we do not think 100,000 years is appropriate in our case. [Müller Wenk 1998-2] proposes to use a restoration time of 30 years as a general default.

5.4.5. Data for the species number per land-use type

There are many different types of land-cover, which makes it rather difficult to develop a coherent set of characterisation values. [Köllner 1999] uses the [Corine 1991] land-cover nomenclature and definitions. The Corine system (see annexe report) describes land-cover types on different levels of detail. The data found by [Köllner 1999] is not always very suited for LCA practitioners. For instance he finds data on different types of fallow land. Fallow land is not a result of economic activities modelled in LCA, therefore they are not included here.

[Köllner 1999] has carefully analysed available data, mainly from [Reidl 1989] for artificial surfaces and [Streiber 1995] for agricultural surfaces. Figure 5.3 shows the species-area relation for eight area types on a log-log scale.



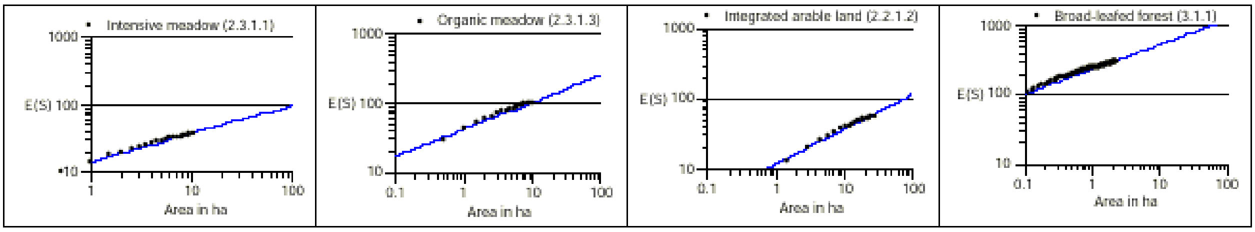


Figure 5.:3 Examples of species area relationship for six area types, taken from [KÖLLNER 1999]. The dots represent empirical data. The line is the best fit for the species area relation. Please note both scales are logarithmic. The upper row is based on observations in Essen, the German Ruhr area, the lower row is based on observations in Switzerland.

The example chosen in figure 5.3 show some of the problems with such data. For instance the curve for the continuous urban land-use (top left) is based on just 10 observations. As a result the slope of the species area relationship is very uncertain in this case. The match for agricultural systems seems very good, but this is partially an effect of the way the data points are calculated.
The data on agricultural land-use is based on observations on the fields only. The diversity of the edges, waterways and hedges in between the plots are not included. It is clear that these areas usually contain a relatively high species diversity. We assume that the real species diversity lies between a factor of 1 and 4 higher. Therefore we propose to assume that the actual species diversity (S) in agricultural areas is twice as high as is indicated in the data presented by Köllner, with a σg^2 of 2. Of course this is a somewhat arbitrary choice, but we feel this correction factor gives a result that is more realistic than applying no correction factor at all.

Although the species area relationship is a well-established phenomenon, it is not very easy to use in LCA. In fact it is not only a very complicating factor, it also leads to fundamental allocation problems. As is discussed before in the description of the flux pulse problem (box 3.1), it is usually not possible to establish the absolute size of the area influenced by a product life cycle.

For instance, if we are analysing an electric razor, we know that the razor factory occupies a certain amount of space, let us assume 1 hectare. If the factory produces 1 million razors per year, we can say the hectare is needed during one millionth of a year for a single razor. However, we could also say that a single razor needs one 1 square meter during 3.65 days. Both types of reasoning are correct.

The example shows that the species area curves have no direct meaning in LCA application, as we do not know the area without specifying a time period. There are several ways to use the species area relationship to derive values for the S (species-number) parameter in equation (3):

1. Use the species accumulation factor (term b in equation (1))
2. Use the species richness factor (term a in equation (1))

In figure 5.4 the species richness and species accumulation factors are presented for a number of area types. It is clear that the factors result in very different (and practical inverse) ranking.

[LINDEIJER ET AL 1998] proposes to use the species accumulation factor b. [KÖLLNER 1999] argues that the accumulation factor is not a very good indicator for the quality of ecosystems. In fact most natural ecosystems have a relatively low accumulation factor, compared to artificial or agricultural systems. This means the species number increases rapidly when small areas are increased, but when areas become bigger the species diversity is not increasing very much anymore.
The species richness factor is an inherent property of the land cover type. Therefore we use the species richness factor to characterise the differences between the land-use types. This means the species number S can now be replaced by the species richness factor a in equation (3).

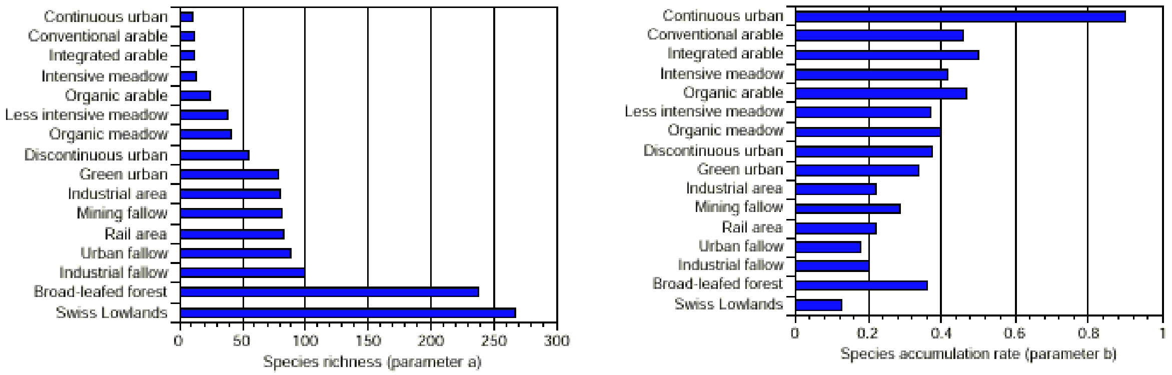


Figure 5.4: The difference between the species richness factor a and the Species Accumulation factor b for a number of area types. The result of the ranking shows very big differences. Taken from [KÖLLNER 1999]

For occupation the reference should be the species richness factor of the natural systems. As the term “natural system” is not very well defined, and as natural systems have very different characteristics in different parts of Europe, there is a serious problem here.
As a temporary solution, we propose to use the species richness factor calculated by [KÖLLNER 1999] for the Swiss lowlands. Due to simple lack of data, we do not know if this is a proper choice[21].

In table 5.6 the values for the species richness factors found by [KÖLLNER 1999] are summarised. This source also presents upper and lower limits of the 95% confidence interval. The column correction is the correction factor discussed above. The PDF values are calculated with the species richness of the Swiss lowlands as the reference values. These PDF values can thus be directly applied for occupation. In the calculation of the uncertainty we have taken into account the uncertainty in PDF and we have assumed the correction factor could be between 1 and 4[23].

	CORINE classification	**n**	**a**	**a** $_{low}$	**a** $_{high}$	**correction**	**PDF** $_{natural\rightarrow use}$	σ^2_g
Continuous urban	1.1.1	9	11.0	0.7	164.0	1.0	**0.96**	2.4
Discontinuous urban	1.1.2	59	54.6	36.6	90.0	1.0	**0.80**	1.2
Industrial area	1.2.1	29	81.5	27.1	244.7	1.0	**0.70**	7.3
Rail area	1.2.2.2	41	81.5	73.7	90.0	1.0	**0.70**	1.05
Green urban	1.4.1	75	81.5	73.7	90.0	1.0	**0.70**	1.05
Conventional arable	2.2.1.1	16	12.2	11.0	13.5	2.0	**0.91**	1.2
Integrated arable	2.2.1.2	18	12.2	11.0	13.5	2.0	**0.91**	1.2
Organic arable	2.2.1.3	12	24.5	24.5	27.1	2.0	**0.82**	1.5
Intensive meadow	2.3.1.1	20	14.9	13.5	14.9	2.0	**0.89**	1.2
Less intensive meadow	2.3.1.2	17	40.4	36.6	40.4	2.0	**0.70**	2.1
Organic meadow	2.3.1.3	20	40.4	40.4	44.7	2.0	**0.70**	2.5
Broad-leafed forest	3.1.1	126	244.7	244.7	244.7	1.0	**0.10**	1.0
Swiss Lowlands (nature)		46	270.4	200.3	330.3	1.0	**0.00**	

Table 5.6. Overview of the PDF values for the local effect, using mixed broad-leafed forest as a reference. a is the species richness factor, n is the number of observations on which this factor is based. The mean, low and high value for a represent the 95 percent confidence interval. The uncertainty for industrial areas is very big.

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[21] One argument to support this choice is the likelihood that if the land occupation takes place in a region (climate zone) with a much higher or lower natural species richness factor, the species richness for artificial areas is also higher or lower in about the same proportions. As a result, the relative differences, as used in the PDF could be quite insensitive to this choice of reference.

5.4.5. Modelling the local damage

With the data presented here we can calculate the local damage to conversion from land-use type 1 to type 2 with the following formula:

$$EQ_{local,conversion,1\rightarrow 2} = \frac{a_1 - a_2}{a_2} * A * t_{rest.} = PDF_{1\rightarrow 2} * A * t_{rest.} = \left(PDF_2 - PDF_1\right) * A * t_{rest.} \qquad (4)$$

For instance, if a organic meadow is converted into a continuos urban area, the damage to ecosystems is (0.96-0.7)* area*5 years. Please note that land conversions can have a positive and a negative effect.

For the local damage due to occupation the damage formula is:

$$EQ_{local,occupation} = \frac{a_{natural} - a_{use}}{a_{natural}} * A * t_{occupation} = PDF_{natural\rightarrow use} * A * t_{oc\,\mathrm{sup}\,ation} \qquad (5)$$

For instance, if after conversion the urban area is used 50 years, the damage is 0,96*area*50 years. Occupation should not have a negative damage, as the reference is always the natural system, which is supposed to have the highest species richness.

5.4.7. Modelling regional effects

When a natural area is transformed into an industrial complex, the species area relationship (eq.3) dictates that the species number in the remaining natural area will decrease. At the same time there will be a slight increase in the industrial area, as its size increases. In the introduction of the land-use model we have already stated that this increase is insignificant, and thus can be neglected. The decrease of species in the natural area is the regional effect of land conversion.
The regional effect for land occupation can be described in a similar way. As long as the area is prevented from returning to the natural state, the species number in the natural area is kept low in comparison to the natural state. The only difference between the regional effect of occupation and conversion is the use of different types of time periods (see also table 5.5). In this paragraph the term conversion is used, to avoid having to write "conversion or occupation" many times.

In order to explain the regional effect we assume there are only two types of land-use, called Hi (High intensity) and Li (Low intensity) use of land, as proposed by [MÜLLER-WENK 1998-2]. Low intensity (Li) used land can be interpreted as areas that are not under pressure from human activities. Li land does not necessarily need to be a forest. Some forests are under high pressure, while on the other hand abandoned railroads and industrial areas can have many species. In this simplified approach the species diversity on the Hi used land is not taken into account.

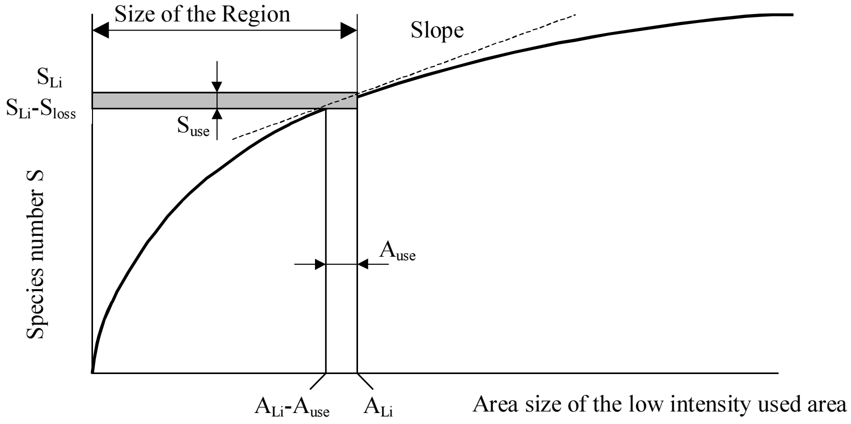

Figure 5.5: The species area relationship for Low intensity (Li) land plotted on a linear scale. A decrease of the Li area will result in a decrease of the species number. The grey area is the regional effect.

The regional effect is illustrated in figure 5.5. In this figure a hypothetical species area relation is plotted for the Li area. When the current size of the region A_{Li} is reduced with an certain amount A_{use}, the species number in the whole Li area drops from S_{Li} to S_{Li}-S_{use} . The regional damage can be illustrated by the grey area in the graph, The size of that area is A_{Li}*S_{use}. S_{use} can be calculated by multiplying the slope of the species area relationship with A_{use} .

$$S_{use} = \frac{dS(A_{Li})}{d(A_{Li})} * A_{use} \quad (6)$$

As the damage to ecosystems EQ is defined as a relative damage, we must divide S_{use} by S_{Li} . This results in the following equation, in which (the first derivative of) equation (1) can be substituted:

$$EQ_{reg\cdot Li\rightarrow Hi} = \frac{S_{use}}{S_{Li}} * A_{Li} * t = \frac{\frac{dS(A_{Li})}{d(A_{Li})} * A_{use}}{S_{Li}} * A_{Li} * t = \frac{a * b * A_{Li}^{b-1} * A_{use}}{a * A_{Li}^{b}} * A_{Li} * t = b * A_{use} * t \quad (7)$$

The result is remarkably simple. The damage to Ecosystem Quality is directly proportional to the species accumulation factor b of the Li area, the size of the converted area and the time. Contrary to what one would expect, the regional effect is independent of the size of the region.

This phenomenon can be explained in a number of ways:
- Taking a hectare from a small region will cause a bigger loss in species, compared to taking one hectare from a large area. However, to calculate the damage, the greater species loss is multiplied by a small region size, while the small species loss in the big region, must be multiplied with the larger area.
- There are some similarities with the flux pulse problem. It is impossible to know the size of the area or the change in the number of species separately, but we do know the product of area and change in species.
- One of the properties of the exponential curve is that both effects compensate each other completely.

An important, but unanswered question so far, is the value of the species accumulation factor b for Li areas. From the data collected by [KÖLLNER 1999] it is clear that more or less natural systems tend to have a relatively low species accumulation factor and a high species richness factor. [KÖLLNER 1999] finds for the Swiss lowlands a species accumulation factor of 0.13, and a value of 0.36 for broad-leafed forests. We propose to use a factor b=0.2 as a default, with a σ_g^2= 2 to reflect the uncertainty in this assumption. From figure 5.4 it can be concluded that this is more or less the average value.

To verify this choice, we have studied the work of [MÜLLER-WENK 1998-2]. In a completely different approach [MÜLLER-WENK 1998-2] also presents a calculation for the regional effect for the Swiss lowlands and for the former West Germany[22]. His results are summarised in table 5.7

Region	**Area size [km^2] (~A_{Li})**	**Change in species number [%] (~S_{use})**	**Regional effect factor (area size times change in species number) (m^2)**
Swiss lowlands	798	0.033%	0.263
West Germany (low)	8900	0.001%	0.089
West Germany (high)	9650	0.0018%	0.174

Table 5.7: Some results for the calculation of the regional effect as performed by [MÜLLER-WENK 1998-2]. The last column is not from this source, but calculated by the authors

Our interpretation of the findings of [MÜLLER-WENK 1998-2] is that there is a good match between the values he finds and the value of 0.2 we have proposed.

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[22] Müller Wenk introduced to us the concept of regional damage. In his view the regional damage should be calculated for the area size of a whole region (for instance on a country level), and not just the area size of the natural soil. This leads to a totally different result. However, the basic data collected by him can be used to check our model.

5.4.8. Differentiating between land-use types in the regional effect

The simple distinction between Hi and Li is rather crude to model real life situations. Changing one hectare of a natural area into a “less intensive meadow” will have a smaller regional effect compared to changing one hectare into a continuous urban area, as a part of the natural species will still continue to exist on this “less intensive meadow”. Apparently, for some of the species the meadow will not cause a reduction of the area size. Apparently the regional effect is smaller when the land-use type that is being created has a higher species number or a higher species richness factor.

We propose to use the PDF for the local effect as a factor that expresses this adequately. For occupation this means we use the $PDF_{natural\rightarrow use}$. for conversion we use the $PDF_{1\rightarrow 2}$.

As a result, the regional damage for conversions can be written as

$$EQ_{regional,conversion,1\rightarrow 2} = PDF_{1\rightarrow 2} * b * A_{use} * t_{restoration} \qquad (8)$$

For occupation the regional effect is:

$$EQ_{regional,occupation} = PDF_{natural-use} * b * A_{use} * t_{occupation} \qquad (9)$$

5.4.7. Combining Local and regional effects

To reflect the total damage caused by land-use changes, or land occupation, the regional and local effect must be added:

$$EQ_{occupation} = EQ_{regional} + EQ_{local} = PDF_{natural\rightarrow use} * b * A_{use} * t_{occupation} + PDF_{natural\rightarrow use} * A_{use} * t_{occupation}$$
$$= (b+1) * PDF_{natural\rightarrow use} * A_{use} * t_{ocupation}$$

(10)

For conversions the total effect is:

$$EQ_{conversion,1\rightarrow 2} = EQ_{regional} + EQ_{local,1\rightarrow 2} = PDF_{1\rightarrow 2} * b * A_{use} * t_{restoration} + PDF_{1\rightarrow 2} * A_{use} * t_{restoration}$$
$$= (b+1) * PDF_{1\rightarrow 2} * A_{use} * t_{ocupation}$$

(11)

Assuming the value of 0,2 for b is correct, this means effectively that the total effect is increased by 20% due to the regional effect.

5.4.9. Practical values for LCA

Table 5.8 summarises the values for the regional, local and total effect for occupation and conversion. The damage to Ecosystem Quality can be calculated by multiplying the proper PDF values from the table with the area and the occupation or restoration time to get the damage to ecosystems.

	PDF Occupation		**Local and regional PDF of conversion from column to row**												
	Only local	regional plus local	Continuous urban	Conventional arable	Integrated arable	Intensive meadow	Organic arable	Less intensive	Organic meadow	Discontinuous urban	Industrial area	Rail area	Green urban	Broad-leafed forest	Swiss Lowlands
Continuous urban	*0.96*	1.15		0	0	0.01	0.05	0.11	0.11	0.16	0.26	0.26	0.26	1.04	1.15
Conventional arable	*0.95*	1.15	0		0	0.01	0.05	0.10	0.10	0.16	0.26	0.26	0.26	1.03	1.15
Integrated arable	*0.95*	1.15	0	0		0.01	0.05	0.10	0.10	0.16	0.26	0.26	0.26	1.03	1.15
Intensive meadow	*0.94*	1.13	-0.01	-0.01	-0.01		0.04	0.09	0.09	0.15	0.25	0.25	0.25	1.02	1.13
Organic arable	*0.91*	1.09	-0.05	-0.05	-0.05	-0.04		0.06	0.06	0.11	0.21	0.21	0.21	0.98	1.09
Less intensive meadow	*0.85*	1.02	-0.11	-0.10	-0.10	-0.09	-0.06		0	0.05	0.15	0.15	0.15	0.91	1.02
Organic meadow	*0.85*	1.02	-0.11	-0.10	-0.10	-0.09	-0.06	0		0.05	0.15	0.15	0.15	0.91	1.02
Discontinuous urban	*0.80*	0.96	-0.16	-0.16	-0.16	-0.15	-0.11	-0.05	-0.05		0.10	0.10	0.10	0.84	0.96
Industrial area	*0.70*	0.84	-0.26	-0.26	-0.26	-0.25	-0.21	-0.15	-0.15	-0.10		0	0	0.72	0.84
Rail area	*0.70*	0.84	-0.26	-0.26	-0.26	-0.25	-0.21	-0.15	-0.15	-0.10	0		0	0.72	0.84
Green urban	*0.70*	0.84	-0.26	-0.26	-0.26	-0.25	-0.21	-0.15	-0.15	-0.10	0	0		0.72	0.84
Broad-leafed forest	*0.10*	0.11	-1.04	-1.03	-1.03	-1.02	-0.98	-0.91	-0.91	-0.84	-0.72	-0.72	-0.72		0.11
Swiss Lowlands	*0.00*	0.00	-1.15	-1.15	-1.15	-1.13	-1.09	-1.02	-1.02	-0.96	-0.84	-0.84	-0.84	-0.11	

Table 5.8: Values for the relative species change for local and the local plus regional occupation in the first two columns. The matrix in the other columns can be used to calculate the effect of conversions.. The damage to Ecosystem Quality can be found by multiplying the proper PDF value in this table with the area and the occupation or restoration time.

5.4.10. Separating damage from land-use and emissions

The species richness indicator is based on empirical data. It is clear that changes in species richness are partially caused by emissions to soil. Especially on agricultural land the application of pesticides and fertiliser will have very important impacts on the species richness.

To avoid double counting the following rules should be observed in the inventory by applying this concept in an LCA-study:

- Fertilisers that are directly applied on agricultural soil should not be counted as an emission. The damages caused by these substances are adequately represented in the species diversity of agriculturally managed land. This means the Eco-indicator does not allow for modelling subtle changes in the application of such substances. Only larger changes, like moving from conventional farming practices to biological practices can be modelled.
- Fertilisers that evaporate or leach out to the water should be taken into account. For evaporation an air emission of N should be included. For the emission to water no damage model is included in the Eco-indicator 99. Additional damage assessment for this emission is necessary.
- Pesticides that are directly applied onto the agricultural soil should be included as an emission to agricultural soil in an inventory. The fate and effect of this emission has been calculated in the damage model for ecotoxicity described in section 5.2. In the ecotoxicity damage model the damage that occurs in water and areas surrounding the agricultural land is included, but the damage on the agricultural soil itself is excluded (set to 0) in the calculation to avoid double counting with land-use.
- Pesticides in the form of application losses, which are emitted directly to air or water during application should be included in the inventory respectively as emissions to air or water.

These rules are important for the consistency of the application of the method. The benefit of the rules is the simplicity of application, combined with the rather sophisticated modelling. The disadvantage is that agricultural practices can only be differentiated if sufficient data on area-species relationships are available.

5.4.11. The role of cultural perspectives

The uncertainties in the modelling of land-use seems to be dominated by data uncertainties and not by value choices. However, there are a number of value choices:

- In the egalitarian perspective it could be maintained that indeed regional effects are more important than local effects, as regional effects are probably the most important reason for the world wide, and thus irreversible, extinction of plant species. On the contrary, Individualists will probably see the regional effects as less important, as these effects are not really visible on the short time scale, and perhaps even difficult to prove. A problem is that we do not know how much more or less important the regional effect is for these perspectives.
- The Hierarchist would also argue that the local effect is the most important, as local changes can be monitored and regulated.
- The Egalitarian and the Hierarchist would probably use a longer restoration time, compared to the Individualist.

For the time being these aspects are not included. Land-use is treated the same for all perspectives.

5.4.12. Sources of uncertainty

The following uncertainties are calculated or estimated:

- For the local effect there is uncertainty in the data needed for the species area relation. We can distinguish different quality levels depending on the number of observations and the quality of the observations. The data presented in table 5.6 reflect this uncertainty. It also reflects the estimated uncertainty in the correction factor. The σ_g^2 ranges from 1.04 to 7.3.
- For the regional effect the largest uncertainty is the difficulty to assess a proper value for the species accumulation factor b. This adds a σ_g^2 value of approximately 1.1 for the combined regional and local effect.

For conversions the uncertainty can become very big if two PDF values that are close to each other have a high uncertainty.

5.5. Combining the different damages to Ecosystem Quality

As described in the introduction of the Ecosystem Quality chapter, this damage category is far less homogeneous than the other two. As illustrated in table 5.1 there are basically two different definitions of the damage to ecosystems:

1. Toxic stress, expressed as Potentially Affected Fraction of species (PAF)
2. The disappearance of plant species, expressed as the Potentially disappeared Fraction (PDF)

The damage to ecosystems is found by multiplying the PAF or PDF with an area and a time.

The question is if, and how, PAF and PDF can be combined. The solution for this problem as proposed here is an intermediate one, as there is not enough scientific information to make such a combination in an elegant way.

Combining the effects of land-use and eutrophication/acidification is relatively straightforward, as both measures of damage express the relative number of (target) vascular plant species that will disappear on a certain area. Apart from this similarity, there are also a number of differences:

- In eutrophication/acidification modelling, a plant species was considered to have been disappeared if the PDF was higher than 97.5% (the probability of occurrence was less than 2,5%). In the case of land-use, no such data is available. The disappearance is determined by using empirical data, and not by using models.
- In land-use modelling, no difference between target and other species is made, but also the definition of disappearance of species is practically the same.

In spite of these differences, it seems reasonable to combine the damage from land-use and acidification/eutrophication directly.

The combination of toxic pressure on a broad range of (mostly lower) organisms and the potentially disappeared fraction of plant species is very difficult for a number of reasons:

- The selected species groups for the calculation of toxic pressure play a different role in ecosystems than plants. It is unclear if we can give the same value to the disappearance of 10% of the vascular plant species, as to the disappearance of 10 of the selected species groups for PAF calculation .

- Toxic pressure (based on NOECs) does not represent real damage, as even at a very high PAF only a small number of species (if any) may disappear completely.

The first point is a valuation problem. For the time being we propose to treat both cases equal.
The second point seems a mere scaling problem. For instance if the PAF curves would have been based on LC50 values, instead of NOECs, a damage expressed in PAF would have a closer similarity to the potentially disappeared fraction of plant species. A PAF based on LC_{50} expresses real damage, since an LC_{50} means 50% of the population has died. Although it is difficult to make general statements about the relation between NOECs and LC50 values, the following table, compiled from a random selection of examples, suggests that the PAF curve based on a LC50 values would be positioned at a concentration level that is 5 times the original PAF.

Substance	**Selection**	**LC_{50} / NOEC**	**Number of tests***
Benzene	all data on aquatic species	1.7	36
Toluene	all data on aquatic species	4	28
Naphthalene	all data on aquatic species	1.7	17
Cadmium	data on crustaceans	2.8	17
	all data on aquatic species	4.9	47
Mercury	data on crustaceans	17	19
Nickel	all data on aquatic species	3.0	15
PCP	all data on aquatic species	4.6	62
Fluoranthene	all data on aquatic species	4.3	31
Average		4.9	

**) at least 3 or more NOEC tests*

Table 5.9: Selection of NOEC and LC50 toxicity data for several toxic substances

A PAF_{LC50} of 50% could be interpreted as the situation in which for 50% of the species the population is halved. An average PAF_{LC50} for all species means that for 50% of the species 50% has died, resulting in a total of 25% loss of individual organisms. When we consider the meaning of a PAF_{LC50}, we could expect a relation between the effects on the population size (more than 50% of the population has died) and the number of species that have merely disappeared.
Knowing this relation cannot be proven we postulate that a PAF_{LC50} of 50% means that 25% of the species have almost completely disappeared. In general terms, a PAF_{LC50} value of x% indicates that x/2% of the species have disappeared.

Combining these two postulates, we propose to divide the PAF results by a factor 10, before they can be added to the damages from eutrophication/acidification and land-use. As a result:

PDF=PAF/10

An uncertainty estimate is difficult to make, because most of the uncertainties are not related to data uncertainties, but to conceptual uncertainties. Cultural perspectives are not useful here.

5.7. Missing impact categories

5.7.1 Acidification and eutrophication by waterborne emissions

So far it has not been possible to develop a consistent model for the damage caused by eutrophication and acidification of rivers and lakes. There are a few important problems:

- So far no data has been found establishing a link between the nutrient level respectively the acidity and the occurrence of species.
- The damage to aquatic systems will have to be expressed as the occurrence of species per m^2.yr. The relative surface area of aquatic systems in Europe is low (3% of the total surface area) compared to the land area [EUSES 1996]. If the importance of aquatic systems would be based only on their surface area, many people would find that their importance is underestimated.

5.7.2 Damage to Ecosystem Quality caused by climate change

In the Eco-indicator 95 method the only damage model available for climate change was for damage to Ecosystem Quality. In the Eco-indicator 99 method, the only damage model for climate change that is used is for damage to Human Health.

For the Eco-indicator 95 model the [GLOBE 1992] report from the RIVM was used to model damage to ecosystems. In discussions with RIVM we found the model is no longer supported. No new model could be found to model the effect of climate change on ecosystems.

5.7.3. Damage to Ecosystem Quality caused by increased UV radiation

Also for damages caused by UV radiation, no damage model could be found. [UNEP 1998] presents a good overview of the research results on this subject. It is clear that Ecosystem Quality can be negatively, but also positively affected.

6. Resources

6.1. Description of the problem

The damage category Resources is new to the Eco-indicator methodology. Unlike the damage categories Human Health (DALY) and Ecosystem Quality (Biodiversity), we have not found an internationally accepted standard to express the damage to Resources. A new concept had to be developed. Before damages can be calculated, it is necessary to analyse the meaning of "damage to Resources".
The first part of the model, the "Resource Analysis" can be compared with a kind of inverse fate modelling: The decrease in resource concentration due to extractions is modelled; the second part is the actual damage model, where decreased concentrations are translated into the concept of surplus energy.

There are different ways to group resources:

- Mineral resources, like metals
- Bulk materials such as sand, gravel and lime
- Energy resources, such as fossil fuels
- Flow resources, such as solar energy, hydropower etc.
- Environmental resources, like soil, water and air
- Biotic resources, such as biodiversity and silvicultural products (wood, fish, etc.)

This rather wide definition of resources partially overlaps with the other damage categories, especially in the field of environmental resources. The availability of clean water and air as well as biodiversity are covered in the other damage categories. In the Eco-indicator 99 methodology we only model mineral resources and fossil fuels. The use of agricultural and silvicultural biotic resources and the mining of bulk resources such as sand or gravel are considered to be adequately covered by the effects on land-use[23].

Box 6.1. Some problems with resource depletion

In general there are three important problems when resource depletion is described:

- The stock size (or in the case of flow resources: the supply rate) is very much dependent on the effort mankind would like to make to get a resource. As will be shown later, we cannot determine the stock size or the maximum flow of most resources without specification of the efforts to extract the resource.
- To some extent, most resources can be substituted by other resources. Even between the categories of resources substitution is often possible (replace steel by wood). Because of this it is difficult to determine the essential property of a resource, and thus why depletion of such a resource would be a problem. The essential property determines the primary function the resource has to mankind. Usually this is an economic function.
- Some resources are not really used in the sense that they disappear after use. In principle all minerals stay on earth, and can theoretically be recycled. This is not the case for fossil fuels. Although they do not disappear, their useful essential property is lost. Following [MÜLLER-WENK 1998-1] we distinguish dissipative and non-dissipative use of resources.

Box 6.2. Some proposed methods to deal with resource depletion

[MÜLLER-WENK 1998-1] gives a critical analysis of the methods that have been proposed for LCAs. Non of the approaches seem to be completely satisfactory. We can summarise the problems as follows:
The use to stock ratio, as for instance is proposed in [HEIJUNGS 1992] suffers from the difficulty mentioned above that the stock size is completely determined by the lowest grade mankind is willing and able to exploit. [GUINÉE ET AL 1996] shows that depending on this definition the stock size can differ up to 6 orders of magnitude. Furthermore, the resource estimates are heavily influenced by political considerations [CAMPBELL and LAHERRÈRE, 1998]. They show that, as soon as the OPEC linked the production quota of the member countries to the reserve base, the reserve base of almost all countries grew very fast in a year. After this miraculous increase the reserve base remained constant. These figures simply can not be correct.

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[23] Bulk resources are abundantly available for the foreseeable future in most regions. Therefore we assume that the surplus energy will not increase. In many countries the real problem is the land conversion problem. For instance in the Netherlands the extraction of lime and gravel will stop completely within a few years, while the proven reserves for lime would at least cover the present consumption rate for 300 years.

The exergy method[24]. [FINNVEDEN 1996] proposes to apply the exergy concept as a way to characterise mineral depletion. Although exergy analysis is a very useful way to analyse the energy efficiency for chemical processes there are several problems in this proposal: [Müller-Wenk 1998-1] notices that the exergy concept is directed at conserving exergy of ores and not of metals. He demonstrates that most of the exergy in many ores are in parts of the ore that are not used. This makes the use of exergy a rather abstract measure, and he remarks that this loss of exergy has no relation with the reasons why mankind is worried about the loss of resources.

The sustainable process method, as used in the EPS system. [STEEN 1992] basically assumes that ultimately mankind must extract minerals from average earth crust material, such as rock. It is clear that this would require massive amounts of energy. Steen assumes that, by extracting a resource now, future generations will need this massive amount of energy to continue extracting the resource. Steen does not take into account that by that time it might be much more attractive to substitute the resource with another, or to increase efforts to recycle materials. He weights the present day extraction with the energy requirements that would be needed in the far future, using present technological skills. This assumption is an extreme case, as it does not indicate when this situation could occur (in many cases the time scale could be millions of years).

6.2. Description of the damage category Resources

After this critical analysis, a number of requirements for the method that deals with mineral resources and fossil fuels can be defined:

1. The method should not be based on the estimated *quantity* of the remaining stocks, as it is impossible to determine quantity without defining the quality requirements for the resource. This means that not the quantity of the resource is the limiting factor, but the quality.
2. The method should not be based directly on speculative future scenarios, as these are impossible to predict in a meaningful way. Of course it is impossible to avoid looking in the future when we discuss damage to the resource base.
3. The method should reflect the “real” reason why mankind is worried about the depletion of mineral resources and fossil fuels.

We propose a method that only takes into account the long-term trends of lowering resource quality. The primary assumption in this method is that if the resource quality is reduced, the effort to extract the remaining resource increases. Plain market forces will ensure that mankind always exploits the resources with the highest quality. This means each time a kg of a resource is used, the quality of the remaining resources is slightly decreased and thus the effort to extract the remaining resources is increased. This decrease of quality and thus increase of future effort is used to express the damage to Resources.

A complicating factor in the modelling of decreasing quality of our resources, is the fact that the geological processes leading to mineral and fossil resources are completely different. Mineral resource formation started through a series of very complex processes from the early stages of the formation of the earth. Fossil resources formation occurred through a series of much better understood processes in a much more recent period in history, “only” half a billion year ago. Because of this difference, we will treat the resource analysis of mineral and fossil resources separately:

- For mineral resources (including uranium), the most important quality parameter is the concentration[25]. The lower the concentration, the greater the effort to extract the resource.[26]
- For fossil fuels the concentration parameter is not really useful. Instead we will use the necessary “effort” to extract the resource.

Our proposal has some weak points:

- The assumption that there will be no sudden and discontinuous changes in the gradual decrease of resource quality. Geostatistical models indicate that such sudden decreases in mineral resource grades are unlikely. However in the case of oil and gas some abrupt changes are to be expected.

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[24] Exergy can be described as quality of an amount of energy.

[25] There are also other parameters, such as the location of the resource. However we assume that there is no difference, from a resource depletion model point of view, between extracting from a remote location or a easily accessible location. Of course in a proper LCA, the effect of the transport to remote locations is included.

[26] The concept has many similarities with the fate modelling, as we transform an extraction of a kg in a change of concentration. However, in the case of resources we do not have to use a time allocation, as there is no equivalent of degradation time here. Extraction of a kg has a permanent and not a temporal effect on the remaining resource concentration.

- All mineral resources are considered to be of equal importance to mankind. We do not take into account that it would perhaps be more useful to mankind to conserve copper instead of mercury, or the other way around[27].
- The possibilities of substitution of a mineral by another are not taken into account. Likewise, the possibilities of future recycling of minerals are used in a non dissipative way are excluded. Of course in a proper LCA, the recycling rate of a particular product is included, but one could argue that it is a more serious matter if copper is dispersed as a fungicide in wine fields or "stored" as copper tubes on a landfill. In the later case future reclamation may become possible, in the first case this seems highly unlikely.

On the other hand there are some important strong points:
1. The model is not *directly* dependent on estimates of future annual consumption, which will be very much influenced by recycling, substitution and other economic changes, and the model is also not dependent on the public communications of mining companies on their prospecting results.
2. The expected increase in the effort to extract resources seems to reflect a real concern of mankind.

6.3. Resource analysis, using geostatistical models

Geostatistical models can be used to analyse the relation between availability and quality of minerals and fossil fuels. This step could be described as "resource analysis" in analogy with the fate analysis. Instead of modelling an increase of a concentration resulting from an emission, we model the "decrease" of a concentration as a result of an extraction.

6.3.1. Minerals

In geostatistical models for minerals, it is generally accepted that the distribution of concentrations of mineral resources is lognormal if we plot quantities against grade. This phenomenon has been described, for single deposits, as Laski's law [DEYOUNG 1981]. There is a wide agreement amongst resource geologists that the lognormal ore grade distribution is a reasonable approximation also for the world-wide ore occurrences of a large part of minerals. Although real proof for this relation is not easy to provide, an illustrative example for the case of uranium is available from Deffeyes [DEFFEYES, 1964].

Deffeyes has determined the average concentrations of uranium in different types of rock and in water. This data, combined with data on the world distribution of rock types has been combined in figure 6.1. The grade varies in this graph from 50000 to 0.0005 parts per million. From this graph we can see that the size of the resource stock is completely dependent on the grade we are willing to mine.

For our purpose we do not have to take the full distribution into account, as we are only interested in the marginal effect of present extractions. Like in the case of toxicity models, the slope of the damage curve is needed, at the present average concentration. This is at the far left-hand side of the curve.
In the same report Deffeyes also presents data on the distribution of other resources. [Chapman and Roberts 1983] refers to the work of Deffeyes and bases his analysis of the seriousness of mineral extraction on data from Deffeyes. Figure 6.2 (taken from Chapman) shows the relation between resource availability and the concentrations. If the slope is steep, the resource availability increases sharply as the concentration decreases slowly. The quality of minerals with a steep slope decreases relatively slowly when the extraction continues. Chapman refers to the slopes as the M values.

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[27] This is the default setting for the method here. In principle there is no problem to add a weighting factor to the surplus energy values developed below, for instance to express the opinion that copper is a more valuable resource than mercury, or even oil. Setting such weights would require a carefully conducted procedure.

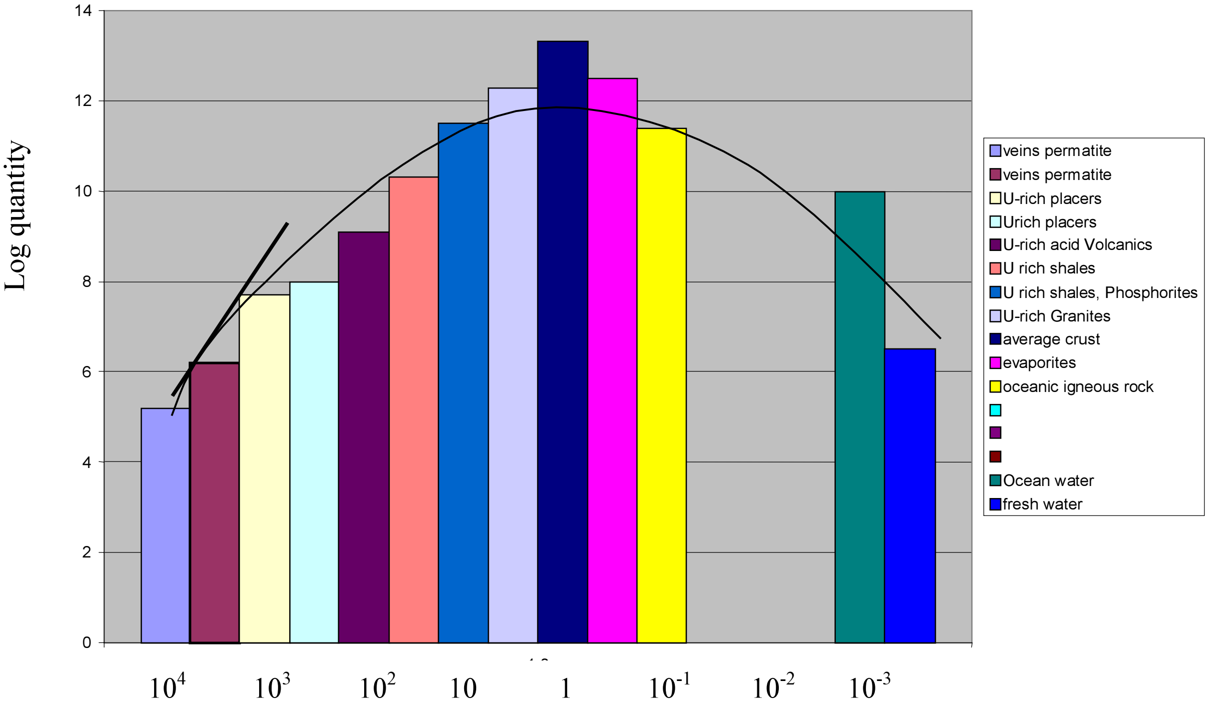

Figure 6.1: Distribution of the Uranium over the worlds crust, measured data according to [DEFFEYES 1964]. The grade on the horizontal axis ranges from 50000 to 0.0005 parts per million; every two bars is one order of magnitude.

Box 6.3. Unclear data references

Unfortunately the source of the data used by Chapman is not clear. His only reference seems to be [DEFFEYES, 1964] ,and in this report we only find a graph, without a table and any reference to the origin of the data. An extensive literature search and attempts to contact the authors did not yield any results. [MÜLLER-WENK 98-1] Has critically analysed Chapman's and Deffeyes' data and compared it with another independent source [DE VRIES 1988].

Table 6.1 shows the differences in the slopes of the distributions according to Deffeyes and de Vries. The table uses the M values, which are defined as the slope of the quantity/quality relation.

For some minerals the data show good correspondence, while in the case of Chromium and Copper the correspondence is bad. We follow Müller-Wenk and use the Chapman data, plus the data for Tin and Iron from de Vries. .

	M according to Chapman	M according to de Vries	Accordance
Aluminium	25	22	good
Chromium	17	4	bad
Copper	1.6	5	bad
Lead	3.4	3	good
Manganese	19	7	medium
Mercury	2	3	good
Nickel	2.9	6	medium
Tungsten	6	3	medium
Zinc	7.3	5	good
Tin		1	
Iron		18	

Table 6.1 Comparison of the calculated slopes according to Deffeyes and de Vries; taken from [MÜLLER-WENK 1998-1]

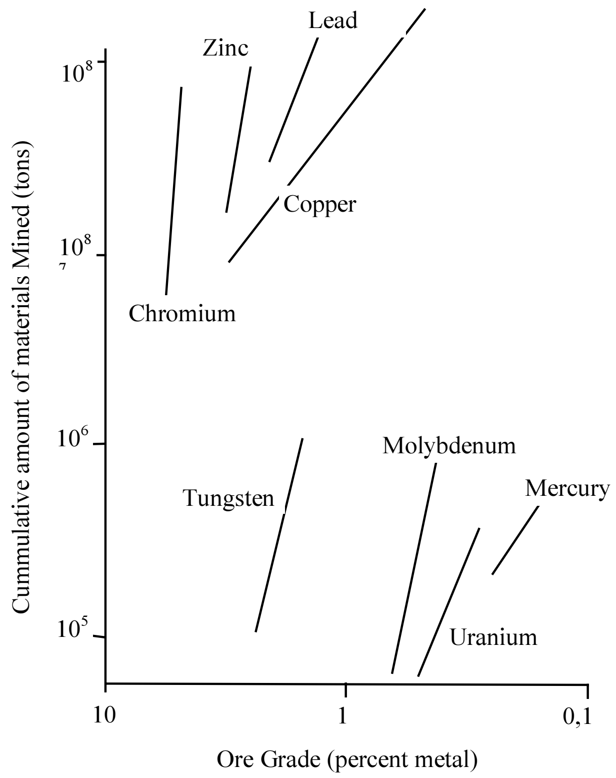

Figure 6.2: Slope of the availability against grade. Curve taken from [CHAPMAN AND ROBERTS 1983]

6.3.2. Fossil fuels

The formation of fossil resources has occurred on a completely different time scale, trough completely different processes. A brief description of these processes can be found in box 6.4. and 6.5.

For fossil fuels the term "concentration" is not a very good indicator for the resource quality. The processes that have produced and distributed the fossil fuels are quite different from the processes that have caused the lognormal distribution in the earth crust. This means that the log normal distribution of resource concentration is not directly applicable to fossil fuels.
Basically we can distinguish three types of fossil fuels. These three types can be distinguished in a number of sub categories:

	sub category	
1 Oil	1.1 Conventional oil	1.1.1 All currently produced oil, that easily flows out of large wells
	1.2 Unconventional oil	1.2.1 Tar sands 1.2.2 Shale 1.2.3 Secondary oil (produced from existing wells with steam injection) 1.2.4 Tertiary oil (oil from infill drilling, reaching pockets that were originally bypassed)
2 Gas	2.1 Conventional gas	2.1.1 Wet gas, associated with an oil accumulation 2.1.2 Dry gas, unrelated to oil fields
	2.2 Unconventional gas	2.2.1 Natural gas liquids (condensed gas) 2.2.2 Gas from coal-beds 2.2.3 Gas from tight reservoirs 2.2.4 Others, like mantle gas from deep in the earth crust 2.2.5 Hydrates: Gas in ice-like solid concentrations in oceans and polar regions
3 Coal	3.1 Conventional coal	3.1.1 Open pit mining (Hard coal or Lignite) 3.1.1 Underground mining

Table 6.2. Overview of fossil resources

This brief overview demonstrates that apart from the conventional sources there are several alternative (unconventional) sources for oil and gas. Like in the case of the minerals, until now only the conventional sources are used, as these can be extracted with the least effort.

Quite unlike in the case of minerals, the effort of exploiting a resource does not decrease gradually when the resource is extracted. As long as sufficient conventional oil can be found, the effort to extract the resource does not increase significantly, as long as the oil keeps flowing.
Only when conventional resources become really scarce, mankind will have to start to explore unconventional resources. In this case the effort to extract the resource does increase. In the example of oil this could mean that additional drilling and pumping or even steam injection is needed.
So instead of a continuous decrease of the resource quality, we can observe a stepwise resource decrease, while between these steps the effort to extract is basically constant.

Box 6.4. Conventional oil and gas

Conventional oil (and gas) has been formed during certain distinct periods in distinct places. For instance the huge oil resources in the Middle East, the North Sea and Siberia were formed in the late Jurassic, some 150 million years ago. Another period was the Cretaceous, some 90 million years ago, which was responsible for the formation of oil in Northern South America. The Oil in North America dates from the Permian, some 230 million years ago. Oil and gas usually formed in shallow seas or lakes in areas around the tropics. Stagnant sinkholes and lagoons where perfect places to preserve organic material. Later, When such sinks were covered by a layer of sediment or salt and the temperature and pressure had increased sufficiently, all the conditions to form deposits were present.
During the formation of oil and gas, the pressure increases and usually the oil migrates upwards through the pores in the sediment into porous and permeable layers until it encounters a non permeable layer, under which it is trapped. According to [CAMPBELL 1998], only a fraction (about 1%) of the oil is trapped in oilfields that are large enough to exploit. The older the deposit the more has leaked. Especially gas will get lost through leaking very easily.

Unlike the formation of minerals, the formation of fossil resources can be deducted from our knowledge of the plate tectonics the climate changes and other processes that occurred the last half a billion years of the earth history. In global terms the bulk of oil and gas occurs in a geological "province" called the Thetys; a zone of rifting between the southern and the Northern continents, of which the Middle East, the Mediterranean and Mexico are remnants.
The detailed geological mapping has revealed where suitable formations, under which oil could have been trapped, are located. Because of this understanding, we can conclude that the world has now been so extensively explored, that all the large oil resources have been found and the scope for finding an entirely new one of any size is now greatly reduced, if not entirely removed [CAMPBELL 1998]. This can also be demonstrated by the next two figures:

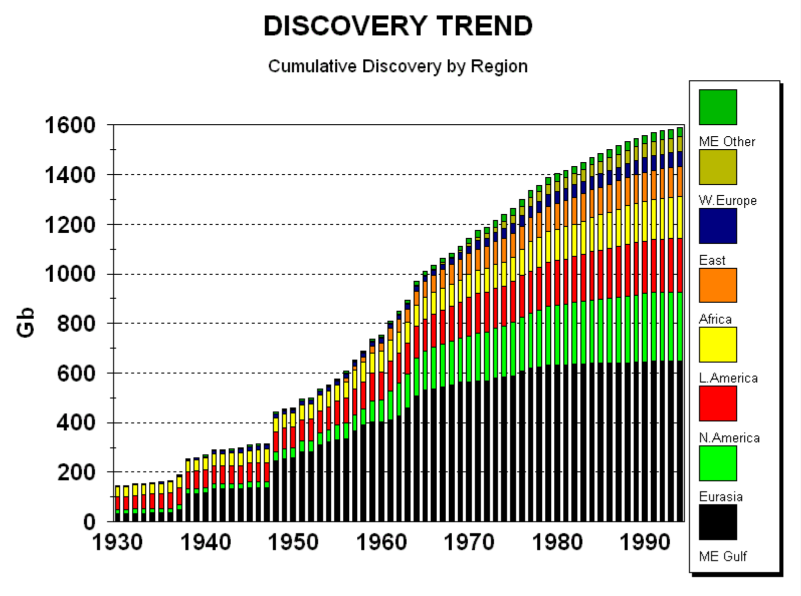

Figure 6.3: Cumulative discovery trend for oil. Clearly the discoveries are gradually becoming smaller, and the availability is reaching its maximum. [CAMPBELL 1998]

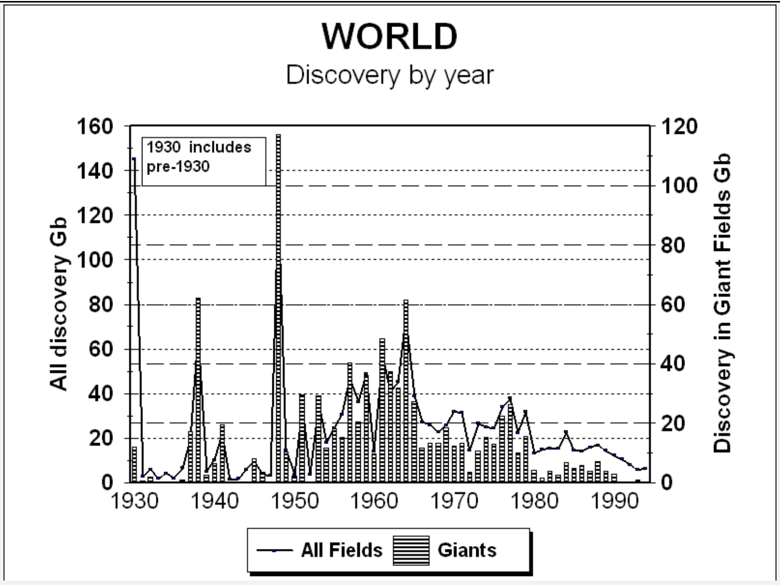

Figure 6.4: This figure clearly shows that since the seventies no new major recoveries were made. At present the discovery rate is about 6 Gigabarrel, while the production rate is about tenfold. [CAMPBELL 1998].

Figure 6.3 and 6.4 show the problem with conventional oil in a nutshell. The last two decades the discovery rate is 6 to 10 Gigabarrel, while the consumption is in the order of 60 to 70 Gigabarrel.

In 1956 Hubbert proposed a prediction technique that is based on the observation that in any region, the unrestrained extraction of a finite resource rises along a bell shaped curve, that peaks when about half the resource has gone [CAMPBELL and LAHERRÈRE, 1998] and [KESSLER 1994]. With this model Campbell has surprisingly accurate modelled the increase and decline of the US oil production and to some extent also the world oil production, as can be observed from figure 6.4. According to Campbell this means that the peak in the world oil production may be expected around 2010. Also the International Energy Agency has adopted this view as can be seen from figure 6.5. [IEA 1998]

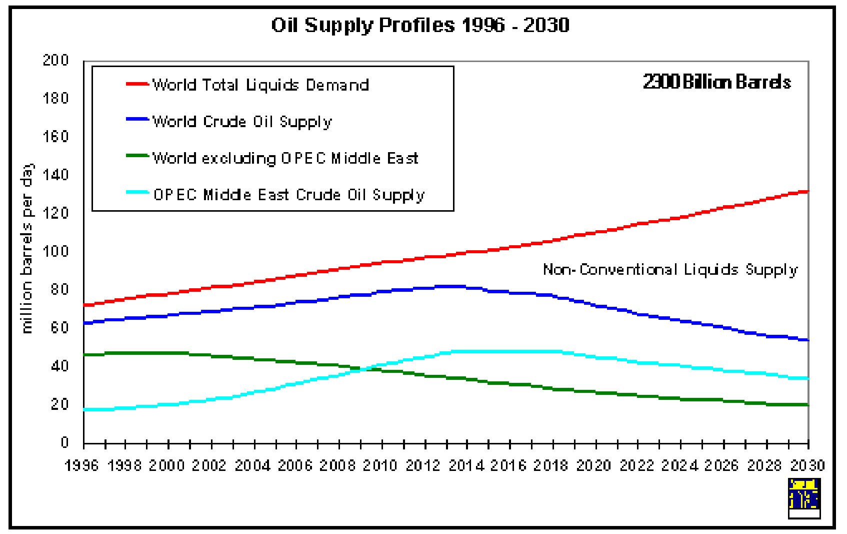

Figure 6.5: International Energy Agency projections for the future oil supply for the G8 countries. This graph clearly shows that the conventional oil supply will reach its peak around 2013, as around that time the OPEC production capacity will start to decrease. [IEA 1998]

As figure 6.5 shows, the transition from conventional to unconventional oil will be a gradual transition. However, the International Energy Agency recognises that around the peak of the conventional oil production, there will be a big shift in market prices, as from that time on the marginal oil production will be the unconventional oil. Also we will assume that at that moment we will assume oil is extracted from the unconventional sources.

Box 6.5. Coal

The biologic and geologic processes that produced coal resources are different from the processes that were responsible for the formation of oil and gas. In the first place, coal was not mainly formed from biological processes in aquatic conditions, such as in the case of oil and gas. Coal formation started with the build up of peat layers mainly from land-based vegetation such as woods, reeds and other plants. These formations occurred in a period between 250 and 50 million years ago in a much wider geographical area than the Thetys area that was the main place for oil and gas formation. This is one reason why coal resources are spread over a wider region than oil, although most coal is found in Europe, North America, the former USSR and China. During the formation of the coal or lignite layer light hydrogen rich substances escaped from the layers and sometimes created gas deposits. As a result, coal has a high carbon content. In general the older layers have the highest carbon contents. Lignite is usually a relative young deposit with a carbon content of around 60 to 70%.
The resource availability for coal is much higher than for conventional oil or gas. The proven resources should be sufficient for about 200 to 300 years, if the present extraction rate is sustained, and if no major discoveries are made.

6.4. Damage to Resources caused by depletion of minerals and fossil fuels

Unlike the damage categories Human Health and Ecosystem Quality we have not found a more or less accepted unit to express damages to Resources. If the resource quality decreases, economic factors and environmental burdens associated with mining low grade ores will become the real problem. The latter includes the land-use for the mining operation and the amount of energy to extract the resource from the low-grade ore. The availability of land and energy could thus form the real limitations and land-use and energy use will probably be the most important factors. This is the basis for the proposal of [BLONK 1996].
When we look at alternative energy resources, another additional option is to translate increased energy consumption into increased future land use, as most non-fossil energy sources use a relatively large area. [ROS 1993] proposes some land-use values for the most important solar and wind based technologies.

[MÜLLER-WENK 1998-1] follows this line of thought. However, instead of land use, he uses the concept of *surplus energy*. The surplus energy is defined as the difference between the energy needed to extract a resource now and at some point in the future. He calculates the future surplus energy at Q*N, in which Q represents the total amount that has been extracted by mankind before 1990 and N represents the number of times this amount is extracted. Müller-Wenk uses N=5 and N=10[28].

The choice of the factor 5 is arbitrary, We could also have selected the point on the damage curve at 10*Q or 2*Q, as we assume the damage curve is linear. The consequence of this arbitrary choice is that the absolute value of the surplus energy has no significance. The only purpose of the surplus energy concept is to have a relative measure for the damage the depletion of a mineral or fossil resources creates. In a way the surplus energy is used as a characterisation method, since the choice of N is only used as a reference. [29]
As we will see later in the damage assessment, the lack of absolute meaning of the damage to Resources does create some problems in the presentation of questions to the panel.

6.4.1 Surplus energy for minerals

[CHAPMAN AND ROBERTS 1983] analyses the relation between energy use and the lowering of ore grades for the most common minerals. Chapman states there are three effects:

1. The amount of energy needed to change the chemical bonds in which the mineral is found is by definition constant. It is not possible to reduce this energy requirement by efficiency improvements or technological developments.
2. The energy requirements needed to extract, grind and purify an ore goes up as the grade goes down
3. The energy requirements needed to extract, grind and purify an ore goes down with efficiency increases and technological developments.

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[28] [Müller-Wenk 1998-1] shows that even at Q*10, the total surplus energy requirements are in the order of magnitude of 1% of the current energy use. This would indicate that the availability of energy may not become a very serious limitation.

[29] Müller-Wenk does use the absolute value when he combines the surplus energy concept with the availability of energy due to increases in energy efficiency. We do not follow this approach; see [MÜLLER-WENK 1998-1 Müller-Wenk 1999]

Chapman shows convincingly that until now the 3rd mechanism is stronger than the second. This means that although the grade of all ores decreases, historically the energy requirements also decrease. Chapman shows that this trend will continue many decades from now. In the case of copper we can extract about 100 times more than mankind has done so far before the actual energy requirements get higher than the present values. For most other metals the situation is even better.

Future efficiency increases are not taken into account in LCA. This is consistent with the other damage models. For instance we do not take into account the possibility that the treatment methods of cancer will be improved, when we look at long term exposure. It is also common practice in LCA not to take possible remediation technologies into account.

6.4.2 Surplus energy for fossil fuels

With the descriptions of the typical characteristics of the fossil resources in the resource analysis and with the data on the increased extraction energy for non-conventional resources, we can begin to construct the model for the surplus energy. However, in the case of fossil fuels we need to discuss two specific problems:

1) The discontinuous or stepwise character of the quality decrease for fossil resources
2) The possibility of substitution between fossil resources

In the case of minerals, we could assume that the decrease of mineral resource concentrations is almost a straight and continuous line. In the case of oil and gas extraction, we are faced with the problem that the extraction will cause rather abrupt steps in the resource quality, when the marginal production of oil and gas switches from conventional to unconventional resources.

In an annexe to his earlier report [Müller-Wenk 1999] calculated surplus energy for minerals at a time when the extraction has reached a quantity equal to 5 times the historical extraction before 1990 (5Q). For fossil fuels he demonstrates that at a similar point (5Q0, there will be no gas and oil, but only coal, shale and tar sands.

In mineral resource analysis we did not take substitution between resources into account, as we stated that the possibilities for substitution are dependent on future changes in demand and technology development. In the case of fossil fuels the possibility for substitution are much more logical to assume, as all the fossil fuels share the same essential property, that is that they supply energy. It is even possible to produce an oil replacement from coal. This is in contrast with the case of minerals: we cannot say that mercury and iron have the same essential properties. Again we have two alternatives here:

a) We assume full substitution, and argue that the future energy mix will be a combination of oil and shale, possibly including the need to convert some coal to a liquid oil replacement for transport equipment. As a result we will calculate a single score for surplus energy, per MJ of fossil energy, independent of the source of energy.
b) We do not assume full substitution, and assume unconventional gas and oil or shale will replace conventional gas and oil, while coal will basically be extracted as it is now. As a result the extraction of coal will be seen as having a low surplus energy, while the extraction of oil and gas will get a high surplus energy score[30].

In case we assume full substitution, we are faced with the problem that we will have to assume a future fuel mix, as without such a mix, it is not possible to calculate the future surplus energy.

Müller-Wenk argues that it is fair to assume that, in the case of substitution, about 50% of the fuel will be a liquid, as such fuels are easy to handle and transport. He also shows that coal liquefaction is very energy intensive; about 50% of the energy produced is lost. He therefore argues that it is not likely that in a future energy mix coal will be converted on a large scale[31]. As a result, coal will have a share of less than 50%. Therefore he proposes to assume an energy mix of 50% shale, and 50% coal. He also assumes that coal mining will be mainly practised in a mode that has the same approximate energy

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[30] The usually much higher emissions from the use of coal are expressed in other impact categories. Here we only consider the valuation of the reduction of resource quality.

[31] Coal liquefaction has been developed and used mainly in South Africa during the time an oil embargo was imposed.

requirements as underground mining of hard coal or mining of lignite. Müller-Wenk does not include the foreseen increase of alternative energy sources within this future energy mix.

6.5. The role of cultural perspectives

Resource depletion is a typical subject in which cultural perspectives can lead to different approaches. table 2.1 in chapter 2 summarises the most relevant views of the three archetypes

The most direct observation is under "energy future"; Individualists, do not consider fossil fuel resources as a problem, and they advocate a business as usual attitude. Furthermore, Individualists would argue, that based on experience (especially after the so-called oil crisis) fossil fuel depletion is not really an issue. Furthermore, as the long time perspective is not relevant for him, he would not give much weight to future problems that might occur.
In the case of mineral resources we assume that the Individualist would accept the steady decrease of resource concentrations as a fact. Therefore we propose to exclude the whole issue of fossil fuels, but to include mineral resource depletion for the Individualist perspective. One could argue that it is not consistent to use surplus energy as a basis for characterisation of mineral resources, while the depletion of fossil fuels is not an issue. However, also to Individualists, an increased use of energy will be regarded as a problem, although perhaps a minor one.

For the other perspectives mineral and fossil resources are considered to be a serious problem. The only difference of opinion is in the assumption of substitution between fossil resources.

Egalitarians have a different conceptual difference with the Hierarchists in the "View of needs and resources". Egalitarians assume resources cannot be managed, while needs can. For them it is logical to assume substitution, as they are not really interested in the differences between resources; fossil fuels are all belonging to the same "unmanageable' group of resources. For them it is important to implement a need reducing strategy for fossil fuels as a group, for instance by stimulating alternative energy sources. Hierarchists assume needs cannot be managed, but resources can. For them it is important to look carefully at the differences between the resources in order to develop management strategies.

Current fuel	**Hierarchists**	**Egalitarians**	**Individualists**
Conventional natural gas	oil shale	coal-shale mix	conventional gas
Conventional oil,	oil shale	coal-shale mix	conventional oil
Hard coal, open pit mining	brown coal	coal-shale mix	hard coal, open pit
Crude oil, secondary extraction	brown coal	coal-shale mix	oil, secondary
Hard coal, underground mining	brown coal	coal-shale mix	hard coal, underground
Brown coal, open pit mining	brown coal	coal-shale mix	brown coal, open pit
Crude oil, tertiary extraction	oil shale	coal-shale mix	oil, tertiary
Crude oil from oil shale	oil shale	oil shale	oil shale
Crude oil from tar sand	tar sand	tar sand	tar sand

Table 6.3: Assumed fossil fuels that will replace the current fuels for the three archetypes. The surplus energy calculation is based on this table. The coal-shale mix is the assumed future energy mix for the Egalitarian perspective.

Although Egalitarians are very much in favour of eliminating the need for fossil resources, we cannot assume this to happen in their perspective. The main reason is that also other societal aspects, such as the distribution of wealth are important. As long as alternative energy sources are more costly they would argue that fossil fuels could not be excluded[32].

Based on these characteristics, we propose to use assume substitution for Egalitarians, while for Hierarchists we do not assume substitution.

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[32] For egalitarians the availability of coal for developing countries has an important meaning, as this resource is cheap and distributed relatively evenly over the world. If we assume substitution, the surplus energy for coal would be higher, which would mean that coal resources are better protected.

Table 6.4 shows the consequences of this choice. The surplus energy can now be calculated by subtracting the current energy requirement for a fuel from the energy requirement for the replacing fuel or mix. Tar-sand and oil shale are special cases, as for the time being we would not like to consider what the next fossil resource would be. As both types of fuels will not be encountered in most LCAs we choose to assume these resources replace themselves. The net result is that the surplus energy is zero.

6.6. Sources of uncertainty

The uncertainties in resource models are considerable for minerals. It appears that there are very significant differences of opinion between the two sources used by [Müller-Wenk 98-1] for some minerals, while for other minerals there is a considerable level of agreement.

For fossil resources there are not very big uncertainties, except for the important choices made in the previous paragraph. This is because the energy requirements for oil shale and coal are well known for the present level of technology, and because there is a reasonable high certainty that there will be sufficient resources when 5 times the present resource extraction is extracted.
The Egalitarian perspective has a big uncertainty in the assumption of the future coal-shale mix.

7. Damage assessment

7.1. Description of the problem

The calculation of the total scores for the three damage categories concludes the environmental modelling. In some cases these results provide sufficient information and further steps are not needed. Such situations occur if all damage scores for product A are higher than for product B, or if the predefined aim of a study was limited to analyse only one damage category.

In the ISO 14042 documents, the normalisation and weighting steps are referred to as "optional elements". In fact, the documents do not only refer to normalisation and weighting, but also to ranking. Ranking is a procedure in which the impact categories are put in an order of descending importance. In some applications, especially in the case of comparative assertions, disclosed to the public, the weighting step is not allowed; however, ranking is allowed in this case.

It is up to the user of this methodology to decide if and how the damage assessment is applied, and if it is necessary to follow or deviate from the ISO standards. In any case, we encourage users to critically analyse the default weighting factors presented in this project and to propose other factors. However as we aim to produce single scores, to be used for designers that are usually not in the position to develop their own weighting factors we need to develop the damage assessment here to provide a default weighting set.

Before the weighting step is performed a normalisation procedure takes place. In the normalisation step the results of the damage calculations are divided by a reference (“normal”) value. The result of this step is a set of factors that have the same (or no) dimension, and that reflect the relative share of the calculated damages to the reference.
It is clear that the reference needs to be carefully defined. A new feature in this methodology is that we can actually give a meaning to the normalisation factors for Human Health and Ecosystem Quality. This is because many studies have quantified the actual damage caused by environmental pollution. Some even have expressed those damages as DALYs or a definition that is closely related to the concept of PDF. As we will see, this aspect creates some particular questions when we want to interpret the results of the panel study.

Weighting factors are by definition normative and cannot be true or false. The only judgement can be if they are properly reflecting the views of the appropriate stakeholders. Basically this means there are two quality criteria for weighting factors:

- The stakeholder group is properly defined and represented in the methodology used.
- The methodology and procedure is designed, performed and interpreted in a carefully conducted consistent and scientifically valid procedure. The term scientific refers here to the social sciences.

Box 7.1. The triangle concept as alternative to fixed weights

For those users that do not want to use the default weighting factors, we recommend using the mixing triangle developed by [HOFSTETTER 1999]. This triangle (see figure 7.1) can be used to graphically depict the outcome of product comparisons for all possible weighting sets. Each point within the triangle represents a combination of weights that add up to a 100%.

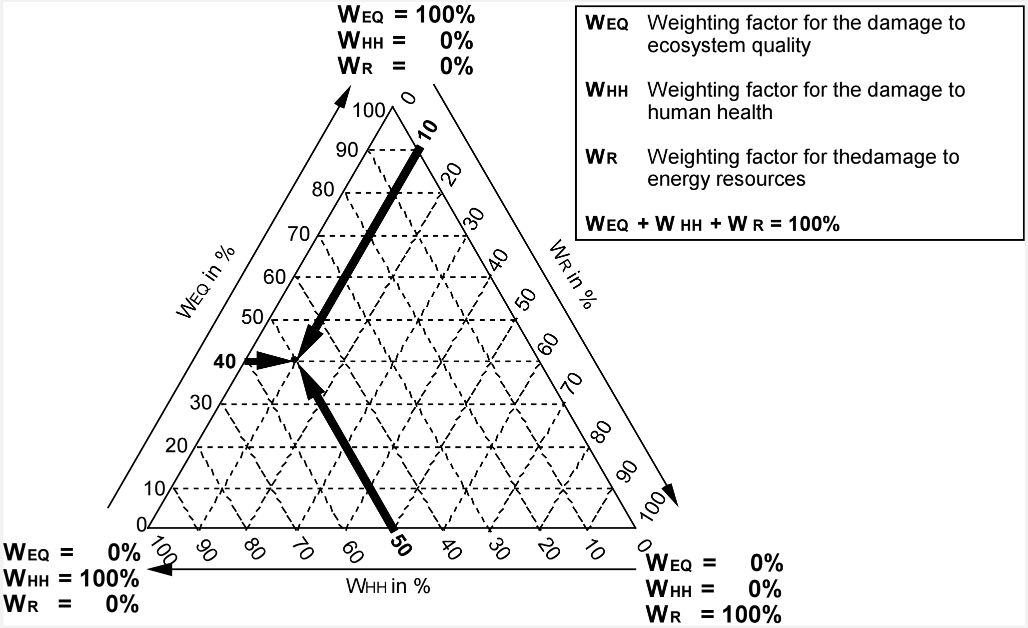

Figure 7.1: The mixing triangle: The marked weighting point is positioned where Human Health is weighted 50%, Ecosystem Quality 40% and energy Resources 10%. The point is defined by following each side until the dotted flashes leave towards the point in the triangle (based on Hofstetter 1998).

A key feature is the possibility to draw lines of indifference (figure 7.2). These are lines representing weighting factors for which product A and B have the same environmental loads. The lines of indifference divides the triangle into areas of weighting sets for which product A is favourable to product B and vice versa.
The weighting triangle can display the result of an LCA without knowing the weighting factors. According to Hofstetter, such a representation is a very useful tool to enhance the transparency of the weighting process, as it shows under which conditions (which weighting factors) product A is better than B. The stakeholders do not have to set discrete weights, but they have to agree whether it is plausible that the weights would fulfil the conditions under which A is better than B or not. Such a discussion process turns LCA into a consensus building process, instead of a tool that produces simple single truths.
Our mission is not the development of a consensus building tool, but there is no reason not to use this methodology as such, if the conditions facilitate an open discussion with the stakeholders. More information on this subject can be found in [HOFSTETTER 1999]

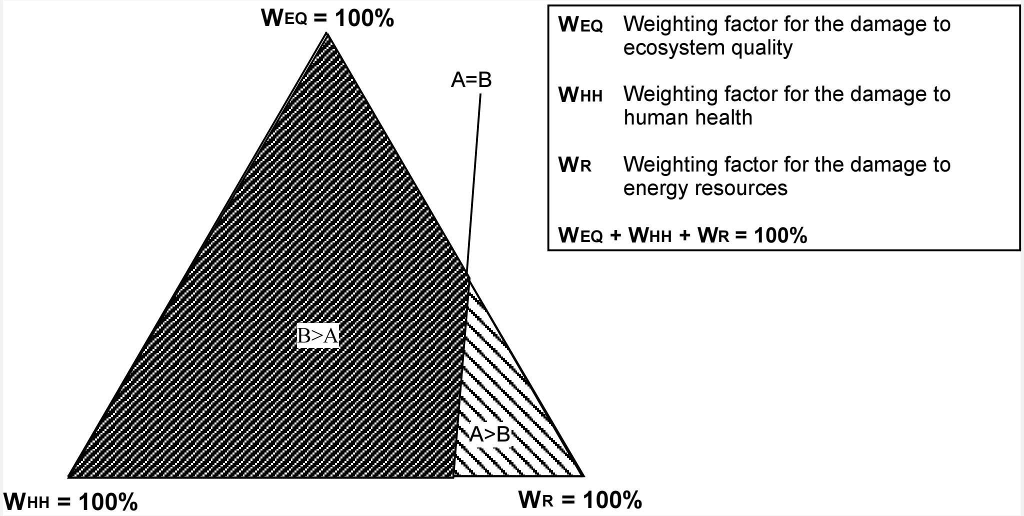

Figure 7.2: The line of indifference in the weighting triangle and the sub-areas with their specific ranking orders (B>A means that alternative B is environmentally superior to A and the eco-index A is higher than B).

7.2. Normalisation

7.2.1. The procedure for normalisation

In normalisation, we determine the relative contribution of the calculated damages to the total damage caused by a reference system. The purpose of normalisation can be manifold, such as:

1. Error checking. By comparing the impact category result with a reference, one could check if the results have the correct order of magnitude.
2. Providing information on the relative significance of the impact category results.
3. Preparing for additional procedures such as grouping and weighting.

The last application is clearly the most relevant in this project.

The reference system can be chosen in many ways. Usually the reference system is the sum of all emissions and all resource extractions in the world or in a given part of the world, during a certain time. For instance [BLONK ET AL 1997] describe normalisation values on the level of the European and Dutch geographical area, as well on the level of the total consumption of the average consumer. In each case the period is a year, and in each case the results are divided by the number of inhabitants of the region.

The calculation of normalisation values consists of two steps:

1. Find the total emissions and resource consumption caused by the reference system during a reference period (usually a year)
2. Calculate the impact categories and, if applicable, the damage scores, using the characterisation and damage factors.

If desired, the results can be divided by the number of inhabitants, to get personal equivalents. The main reason for that is pragmatic; the usually very high normalisation values get a value that is easier to relate to when personal equivalents are calculated. Furthermore it becomes easier to compare normalisation sets from different regions.

[HAUSCHILD AND WENZEL 1998] also use this division by inhabitants, as he differentiates between the scale of the normalisation area. Global impact categories are normalised by global average personal equivalents, while regional impact categories are normalised by regional average personal equivalents. The use of such differentiated normalisation basis is not compatible with the marginal damage model we use.

Box 7.2. Marginal normalisation and weighting

A unique property of the methodology described here is that the normalisation values do have a distinct meaning on themselves. For instance, the result for Human Health is a number of DALYs/yr. It seems tempting to compare this with other, independent studies. For instance [HOFSTETTER 1998] estimates the number of DALYs caused by environmental problems is presently 10% in Europe. The percentage of endangered plant species in Europe is about 10 to 25%. These damage levels are referred to as the “actual” damages. For Resources we do not have an “actual” damage level. The absolute value of the surplus energy is directly dependent on the time perspective, and therefore has no real meaning. [Müller-Wenk 1998-1] shows that when the cumulative extraction of the resources reaches a level of 5 times the cumulative extraction until 1990, the surplus energy for minerals will be in the order of 1% of the current total energy use. However, if we replace the factor 5 by ten theis figure changes too.

Although this is very useful to have such data as a reference that can be presented to the weighting panel (see box 7.5), these values cannot be used as a basis for normalisation, as there is a distinct difference between the actual damages, and the damages that are calculated according to the two step procedure:

When we apply the traditional two step procedure to calculate normalisation values, we use a marginal approach to calculate normalisation values. This mismatch can be explained as follows. Our damage modelling is explicitly based on the slope of the damage curve, and not on the (often non-linear) total curve. This means our models are linear (and marginal), while for the calculation of the “actual” current damage, we need the original damage curves. The two step procedure can thus never result in the actual; damage. The two step procedure in combination with an average personal equivalent produces a normalisation value that should be understood as the marginal effect of a personal equivalent.

There is a difference between the dynamic and steady state modelling. The actual damage is the result of a combined effect of all historic emissions and land-use changes, as well as resource extractions. In the past emission levels and other environmental interventions have been fluctuating. With the two step approach such fluctuations are not included. The calculated damage would occur if the emissions in the reference year would continue indefinitely. For instance a part of the near extinction of plant species could be caused by the use of pesticides that already have been forbidden. This means that the actual level may be higher than the current emissions would indicate. Likewise, the actual level of damage to the ozone layer is due to the high levels of CFC emissions in the previous decades. On the other hand the CO2 emission level is steadily rising while there is a considerable time lag between the emission and the occurrence of damage. This would mean that the current damage created by climate change is much lower, as it lags behind, than the current emissions would indicate if we were to use the two step procedure.

The fundamental question arises, which level of normalisation is to be used. We can look at this question from two perspectives:
If normalisation is seen as a procedure to give information on the significance of the impact category results, the traditional two step procedure should be used. The reasoning is that the normalisation values are calculated in the same way as the results for an LCA: *The marginal steady-state damage from a product system is normalised by the marginal steady state damage of an average personal equivalent.*
If normalisation is seen as a procedure to prepare for the weighting step, the situation is far less clear. The problem is that the panel bases its values to at least some extent on the perception of the current levels of damage. The panel members get information on the damage levels from all kind of sources, such as media and from their own observations, when walking in natural areas, discussing health problems of friends or discussing fuel prizes. From this perspective actual normalisation values should be used. *The marginal steady state damage from a product system is normalised by actual (non steady state) damages.*

Both versions are to some extent inconsistent. The two step procedure is more consistent when we disregard the weighting step, as the normalisation values are calculated in the same way as the effect scores. The second approach is more consistent when we look at how a panel regards the current seriousness of damages. Although this is still a debate among the developing team, the first approach (the traditional two step procedure) is used here. This means the normalisation values used in this project are marginal values.

7.2.2. Available data for the two step procedure

The calculation of normalisation values has been carried out using the data on resource extraction and emissions, which have been collected previously in a normalisation study carried out for the Dutch ministry of transport and public works and the Dutch ministry of Housing, Spatial planning and the Environment.[BLONK ET AL 1997]
In this study data have been collected for the purpose of developing normalisation values for the Dutch territory and the European territory. Normalisation values are mostly based on environmental interventions resulting from European production in1990-1994. Although these data are based on relatively old original data (mainly before 1993), there has been surprisingly little activity since this report has been published.

The Blonk report acknowledges some important weaknesses in the emission data. On the European level only little data on environmental interventions from individual substances were available. Most available emission data were aggregated in substance groups so the data had to be edited and extrapolated to the level of individual substances. The most important adaptations carried out for the determination of normalisation values for Europe is the estimation of lacking data from most countries using a rather coarse extrapolation method based on energy use. This extrapolation procedure was developed for the Eco-indicator 95 methodology [GOEDKOOP 1995] and was also used in [BLONK ET AL 1997]. The extrapolation procedure is described in more detail in the annex report.

The data for bulk air emissions in the Blonk report are derived from [CORINAIR 1990]. These data have been updated for the Eco-indicator 99 using the new Corinair Inventory data [EEA 1998]. For emissions to air of heavy metals and some persistent organic chemicals some new data were available also, making the estimates for these substances more reliable.
A new estimation for the emissions of CFC's has been calculated using data on world production and emission from [AFEAS 1999].
European data on emissions to water and soil are not available. Data are extrapolated from the Netherlands [ER 1996]. Land-use was not included in [BLONK ET AL 1997] and this means that additional data for land-use have been collected for Eco-indicator 99.

7.2.3. Sources of uncertainty

Quantitative uncertainty estimates for the normalisation data are not available. The uncertainty has been described qualitatively in [BLONK ET AL 1997]. In some cases the normalisation has improved:

- For bulk emissions to air no extrapolations were necessary, since data were available from all countries. New data have been used.
- For emissions of heavy metals and persistent organic chemicals to air more data were available, thus reliability has improved, since only part of the emissions had to be extrapolated.
- For emissions of ozone layer depleting substances to air, heavy metals and pesticides to water and soil, almost all emissions are extrapolated. These data still contain large uncertainties.

7.3. Weighting

Box 7.3. The panel and the revealed preference approach

As stated before, weighting is a purely normative step. In this step we assign weighting factors to the normalised results. These weights should represent the views of society or a group of stakeholders. For our project the weights should, as far as possible, reflect the views of the European society, as the Eco-indicator is intended to be used for emissions and land-use changes in Europe.

There are basically two ways to get information on the views of a society:
Observation of the actual choices being made by society in other, but related situations. This is often referred to as a *revealed preference* approach. Revealed preference methods are usually based on targets set by governmental organisations or at costs society appears to be prepared to make.
Direct questioning of a representative group in the society. In many cases such a group is referred to as a panel. We refer to this as the *panel approach*.

A typical example of the revealed preference method is the EPS system developed by [STEEN 1992]. He assigns monetary values to 5 endpoints (called safeguard subjects). Without wanting to oversimplify the careful analysis made by [STEEN 1992] we mention that the value of an excess death is based on data from US life insurance companies, while the value of maintaining biodiversity is based on expenditures made by governments to protect certain species from extinction.

We do not use the revealed preference method, mainly for three reasons:
It is often hard to isolate and interpret the basic values that are underlying the decisions of society. Very often the decisions made are complex and in very few cases a single issue is at stake. For instance policy targets set by governments are often a compromise between the need to reduce loads and the preparedness to make the necessary sacrifices.
The use of revealed preference causes problems if we want to include a normalisation step, as after normalisation, the results have lost their unit[33]. There are two ways to solve this, but neither solution is easy or satisfactory:
Include normalisation, and assess the total expenditure society makes for very broadly defined issues, such as protecting Human Health, Ecosystem Quality and Resources. Such an estimate will be very difficult to make. As far as we know, no targets set by governmental organisations for these issues are available.
Skip normalisation and convert the damages directly to monetary values. Although we feel we might be able to get some realistic data on the value of DALYs, we do not think it is possible to give a meaningful valuation to the threatening of plant species or the future surplus energy. This solution has a high price, as normalisation is seen by many as a very useful step providing much additional information without needing to weight.

For these reasons we opt for the panel approach, although we recognise it is clearly not easy to get relevant information from a panel, for a number of reasons:
The basic questions on the importance of Human Health, compared to Resources or Ecosystem Quality are abstract. Many ordinary citizens will not be able to understand these questions, let alone answer these questions.
The exact phrasing of such questions is very important, as this can influence the perception of the questions.
Experiences from other experiments show that there is a wide distribution in the answers, and the meaning of an average value in such a distribution is limited.
There are difficulties in selecting the proper normalisation basis.

A unique property of the methodology developed here is that only three weighting factors are required. In other Life Cycle Impact Assessment methodologies weights had to be generated for 10 or more impact categories. It is clear that assigning so many weights at the same time can give serious cognitive stress for those who are asked to give such weights. Another clear advantage is the possibility to use the weighting triangle, as this concept would become difficult to use if there were more than three damage categories.

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[33] [STEEN 1992] does not use normalisation either

Another important advantage assigning weights to damage categories instead of impact categories, is that the damage categories are much easier to understand. Giving weights to greenhouse effect or acidification requires a great deal of background knowledge on the mechanisms, the effects, the probabilities that the effects will occur, and the way the effects will really cause damage. In the case of the Eco-indicator 99, the panellists can concentrate on the question of the seriousness of the damage. They are asked to give values based on their views, and not on their knowledge. This is essential.

7.3.1. The panel procedure

The panel procedure used in this project was designed and executed for our project by Thomas Mettier. Earlier [BRUNNER, 1998] had analysed the problem and some earlier attempts to use a panel for this application.

Box 7.4. Selection of the respondents

Brunner distinguishes between panel methods that are consensus or discursive oriented. He criticises earlier panel experiments, as they have been too much focussed on building consensus. Building consensus cannot be an aim here. We want to get representative information on the values that are in the society. We do not establish a process in which the panellists are to change their opinion in order to get consensus. For this reason we opt for a written questionnaire and not for a discussion process. Another reason for using a written procedure is the possibility to use a large panel.

Brunner lists four factors that are important for the selection of the panel:
1. The intended representativeness. Clearly a panel consisting of industry officials will not be representative for the whole society.
2. Understanding of the models and terms used. This is clearly a big problem, as the concept of LCA and certainly the concept of the damages is abstract and hard to understand
3. The expected return rate. The panellist must somehow feel motivated to collaborate.
4. Selective return. It may be that the shares of people that do not respond are not equally distributed over the represented groups in the panel.

These criteria were used by Brunner to tentatively characterise some alternative panel compositions. Based on these criteria we have decided to use a panel consisting of LCA experts and LCA users. We are aware of the fact that these are not really representative for society, but their level of understanding and the expected high return rate made this group a favourable choice. As we will see, the real return rate was much lower.

The procedure developed and executed by [METTIER 1999] exists of three steps:
1. Development of a questionnaire. This phase also contained a number of tests, in order to check if the questions were understood in a proper manner.
2. Sending out the questionnaire to 365 respondents. These respondents were all listed, or had been listed as members of the Swiss discussion platform on LCA. The questionnaire was written in German. Members of the Eco-indicator development team were excluded.
3. Analysing the results, and reporting back to the respondents. Unfortunately only 82 questionnaires were returned (22%).

Mettier did not aim for a real statistical representative study of society, for that would need a much larger sample. His aim in this stage of the research was to get statistically significant differences between damage categories. As we will see this aim was reached.

The questionnaire developed by Mettier contained basically five parts
- An introduction, containing a brief description of the purpose, the outline and intended application of the Eco-indicator 99 methodology, and a description of the damage categories. The latter also contained a brief description of the estimated present damage levels in Europe, in terms of the number of DALYs, the percentage of read list plant species and information on surplus energy. This data was not obtained from the normalisation process, but from independent sources. This means they reflect the present and not the steady state damage
- Ranking. Respondents were asked to rank the damage categories in order of decreasing importance (or to state that they were equally important)
- Assigning weights. The respondents were asked to give weights directly. In that process they were also asked to what extent they used the information about the present level of the damages.
- Cultural perspectives. A number of standard questions were used to analyse to what extent the views of the respondents coincided with the basic cultural perspectives.

- Background questions, such as age, sex etc.

In the test Mettier also included some hypothetical LCA results, expressed as three damages, and asked the respondents to state which product would have the lowest overall load. These questions were removed from the questionnaire, as respondents felt they were tested if they had to answer the same basic questions in a different way.

The detailed results are discussed in [METTIER 1999] and summarised in the annexe report but the main results will be illustrated using the mixing triangle concept, discussed above[HOFSTETTER 1999].

7.3.2. Some results of the questionnaire

The triangle will be used to demonstrate the result of the ranking performed by the respondents.

When a respondent states that Human Health (H) is more important than Ecosystem Quality (E) and that Ecosystem Quality is more important than Resources (R), we can interpret this as:

1. Human Health (H) must have a weight higher than 33%, because otherwise E or R would by definition get the highest factor.
2. Resources must have a weight that is lower than 33 %, otherwise it would become higher than either H or E.
3. Ecosystem Quality (E) must have a weight lower than 50%, otherwise it would get higher than H at R=0.

This reasoning can be shown graphically in the triangle as a grey area in figure 7.3

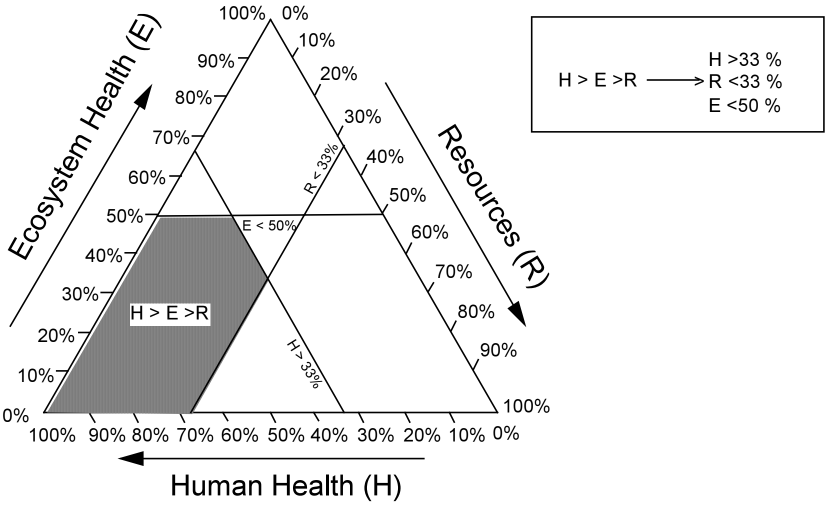


Figure 7.3: Graphical representation of a ranking result. A respondent that ranks the damage categories as Human Health first, followed by Ecosystem Quality, followed by Resources, should have a weighting set that fits in the dark grey area.

With this reasoning the areas for all combinations that have been answered by 75 (out of 82) respondents can be plotted. In figure 7.4 all areas are combined in one triangle with the frequency in which an area or part of an area was chosen by the respondents. This figure must be interpreted with care, as the fields overlap. This can be seen as the sum of all figures is about three times as high as the number of people that have answered the question. The figure 52 in the trapezium a little left from the centre does not indicate that 52 respondents choose the weights in this area, but that this area is *included* in the (much larger) area of each respondent.

In spite of this, the picture shows quite clearly that in the ranking Resources get a relatively low weight, while Ecosystem Quality and Human Health get approximately the same weight.

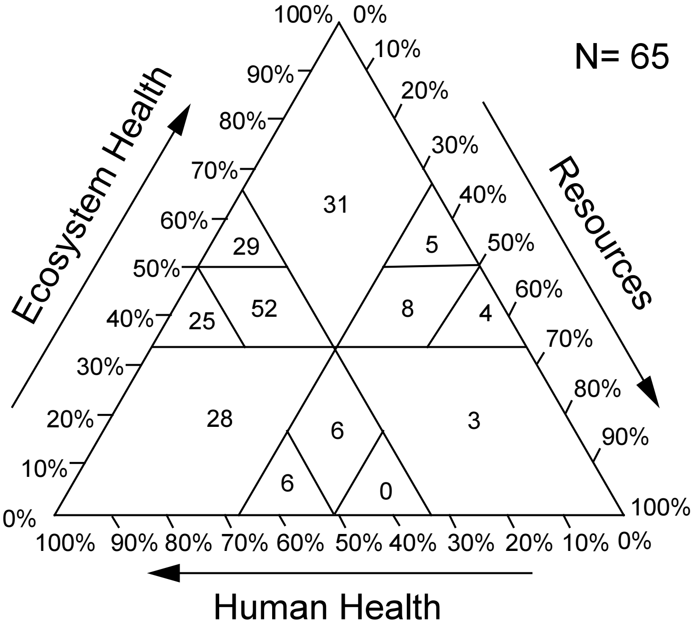


Figure 7.4: Overview of the ranking results of all 77 respondents. The figure must be interpreted with care (see text). It is clear that Human Health and Ecosystem Quality will have higher weights than Resources.

The next set of questions asked respondents to give weights to the damage categories. Only 46 respondents answered this question. The answers are plotted in the triangle in figure 7.5. The large cross in the middle represents 10 crosses of respondents that considered all weights to be equal. Clearly also this figure shows that Resources do not get a very high weight, while Ecosystem Quality and Human Health get a more or less symmetrical but wide ranging set of weights. A closer look reveals that there seem to be clusters of points in the area where Human Health gets around 60% and where Ecosystem Quality get 60 to 70%. It is remarkable how few points are in the corners, only three respondents give weights that are higher than 80%. On the other hand there is quite a large number that gives very low weights (less than 10%) to the damage category Resources.

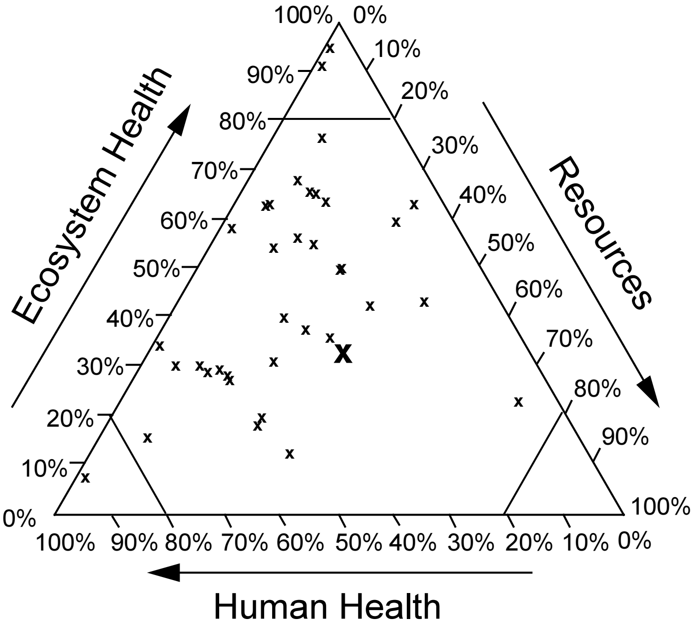


Figure 7.5: Representation of all 46 weighting sets. The large cross in the middle represents 10 respondents that choose to consider all damages of equal importance.

The result of the quantitative section of the questionnaire is summarised in table 7.1.

	Mean	St. Deviation	Median a)
Human Health	36 %	19 %	33%
Ecosystem Quality	43 %	20 %	33%
Resources	21 %	14 %	23%

Table 7.1: Results of quantitative weighting *a) the results do not add to 100%*

The small difference between Human Health and Ecosystem Quality should not be considered to be statistically significant, but the difference with Resources is significant. Because of this we propose the following default weighting set as a first yardstick:

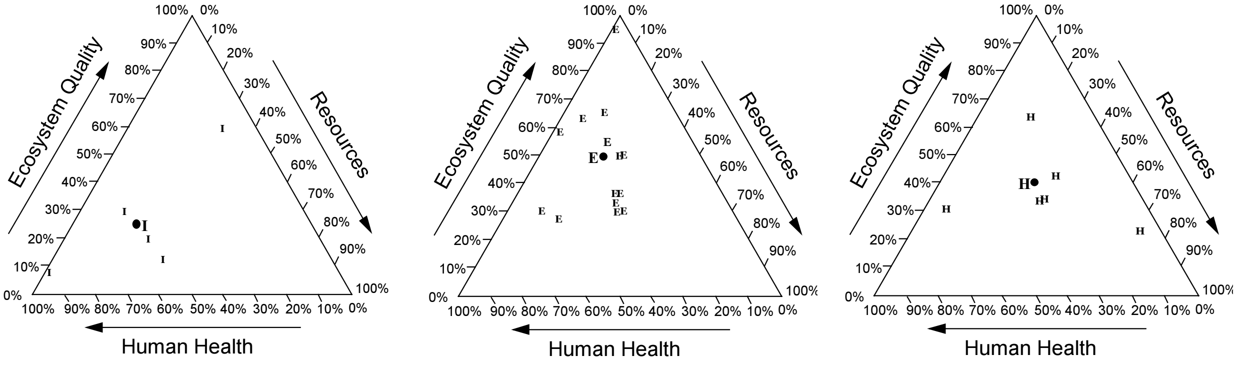
	Average	Standard deviation
Human Health	40%	20%
Ecosystem Quality	40%	20%
Resources	20%	15%

Table 7.2: Default weighting set

7.3.3. The role of cultural perspectives

Mettier also included a set of questions that reveals if and to what extent the respondents adhere to a certain perspective. Furthermore, he included questions that aimed at dividing them into ecocentric and anthropocentric attitudes.
The expected correlation between anthropo- or ecocentric attitudes and their weights for Human Health or Ecosystem Quality turned out to be significant. There is also significant correlation between the ecocentric view and a high weight for Resources.

In total 29 respondents (out of the 46 who gave weights) could be discriminated as adhering to a certain cultural perspective. The Egalitarians and the Hierarchist did not turn out to have statistically significant differences. The differences between Individualists and Egalitarians were significant for Ecosystem Quality and Human Health, but not for Resources. The difference between Individualists and Hierarchists were only significant for ecosystems. The results and the statistical significance are represented in figure 7.6 and summarised in table 7.3.



Individualists Egalitarians Hierarchists

Figure 7.6: Distribution of the weights for the 29 respondents that could be distinguished as adhering to a perspective. The letters represent a weight of a respondent. The dot represents the average

(n=29)	Individualist (n=5)	Egalitarian (n=14)	Hierarchist (n=5)	$p_{E,I}$ a)	$p_{E,H}$	$p_{I,H}$
Ecosystem Quality	25.9%	47.4 %	37.7%	.032*	.285	.289
Human Health	53.0%	32.6 %	30.2%	.041*	*.791*	.049*
Resources	21.0%	20.0 %	32.0%	*.900*	.121	.246

Table 7.3: Differences between the perspectives and their statistical significance
a) Probability that there is no difference between Egalitarians and Individualists

The last three columns indicate the probability that there is no difference between the perspectives. In only three cases (marked with an asterisk) there is sufficient certainty that there is a significant difference. In two other cases (in *Italics*) it is clear that there is a very high probability that the difference is not significant. The remaining 4 cases are less clear.
This information has been used in a procedure to derive at rounded figures that reflect the significant differences. The significant differences are clearly shown. The insignificant differences have been eliminated and the intermediate cases show relatively small differences.

	Average	**Individualist (n=10)**	**Egalitarian (n=14)**	**Hierarchist (n=5)**
Ecosystem Quality	40%	25 %	50 %	40 %
Human Health	40%	55 %	30 %	30 %
Resources	20%	20 %	20 %	30 %

Table 7.4 Estimate of rounded weighting factors per cultural perspective

Box 7.5 Weighting marginal or actual (perceived) damages

An important and not yet resolved discussion in the Eco-indicator development team was whether or not it was appropriate to provide information on the current damage levels in Europe. The objection is that we are not really interested in an assessment of the current damage level, as both our damage model and the normalisation level (see box 7.2) are marginal.
On the other hand it was clear that this information does help the respondents to understand better what the damage categories mean.

The difference between the weights of marginal and actual (perceived) damage can be explained by using the following graph. The graph presents the possible relation between information on the actual damage and the weight given by a panel. The combined weight should add up to 100%. The current (perceived) damage level is supposed to be in A.
The assumption in this graph is that panellist use the actual damage in their assessment of a damage category. This means they do not only give a weight w to the inherent value of for instance Human Health, compared to Ecosystem Quality, but that they let the weight be influenced by the actual (perceived) state of the environment.
For instance in the graph below the situation could be that it appears that 90% of the species are already extinct, while “only” 30% of all DALYs are related to environmental problems. Many panellists would probably argue that, as the damage to ecosystems is already so high, the weight must increase to save the last remaining species.

If this is the case any (marginal) change in the damage level for Ecosystem Quality would be valued higher than the value of the actual (perceived) damage, while for Human Health the weight for marginal damages would be lower. The differences between marginal and actual weights can be seen from the slopes of the weighting curves. Only if the curves are horizontal in A, there should be no difference between the marginal and the actual weights.

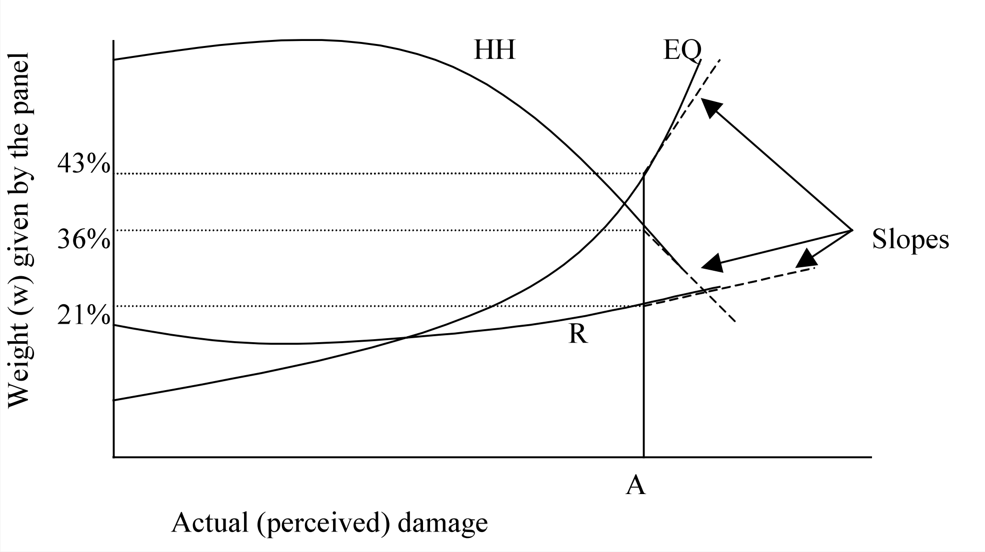


Figure 7.7: Hypothetical relation between the actual damage level and the weights. The actual level could be A.

By supplying data on the actual damage, and by asking panellist to give a weight to the actual damage, basically the result of the procedure should not be interpreted as the pure weight w, but as w*A, in which A is the actual weight.
The question is if this result can be used to value the normalised result of the damage modelling. This normalised result can be expressed as: $\delta D_{FU}/\delta D_{Norm}$, with δD_{FU} representing the marginal damage caused by the functional unit and δD_{Norm} representing the marginal damage caused by the normalisation value (not shown in the graph). When the panel result (A*w) is combined with this marginal result, we get $\delta D_{FU}/\delta D_{Norm}$* A*w. Clearly this is not a dimensionless quantity.

On first sight an improvement is possible by not normalising with a marginal damage, but by an actual damage (see box 7.4). The formula would then be: $\eth D_{FIJ}/A^* A^*w = \eth D_{FIJ}{}^*w$. Also this is not a dimensionless quantity.

The problem is thus that the results of the panel procedure are theoretically not applicable due to the way the questions are formulated. In theory the question should have been to give a weight for a marginal increase of the current damage level, and not on the actual (perceived) damage level. This may look good in theory, but such a question is quite abstract.

In order to see the effect of supplying this information, the respondents were asked to state whether they used the information and, if they did, how this had influenced their weights. 67% answered that they did not use the information, while 22% said it had indeed influenced them. In general they said that Human Health would have had a higher weight or Ecosystem Quality would have had a lower weight if there would be no information. As only a small percentage used the information we can say that the information on the present level of damage had a relatively small impact.

As a result of these answers we propose to assume the results are still very useful. In fact the answers imply that for most respondents the graph plotted above would have horizontal lines. We can interpret their answer as: *No matter the current level or trend in the current level of damage, the weights are as we have specified them.* So we conclude that the answers may be interpreted as a value for w and not (A*w), which is what we need.

7.3.4. Discussion and conclusion.

The panel procedure has not produced a statistically representative result as the panel size and the rate of return was too low. On the other hand we believe that this panel procedure has been carefully conducted and that the results are very acceptable default weights. It is interesting to see that both the ranking and the weighting show that Resources are to be weighted lower than human and Ecosystem Quality.

We have also seen that there are some significant differences between the cultural perspectives, and we have been able to develop separate weighting sets for the perspectives. However, as the sample size for the perspectives was very low, the validity of the weights per perspective is questionable.

There are two ways to interpret the results of the weighting result:

- The three damage models can be combined with their particular weighting set in table 7.4.
- The three damage models can be combined with the default weighting set in table 7.2

In theory the first option is the most consistent, but given the very low number of respondents in the hierarchist set, we propose to use the second option. The first option can still be used in a sensitivity analysis.

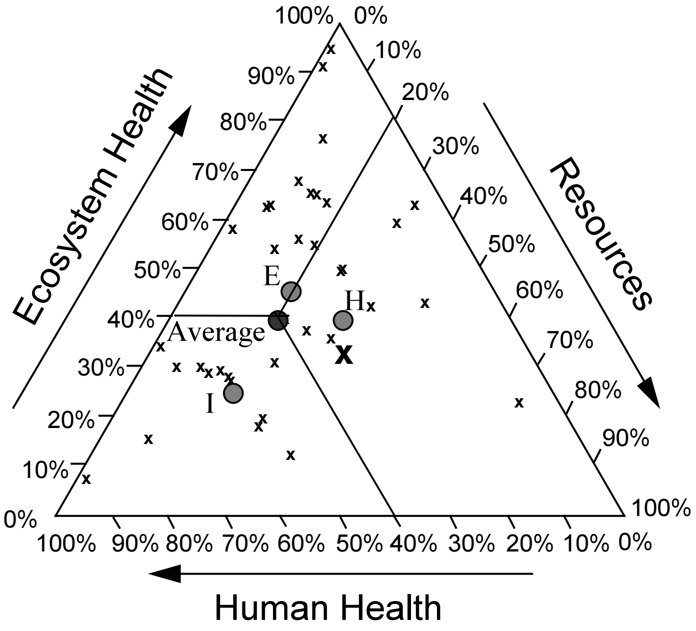


Figure 7.8: Projection of the average result and the result for the three perspectives over the results from the questionnaires.

7.5. The European LCA

Normalisation values for Europe are calculated using the report from [BLONK ET AL 1996]. The values from Blonk are updated with the latest European data on emissions, radiation and land-use (see annex report) In table 7.5 the normalisation values for Europe are summarised.

Egalitarian						
	Air	Water	Industrial soil	Agricultural soil	Total	per inhabitant
Carcinogenic effects [DALY/yr]	1.99E+05	3.10E+05	1.83E+05	6.77E+04	7.60E+05	2.00E-03
Respiratory (inorganic) [DALY/yr]	4.09E+06				4.09E+06	1.08E-02
Respiratory (organic) [DALY/yr]	2.60E+04				2.60E+04	6.84E-05
Climate Change [DALY/yr]	9.08E+05				9.08E+05	2.39E-03
Radiation [DALY/yr]	1.01E+04	9.84E+01			1.02E+04	2.68E-05
Ozone depletion [DALY/yr]	8.32E+04				8.32E+04	2.19E-04
Total Human health [DALY/yr]	**5.31E+06**	**3.10E+05**	**1.83E+05**	**6.77E+04**	**5.88E+06**	1.55E-02
Ecotoxicity (PAFm2yr/yr)	7.02E+11	7.87E+09	2.37E+12	4.32E+08	3.08E+12	8.11E+03
Ecotoxicity (PDFm2yr/yr)	7.02E+10	7.87E+08	2.37E+11	4.32E+07	3.08E+11	8.11E+02
Acidification/nutrification (PDF m2yr/yr)	1.43E+11				1.43E+11	3.75E+02
Land-use (PDF m2yr/yr)	1.50E+12				1.50E+12	3.95E+03
Total Ecosytem Quality (PDF.m2.yr/yr.)	**1.71E+12**	**7.87E+08**	**2.37E+11**	**4.32E+07**	**1.95E+12**	5.13E+03
Minerals [MJ/yr]					5.61E+10	1.48E+02
Fossil [MJ/yr					2.20E+12	5.79E+03
Total Resources [MJ/yr					**2.26E+12**	5.94E+03
Hierarchist						
	Air	Water	Industrial soil	Agricultural soil	Total	
Carcinogenic effects [DALY/yr]	1.99E+05	3.10E+05	1.83E+05	6.77E+04	7.60E+05	2.00E-03
Respiratory (inorganic) [DALY/yr]	4.05E+06				4.05E+06	1.07E-02
Respiratory (organic) [DALY/yr]	2.60E+04				2.60E+04	6.84E-05
Climate Change [DALY/yr]	9.08E+05				9.08E+05	2.39E-03
Radiation [DALY/yr]	1.01E+03	9.84E+01			1.02E+04	2.68E-05
Ozone depletion [DALY/yr]	8.32E+04				8.32E+04	2.19E-04
Total Human health [DALY/yr]	**5.27E+06**	**3.10E+05**	**1.83E+05**	**6.77E+04**	**5.84E+06**	1.54E-02
Ecotoxicity (PAFm2yr/yr)	7.02E+11	7.87E+09	2.37E+12	4.32E+08	3.08E+12	8.11E+03
Ecotoxicity (PDFm2yr/yr)	7.02E+10	7.87E+08	2.37E+11	4.32E+07	3.08E+11	8.11E+02
Acidification/nutrification (PDF m2yr/yr)	1.43E+11				1.43E+11	3.75E+02
Land-use (PDF m2yr/yr)	1.50E+12				1.50E+12	3.95E+03
Total Ecosytem Quality (PDF.m2.yr/yr.)	**1.71E+12**	**7.87E+08**	**2.37E+11**	**4.32E+07**	**1.95E+12**	5.13E+03
Minerals [MJ/yr]					5.61E+10	1.48E+02
Fossil [MJ/yr					3.14E+12	8.26E+03
Total Resources [MJ/yr					**3.20E+12**	8.41E+03
Individualist						
	Air	Water	Industrial soil	Agricultural soil	Total	
Carcinogenic effects [DALY/yr]	1.40E+04	6.20E+04	3.06E+03	0.00E+00	7.91E+04	2.08E-04
Respiratory (inorganic) [DALY/yr]	2.09E+06				2.09E+06	5.50E-03
Respiratory (organic) [DALY/yr]	2.42E+04				2.42E+04	6.37E-05
Climate Change [DALY/yr]	8.72E+05				9.09E+06	2.29E-03
Radiation [DALY/yr]	9.38E+02	5.74E+01			9.95E+02	2.62E-06
Ozone depletion [DALY/yr]	6.73E+04				6.73E+04	1.77E-04
Total Human health [DALY/yr]	**3.07E+06**	**6.21E+04**	**3.06E+03**	**0.00E+00**	**3.13E+06**	8.25E-03
Ecotoxicity (PAFm2yr/yr)	7.37E+10	5.10E+09	6.14E+11	4.32E+08	6.93E+11	1.82E+03
Ecotoxicity (PDFm2yr/yr)	7.37E+09	5.10E+08	6.14E+10	4.32E+07	6.93E+10	1.82E+02
Acidification/nutrification (PDF m2yr/yr)	1.43E+11				1.43E+11	3.76E+02
Land-use (PDF m2yr/yr)	1.50E+12				1.50E+12	3.95E+03
Total Ecosytem Quality (PDF.m2.yr/yr.)	**1.65E+12**	**2.62E+08**	**6.14E+10**	**4.32E+07**	**1.71E+12**	4.51E+03
Minerals [MJ/yr]					5.61E+10	1.48E+02
Fossil [MJ/yr					0.00E+00	0.00E+00
Total Resources [MJ/yr					**5.61E+10**	1.50E+02

Table 7.5: Normalisation values for Europe (totals for Europe and per inhabitant)

With the calculated normalisation figures and the weights it is possible to show the contribution of the impact categories to the three damage categories for the three perspectives:
For the hierarchist perspective (figure 7.9) the default weighting set is used, as we propose this combination as the default method. For the egalitarian and individualist perspective their matching weighting set is used.

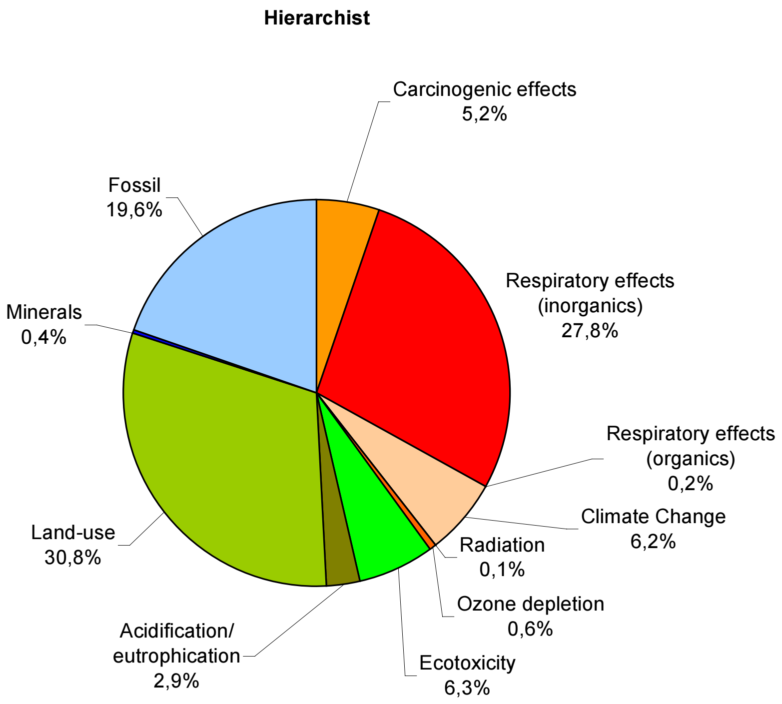


Figure 7.9.: Relative contribution of the impact categories to the European damage according to the hierarchist perspective, using the default weighting set (HH=40%,EQ=40%, R=20%).

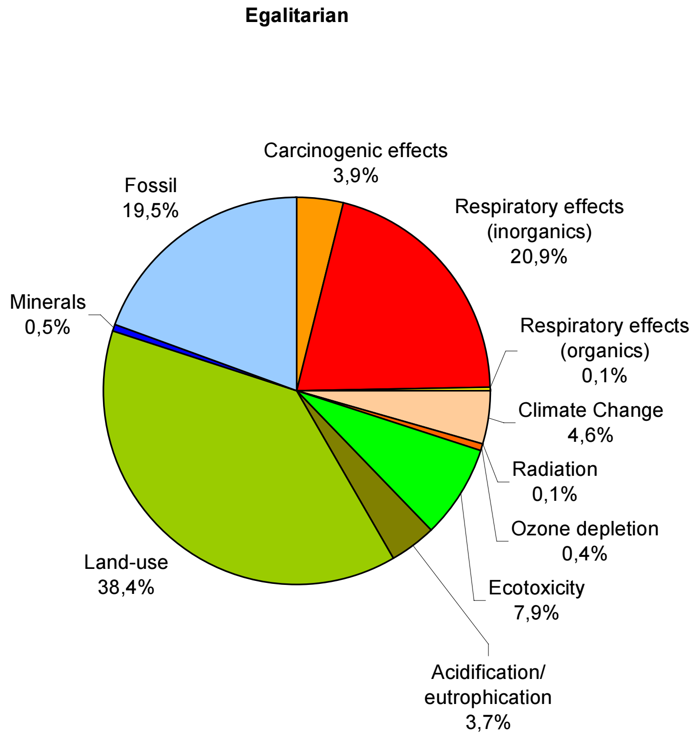


Figure 7.10. Relative contribution of the impact categories to the European damage according to the egalitarian perspective, using the weighting set for the egalitarian perspective (HH=30%,EQ=50%, R=20%).

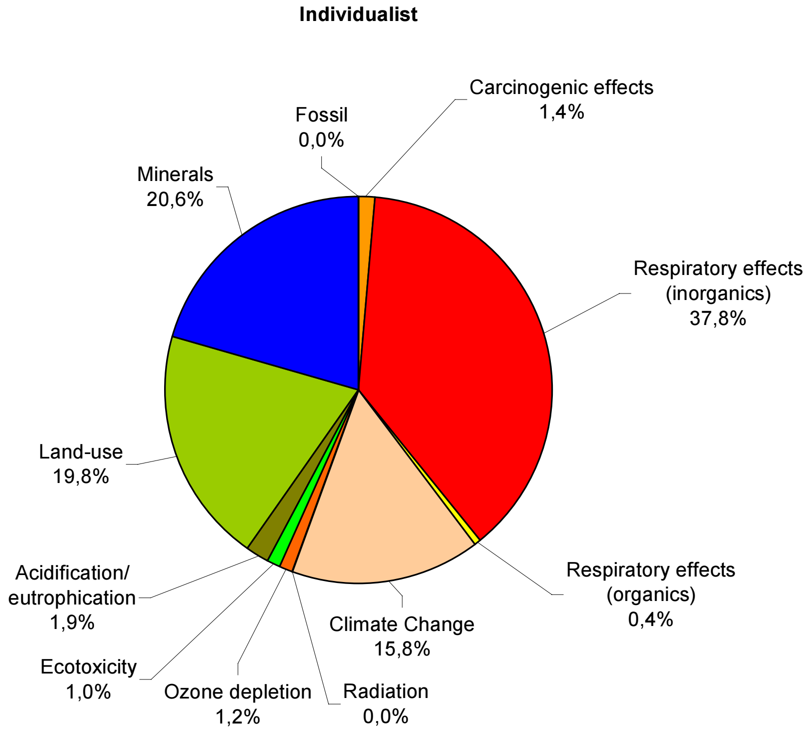


Figure 7.11: Relative contribution of the impact categories to the European damage according to the individualist perspective, using the weighting set for the individualist perspective (HH=55%, EQ=25%, R=20%).

The figures have been compiled by using the normalisation set of the yearly emissions, land-use and resource consumption per European inhabitant, as an input (inventory result) to the methodology. By definition, the result is 1000 Eco-indicator points[34]. The distribution of the shares over this result can be read as follows:

1. The weighting set determines which percentage of the damage is attributed to the damage categories.
2. The normalisation set determines the subdivision within the damage categories.

As explained in box 7.2 the result must be considered as a marginal and not an actual damage. Therefore comparison with other methods to calculate the damages in Europe is not directly possible.

From these figures some important conclusions can be drawn:

- In the hierarchist perspective, combined with the default weighting, land-use, respiratory effects as a result of inorganic substances and fossil fuels are clearly the most important impact categories.
- Surprisingly, some impact categories turn out to be quite insignificant on the European scale. In all perspectives the damages caused by respiratory effects due to organic substances (“summer smog”), ionising radiation, and ozone depletion. Of course this does not mean that these impact categories are a non-issue. For instance if an LCA is made of a paint system with organic solvents, or a nuclear power plant or an airconditioner, these impact categories could become quite significant, or even dominant.
- The high importance of land-use, especially in the egalitarian case, has as a consequence that ecotoxicity and acidification/eutrophication seem relatively unimportant.
- In the individualist perspective, respiratory effects due to inorganic substances is dominating, while for instance carcinogenic effects are quite insignificant. This can be explained as the Individualist perspective only considers a small number of substances. As a result respiratory effects becomes

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[34] The normalisation set is used to calculate the values for the three damage categories. In the normalisation step this result is divided by exactly the same normalisation factors. As a result the three normalised damage scores each have the value of one. As the sum of the weighting factors is 1000 (see box 8.1), multiplication with the weighting set results in the value 1000.

dominant within the human health damage category. As the weight of this category is relatively high (55%) respiratory effects dominate the picture.

- In the individualist perspective fossil fuels are not included, As the only remaining impact category is depletion of minerals, this becomes very important.
- There are very significant differences with the Eco-indicator 95 method in this method, the highest contribution came from ozone layer depletion, followed by pesticides, carcinogenic substances and acidification. The lower importance effects were heavy metals, winter smog and eutropication, while the lowest contribution came from summer smog and the greenhouse effect.

Perhaps the most important conclusion is that a damage approach such as the Eco-indicator 99 can show which impact categories are on average really important, and which are probably only important in specific cases. See also [BRAUNSCHWEIG et al 1998].

8. Conclusions and outlook

Box 8.1 Some hints for the application of the Eco-indicator methodology

In this box we would like to introduce a number of guidelines for the application of the methodology. This is needed, as we have introduced three versions of the damage model and four weighting sets. Furthermore we have experienced that after the publication of the Eco-indicator 95 different versions started to emerge.

Name

We propose to refer to the methodology as EI99(X,Y), in which X and Y have the following meaning:

- X refers to the perspective used in the damage model. It can either be H, E or I.
- Y refers to the weighting set used (if any). Possible letters are H, E and I for the specific weighting sets belonging to the perspectives, or A for the average weighting set or "-"if no weighting set is used.

Scaling factor

Experience shows that in most LCAs a typical product life cycle creates a much smaller environmental damage than the normalisation value (the damage created by an average European during one year). This means for the Eco-indicator 99 that most LCAs results are in micropoints or even nanopoints. In our experience this is not very practical. Therefore we propose to use a scaling factor of 1000. This means that, in stead of weights of 0.4 and 0.2, as proposed in paragraph 7.4 we use weights of 400 and 200. The result is that most LCAs will have a result in millipoints or micropoints.

8.1. Main achievements

The Eco-indicator 99 methodology has resulted in a completely new impact assessment method with a number of important achievements and methodological breakthroughs:

1. A fully consistent and almost complete modelling of the damage caused by a large number of relevant impact categories.
2. The structure of the method (not all details) is fully consistent with the requirements in ISO 14042, although some of the damage models may not fulfil the explicit or implicit standards regarding the scientific basis.
3. A completely new approaches to the modelling of resource depletion, land-use, climate change, ionising radiation, acidification/eutrophication (treated as a single category) and toxicity. In fact all impact categories have been defined in a completely new way.
4. A consistent use of fate models; not only for toxic chemicals, but also for radiation, eutrophication/acidification and greenhouse gases.
5. An almost complete specification of all the technical uncertainties. The squared geometric standard deviation has been specified for practically all factors.
6. A consistent approach towards modelling assumptions that cannot be avoided, and that cannot be covered under standard deviations, using Cultural Theory.
7. A preliminary, but consistently analysed panel assessment for the valuation has been included.

The intended application of this method has not changed; the Eco-indicator methodology is intended to be used as a tool for designers. The most practical application is the calculation of single scores for commonly used materials and processes. Such standard list has proven to be a very useful tool for designers, as they can perform their own LCA in a matter of minutes.

Next to this application it is clear that the methodology will also be used for the analysis of all kinds of LCA results; this also happened with the predecessor, the Eco-indicator 95. In order to facilitate this application the procedure has been split up in a number of stages. This allows the practitioner to study all intermediate results that are available between the stages. Furthermore a suggestion is made how to use the methodology without the final weighting step, using the "Triangle concept". With this concept it is possible to use the methodology in consensus building process instead of a tool that produces "simple truths".

8.2. Further developments and improvements

The framework of the methodology developed in this project seems stable for the coming decade. Not only is it compatible with the structure of the ISO 14042 standard, during the development process we have found it is a good framework to include all kinds of different damage models.

The fate and damage models, as well as the normalisation and panel assessment can still be improved, as the scientific understanding of the Ecosphere is under constant development. New updates will be necessary the coming years. The priorities for improvements are:

- The modelling of the Ecosystem Quality damage category. The conceptual difficulties in combining inhomogeneous damage definitions need to be resolved.
- The missing impact categories in relation to Ecosystem Quality, such as climate change, aquatic eutrophication/acidification and increased UV radiation need to be developed.
- A truly European model for the fate and damage model for acidification/eutrophication is required.
- The EUSES model, especially the problems created by closing Europe, can still be improved.
- Climate change is still a very problematic impact category. The negative damages to Human Health and the difficulties in modelling damages to ecosystems are major problems.
- The lack of reliable data for normalisation is a major problem for some impact categories
- The panel procedure needs to be repeated for a much larger group with a better statistical representativeness.

The chances that these improvements can be achieved within one or two years are quite realistic, as we can mention some important developments:

- A large project at RIVM is aimed at developing a better understanding of the meaning of the PAF and PDF scores and their mutual relation, using field observations.
- A new version of EUSES has already been released. This version has been adopted in such a way that most of the problems concerning a closed Europe model can be solved. The model is applied by [HUIJBREGTS 1999].[35]
- The land-use model of [KÖLLNER 1999] is also being developed further.
- There is continuous improvement in the modelling of climate change.
- The panel procedure will indeed be repeated by Mettier [see METTIER 1999], using a larger sample and incorporating the experiences from the first version. The questions to the panel should be focused on the need for marginal weights.

8.3. Application advice

The Eco-indicator 99 methodology is not perfect and probably never will be. It is based on the best available data and scientific understanding of the environmental mechanisms.

Please be aware of the limitations, and please do not consider the Eco-indicator values as an absolute truth when applying the methodology. It is an INDICATOR aimed at showing the approximately correct direction for designers who want to analyse and minimise the environmental load of product systems.

Neither the researchers nor the commissioners for this project, can accept any responsibility for the consequences and incurred damages in whatever form of decisions, that are apparently based on this methodology.

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[35] As Huijbregts uses a number of different assumptions and data sources it is not advisable to replace the data from this project by his data

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Annex 1: Operational damage factors

This annex contains the necessary information to make the Eco-indicator 99 method operational. The use of the perspectives requires some special attention.

The use of perspectives means that there are several versions of the methodology

- There are three versions of the damage model, as the perspectives are used to guide decision making on subjective choices in the damage model.
- The normalisation set is based on a damage calculation of all relevant European emissions, extractions and land-uses. As there are three damage models, there are also three normalisation sets.
- There are four versions of the weighting set: One average for all panellists, and three versions for subgroups of the panel, that could be regarded as adhering to a perspective.

We suggest to use the following standard codes in reporting results obtained with the Eco-indicator methodology:

- Perspective, abbreviated, to H, E, I
- Weighting set, abbreviated to H, E, I and A (average)

It is not necessary to specify the normalisation set, as the perspective for the damage model and the normalisation set should always be the same.

In principle many combinations are possible, but the most relevant are:

1 The Hierarchist damage model and normalisation with the A*verage* weighting. (H,A)
2 The Egalitarian damage model and normalisation with the Egalitarian weighting. (E,E)
3 The Individualist damage model and normalisation with the Individualist weighting. (I,I)

In principle a fourth version can be made, using the Hierarchist damage model and the Hierarchist weighting (H,H). However, as the number of respondents in the panel was very low, and the standard deviation was very high, this weighting set is too unreliable. The Egalitarian and Individualist damage models can also be combined with the average weighting (E,A and I,A).

We consider the Hierarchist perspective with the Average weighting set (H,A) to be the default version of the methodology. The other perspectives can be used in a robustness analysis. If the conclusions of your LCA (A is better than B) are the same for all perspectives, one may conclude that the result is independent of the perspectives, and thus independent of the time frame, the required evidence on the cause-effect chain or other subjective choices. If the results obtained by one of the perspectives contradicts the results obtained by the others, one can conclude that the result is dependent of the perspective, and thus of some subjective choices. In this case further study is needed to assess the opinion of the stakeholders regarding these choices. The column headings can be explained as follows:

Compartment	This refers to the emission compartment, air, water, and soil. The difference between emissions in industrial or natural soil is indicated in the substance name. This column is not used for land-use and resources
Substance	The substance name
Damage factor	This is the “best estimate” value of the damage factor the unit is specified at the top of every table.
Normalised damage factor	The damage factor divided by the normalisation value for the damage category
Weighted damage factor	The normalised damage factor multiplied by the weighting factor

It was not possible to indicate uncertainty for all damage factors. Therefor the following tables do not list uncertainties.

A best estimate is mate for the damage factor of substance groups like “PAHs” and “metals”. If all substances in a substance group contribute more or less to the same kind of damage, the average characterisation value was taken. For example, carbon substances like alkenes and aromatics may contribute to the formation of trophospheric ozone and thereby to respiratory health problems. If only a few substances in the substance group cause serious damage, the damage factor was estimated based on the composition of the substance group.

For example, the substance groups “metals” contains both heavy metals and light metals. The composition of the substance group “metals” is determined on base of all metals emitted during UCPTE electricity production. Damage factors values for the different perspectives are calculated by multiplying the fraction of individual metals with the corresponding damage factors.

1 Damage factors in the hierarchist perspective (default) (H,A)

This annex lists the Eco-indicator 99 damage factors for the substance lists that can be found in most popular LCA databases. In this case the hierarchist perspective is used, combined with the default (average) weighting factors. Next to the damage factors two columns are added with the normalised and weighted damages. The normalisation factors and the weights are specified below:

	Normalisation	Weights
Human Health	1.54E-02	400
Ecosystem Quality	5.13E+03	400
Resources	8.41E+03	200

Below the impact categories are listed per damage category.

1.1 Damage category Human Health (H,A)

The human health damages are specified in DALYs. This is short for Disability Adjusted Life Years. A damage of 1 means one life year of one individual is lost, or one person suffers four year from a disability with a weight of 0.25.

1.1.1 Carcinogenic effects on humans (H,A)

For the fate and exposure it is important to distinguish emissions to soil between emissions in industrial (ind.) or agricultural (agr.) soil. All emissions of pesticides are assumed to occur in agricultural soil, all other emissions are assumed to occur in industrial (or urban) soil. No direct emissions are assumed to occur in natural soil.
Fate factors are calculated with EUSES. Substances from IARC substances groups 1, 2a and 2b are included.

All damage factors are expressed per kg emission. The unit of damage is DALYs.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,2-dibromoethane	2.60E-04	1.69E-02	6.75E+00
Air	1.2-dichloroethane	2.98E-05	1.94E-03	7.74E-01
Air	1.3-butadiene	1.58E-05	1.03E-03	4.10E-01
Air	1.4-dioxane	1.39E-07	9.03E-06	3.61E-03
Air	2.4.6-trichlorophenol	2.05E-06	1.33E-04	5.32E-02
Air	acetaldehyde	2.16E-07	1.40E-05	5.61E-03
Air	acrylonitrile	1.69E-05	1.10E-03	4.39E-01
Air	alpha-hexachlorocyclohexan	3.00E-04	1.95E-02	7.79E+00
Air	Arsenic	2.46E-02	1.60E+00	6.39E+02
Air	Bis(chloromethyl)ether	7.48E-03	4.86E-01	1.94E+02
Air	benzene	2.50E-06	1.62E-04	6.49E-02
Air	benzo(a)anthracene	5.86E-02	3.81E+00	1.52E+03
Air	benzo(a)pyrene	3.98E-03	2.58E-01	1.03E+02
Air	benzotrichloride	6.60E-03	4.29E-01	1.71E+02
Air	benzylchloride	1.04E-05	6.75E-04	2.70E-01
Air	beta-chlorocyclohexan	9.99E-05	6.49E-03	2.59E+00
Air	bromodichloromethane	8.76E-06	5.69E-04	2.28E-01
Air	Cadmium	1.35E-01	8.77E+00	3.51E+03
Air	Chromium (VI)	1.75	1.14E+02	4.55E+04
Air	di(2-ethylhexyl)phthalate	3.38E-05	2.19E-03	8.78E-01
Air	dibenz(a)anthracene	3.10E+01	2.01E+03	8.05E+05
Air	dichloromethane	4.36E-07	2.83E-05	1.13E-02
Air	Dichlorvos	3.15E-05	2.05E-03	8.18E-01
Air	2.3.7.8-TCDD Dioxin	1.79E+02	1.16E+04	4.65E+06
Air	epichlorohydrin	3.02E-07	1.96E-05	7.84E-03
Air	ethylene oxide	1.83E-04	1.19E-02	4.75E+00
Air	formaldehyde	9.91E-07	6.44E-05	2.57E-02
Air	gamma-HCH (Lindane)	3.49E-04	2.27E-02	9.06E+00
Air	Hexachlorobenzene	8.25E-02	5.36E+00	2.14E+03
Air	metals	5.20E-03	3.38E-01	1.35E+02
Air	Nickel	2.35E-02	1.53E+00	6.10E+02
Air	Nickel-refinery-dust	4.74E-02	3.08E+00	1.23E+03
Air	Nickel-subsulfide	9.48E-02	6.16E+00	2.46E+03
Air	PAH's	1.70E-04	1.10E-02	4.42E+00
Air	particles diesel soot	9.78E-06	6.35E-04	2.54E-01
Air	Polychlorobiphenyls	1.97E-03	1.28E-01	5.12E+01
Air	pentachlorophenol	7.21E-03	4.68E-01	1.87E+02
Air	propyleneoxide	1.17E-05	7.60E-04	3.04E-01
Air	styrene	2.44E-08	1.58E-06	6.34E-04
Air	perchloroethylene	4.82E-07	3.13E-05	1.25E-02
Air	carbontetrachloride	8.38E-04	5.44E-02	2.18E+01
Air	chloroform	2.63E-05	1.71E-03	6.83E-01
Air	vinyl chloride	2.09E-07	1.36E-05	5.43E-03
Water	1.2-dibromoethane	1.24E-03	8.05E-02	3.22E+01
Water	1.2-dichloroethane	2.98E-05	1.94E-03	7.74E-01
Water	1.3-butadiene	3.37E-04	2.19E-02	8.75E+00
Water	1.4-dioxane	9.21E-07	5.98E-05	2.39E-02
Water	2.4.6-trichlorophenol	1.05E-05	6.82E-04	2.73E-01
Water	acetaldehyde	9.23E-07	5.99E-05	2.40E-02
Water	acrylonitrile	4.16E-05	2.70E-03	1.08E+00
Water	alpha-hexachlorocyclohexan	6.85E-03	4.45E-01	1.78E+02
Water	Arsenic	6.57E-02	4.27E+00	1.71E+03
Water	Bis(chloromethyl)ether	1.54E-02	1.00E+00	4.00E+02
Water	benzene	4.12E-06	2.68E-04	1.07E-01
Water	benzo(a)anthracene	6.58E-01	4.27E+01	1.71E+04
Water	benzo(a)pyrene	2.99	1.94E+02	7.77E+04
Water	benzotrichloride	9.46E-03	6.14E-01	2.46E+02
Water	benzylchloride	1.98E-05	1.29E-03	5.14E-01
Water	beta-chlorocyclohexan	5.75E-03	3.73E-01	1.49E+02
Water	bromodichloromethane	9.36E-06	6.08E-04	2.43E-01
Water	Cadmium	7.12E-02	4.62E+00	1.85E+03
Water	Chromium (VI)	3.43E-01	2.23E+01	8.91E+03
Water	di(2-ethylhexyl)phthalate	6.64E-04	4.31E-02	1.72E+01
Water	dibenz(a)anthracene	4.07E+01	2.64E+03	1.06E+06
Water	dichloromethane	4.97E-07	3.23E-05	1.29E-02
Water	Dichlorvos	1.17E-05	7.60E-04	3.04E-01
Water	dioxins (TEQ)	2.02E+03	1.31E+05	5.25E+07
Water	epichloorhydrin	9.90E-07	6.43E-05	2.57E-02
Water	ethylene oxide	1.39E-04	9.03E-03	3.61E+00
Water	formaldehyde	4.97E-06	3.23E-04	1.29E-01
Water	gamma-HCH (Lindane)	4.16E-03	2.70E-01	1.08E+02
Water	hexachlorobenzene	1.25E-01	8.12E+00	3.25E+03
Water	Nickel	3.11E-02	2.02E+00	8.08E+02
Water	Nickel--subsulfide	5.02E-03	3.26E-01	1.30E+02
Water	Nickel-refinery-dust	1.00E-02	6.49E-01	2.60E+02
Water	PAH's	2.60E-03	1.69E-01	6.75E+01
Water	Polychlorobiphenyls	3.91E-02	2.54E+00	1.02E+03
Water	pentachlorophenol	2.29E-02	1.49E+00	5.95E+02
Water	propylene oxide	1.74E-05	1.13E-03	4.52E-01
Water	styrene	1.22E-06	7.92E-05	3.17E-02
Water	perchloroethylene	4,72E-07	3,06E-05	1,23E-02
Water	carbontetrachloride	8.29E-04	5.38E-02	2.15E+01
Water	chloroform	2.60E-05	1.69E-03	6.75E-01
Water	vinyl chloride	2.84E-07	1.84E-05	7.38E-03
Soil	1,2-dibromoethane (ind.)	3.81E-03	2.47E-01	9.90E+01
Soil	1,2-dichloroethane (ind.)	4.58E-04	2.97E-02	1.19E+01
Soil	1,3-butadiene (ind.)	1.20E-05	7.79E-04	3.12E-01
Soil	1,4-dioxane (ind.)	3.10E-07	2.01E-05	8.05E-03
Soil	2,4,6-trichlorophenol (ind.)	2.76E-06	1.79E-04	7.17E-02
Soil	acetaldehyde (ind.)	4.77E-07	3.10E-05	1.24E-02
Soil	acrylonitrile (ind.)	7.01E-05	4.55E-03	1.82E+00
Soil	alpha-hexachlorocyclohexan (agr.)	2.32E-02	1.51E+00	6.03E+02
Soil	Arsenic (ind.)	1.32E-02	8.57E-01	3.43E+02
Soil	Bis(chloromethyl)ether (ind.)	1.68E-02	1.09E+00	4.36E+02
Soil	benzene (ind.)	1.33E-05	8.64E-04	3.45E-01
Soil	benzo(a)anthracene (ind.)	1.60E-01	1.04E+01	4.16E+03
Soil	benzo(a)pyrene (ind.)	2.06E-03	1.34E-01	5.35E+01
Soil	benzotrichloride (ind.)	1.32E-01	8.57E+00	3.43E+03
Soil	benzylchloride (ind.)	4.16E-05	2.70E-03	1.08E+00

Soil	beta-chlorocyclohexan (agr.)	7.36E-03	4.78E-01	1.91E+02
Soil	bromodichloromethane (ind.)	7.82E-05	5.08E-03	2.03E+00
Soil	Cadmium (ind.)	3.98E-03	2.58E-01	1.03E+02
Soil	Chromium (ind.)	2.71E-01	1.76E+01	7.04E+03
Soil	di(2-ethylhexyl)phthalate(ind)	3.18E-07	2.06E-05	8.26E-03
Soil	dibenz(a)anthracene (ind.)	2.44E+01	1.58E+03	6.34E+05
Soil	dichloromethane (ind.)	5.99E-06	3.89E-04	1.56E-01
Soil	Dichlorvos (agr.)	2.25E-05	1.46E-03	5.84E-01
Soil	2,3,7,8-TCDD Dioxin (ind.)	7.06	4.58E+02	1.83E+05
Soil	epichloorhydrin (ind.)	1.30E-06	8.44E-05	3.38E-02
Soil	ethyleenoxide (ind.)	2.38E-03	1.55E-01	6.18E+01
Soil	formaldehyde (ind.)	1.83E-06	1.19E-04	4.75E-02
Soil	gamma-HCH (Lindane) (agr.)	8.64E-03	5.61E-01	2.24E+02
Soil	hexachlorobenzene (ind.)	1.47E-01	9.55E+00	3.82E+03
Soil	Nickel (ind.)	3.94E-03	2.56E-01	1.02E+02
Soil	Nickel-refinery-dust (ind.)	6.37E-03	4.14E-01	1.65E+02
Soil	Nickel-subsulfide (ind.)	1.27E-02	8.25E-01	3.30E+02
Soil	PCBs (ind.)	2.04E-02	1.32E+00	5.30E+02
Soil	pentachloorfenol (ind.)	1.26E-05	8.18E-04	3.27E-01
Soil	propyleenoxide (ind.)	1.40E-04	9.09E-03	3.64E+00
Soil	styrene (ind.)	2.09E-08	1.36E-06	5.43E-04
Soil	perchloroethylene (ind.)	6.00E-06	3.90E-04	1.56E-01
Soil	carbontetrachloride (ind.)	3.99E-02	2.59E+00	1.04E+03
Soil	chloroform (ind.)	4.12E-06	2.68E-04	1.07E-01
Soil	vinylchloride (ind.)	7.67E-07	4.98E-05	1.99E-02

1.1.2 Respiratory effects on humans caused by organic substances (H,A)

This impact category replaces more or less the summer smog category. Fate analysis is based on empirical data.
All damage factors are expressed per kg emitted substance. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.96E-08	1.27E-06	5.09E-04
Air	1,2,3-trimethyl benzene	2.72E-06	1.77E-04	7.06E-02
Air	1,2,4-trimethyl benzene	2.72E-06	1.77E-04	7.06E-02
Air	1,3,5-trimethyl benzene	2.98E-06	1.94E-04	7.74E-02
Air	1,3-butadiene	1.87E-06	1.21E-04	4.86E-02
Air	1-butene	2.30E-06	1.49E-04	5.97E-02
Air	1-butoxy propanol	9.36E-07	6.08E-05	2.43E-02
Air	1-hexene	1.87E-06	1.21E-04	4.86E-02
Air	1-methoxy 2-propanol	7.91E-07	5.14E-05	2.05E-02
Air	1-pentene	2.13E-06	1.38E-04	5.53E-02
Air	2,2-dimethyl butane	5.19E-07	3.37E-05	1.35E-02
Air	2,3-dimethyl butane	1.19E-06	7.73E-05	3.09E-02
Air	2-butoxy ethanol	9.36E-07	6.08E-05	2.43E-02
Air	2-ethoxy ethanol	8.34E-07	5.42E-05	2.17E-02
Air	2-hexanone	1.19E-06	7.73E-05	3.09E-02
Air	2-methoxy ethanol	6.47E-07	4.20E-05	1.68E-02
Air	2-methyl 1-butanol	8.51E-07	5.53E-05	2.21E-02
Air	2-methyl 1-butene	1.70E-06	1.10E-04	4.42E-02
Air	2-methyl 2-butanol	3.06E-07	1.99E-05	7.95E-03
Air	2-methyl 2-butene	1.79E-06	1.16E-04	4.65E-02
Air	2-methyl hexane	8.51E-07	5.53E-05	2.21E-02
Air	2-methyl pentane	9.36E-07	6.08E-05	2.43E-02
Air	2-pentanone	1.19E-06	7.73E-05	3.09E-02
Air	3,5-diethyl toluene	2.81E-06	1.82E-04	7.30E-02
Air	3,5-dimethyl ethyl benzene	2.81E-06	1.82E-04	7.30E-02
Air	3-hexanone	1.28E-06	8.31E-05	3.32E-02
Air	3-methyl 1-butanol	8.51E-07	5.53E-05	2.21E-02
Air	3-methyl 1-butene	1.45E-06	9.42E-05	3.77E-02
Air	3-methyl 2-butanol	7.91E-07	5.14E-05	2.05E-02
Air	3-methyl hexane	7.83E-07	5.08E-05	2.03E-02
Air	3-methyl pentane	1.02E-06	6.62E-05	2.65E-02
Air	3-pentanol	9.36E-07	6.08E-05	2.43E-02
Air	3-pentanone	8.51E-07	5.53E-05	2.21E-02
Air	acetaldehyde	1.36E-06	8.83E-05	3.53E-02
Air	acetic acid	2.13E-07	1.38E-05	5.53E-03
Air	acetone	2.04E-07	1.32E-05	5.30E-03
Air	propionaldehyde	1.70E-06	1.10E-04	4.42E-02
Air	alcohols	7.60E-07	4.94E-05	1.97E-02
Air	aldehydes	1.40E-06	9.09E-05	3.64E-02
Air	alkanes	7.50E-07	4.87E-05	1.95E-02
Air	alkenes	2.10E-06	1.36E-04	5.45E-02
Air	benzene	4.68E-07	3.04E-05	1.22E-02
Air	butane	7.57E-07	4.92E-05	1.97E-02
Air	butanol	1.36E-06	8.83E-05	3.53E-02
Air	butene	2.47E-06	1.60E-04	6.42E-02
Air	cis 1,2-dichloroethene	9.36E-07	6.08E-05	2.43E-02
Air	cis 2-butene	2.47E-06	1.60E-04	6.42E-02
Air	cis 2-hexene	2.30E-06	1.49E-04	5.97E-02
Air	cis 2-pentene	2.38E-06	1.55E-04	6.18E-02
Air	CxHy aromatic	2.10E-06	1.36E-04	5.45E-02
Air	CxHy chloro	3.50E-07	2.27E-05	9.09E-03
Air	CxHy halogenated	3.50E-07	2.27E-05	9.09E-03
Air	cyclohexane	6.21E-07	4.03E-05	1.61E-02
Air	cyclohexanol	9.36E-07	6.08E-05	2.43E-02
Air	cyclohexanone	6.47E-07	4.20E-05	1.68E-02
Air	decane	8.26E-07	5.36E-05	2.15E-02
Air	di-i-propyl ether	1.02E-06	6.62E-05	2.65E-02
Air	diacetone alcohol	5.62E-07	3.65E-05	1.46E-02
Air	dichloromethane	1.45E-07	9.42E-06	3.77E-03
Air	diethyl ether	1.02E-06	6.62E-05	2.65E-02
Air	dimethyl ether	3.74E-07	2.43E-05	9.71E-03
Air	dodecane	7.66E-07	4.97E-05	1.99E-02
Air	esters	3.70E-07	2.40E-05	9.61E-03
Air	ethane	2.64E-07	1.71E-05	6.86E-03
Air	ethane diol	8.26E-07	5.36E-05	2.15E-02
Air	ethanol	8.34E-07	5.42E-05	2.17E-02
Air	ethene	2.13E-06	1.38E-04	5.53E-02
Air	ethers	7.40E-07	4.81E-05	1.92E-02
Air	ethyl t-butyl ether	4.60E-07	2.99E-05	1.19E-02
Air	ethylacetate	4.60E-07	2.99E-05	1.19E-02
Air	ethylbenzene	1.53E-06	9.94E-05	3.97E-02
Air	acetylene	1.87E-07	1.21E-05	4.86E-03
Air	formaldehyde	1.11E-06	7.21E-05	2.88E-02
Air	formic acid	6.89E-08	4.47E-06	1.79E-03
Air	heptane	1.11E-06	7.21E-05	2.88E-02
Air	hexane	1.02E-06	6.62E-05	2.65E-02
Air	i-butane	6.64E-07	4.31E-05	1.72E-02
Air	i-butanol	8.09E-07	5.25E-05	2.10E-02
Air	i-butyraldehyde	1.11E-06	7.21E-05	2.88E-02
Air	i-pentane	8.51E-07	5.53E-05	2.21E-02
Air	i-propanol	2.98E-07	1.94E-05	7.74E-03
Air	i-propyl acetate	4.60E-07	2.99E-05	1.19E-02
Air	i-propyl benzene	1.11E-06	7.21E-05	2.88E-02
Air	isoprene	2.38E-06	1.55E-04	6.18E-02
Air	ketones	8.70E-07	5.65E-05	2.26E-02
Air	m-ethyl toluene	2.21E-06	1.44E-04	5.74E-02
Air	m-xylene	2.38E-06	1.55E-04	6.18E-02
Air	methane	1.28E-08	8.31E-07	3.32E-04
Air	methanol	2.81E-07	1.82E-05	7.30E-03
Air	methyl acetate	1.02E-07	6.62E-06	2.65E-03
Air	methyl chloride	1.11E-08	7.21E-07	2.88E-04
Air	methyl ethyl ketone	8.09E-07	5.25E-05	2.10E-02
Air	methyl formate	7.15E-08	4.64E-06	1.86E-03
Air	methyl i-butyl ketone	1.02E-06	6.62E-05	2.65E-02
Air	methyl i-propyl ketone	7.83E-07	5.08E-05	2.03E-02
Air	methyl propene	1.36E-06	8.83E-05	3.53E-02
Air	methyl t-butyl ether	3.32E-07	2.16E-05	8.62E-03
Air	methyl t-butyl ketone	6.98E-07	4.53E-05	1.81E-02
Air	n-butanol	1.36E-06	8.83E-05	3.53E-02
Air	n-butyl acetate	5.19E-07	3.37E-05	1.35E-02
Air	n-butyraldehyde	1.70E-06	1.10E-04	4.42E-02
Air	n-propanol	1.19E-06	7.73E-05	3.09E-02
Air	n-propyl acetate	6.21E-07	4.03E-05	1.61E-02
Air	n-propyl benzene	1.36E-06	8.83E-05	3.53E-02
Air	neopentane	3.74E-07	2.43E-05	9.71E-03
Air	NMVOC	1.28E-06	8.31E-05	3.32E-02
Air	nonane	8.51E-07	5.53E-05	2.21E-02
Air	o-ethyl toluene	1.96E-06	1.27E-04	5.09E-02
Air	o-xylene	2.30E-06	1.49E-04	5.97E-02
Air	octane	9.36E-07	6.08E-05	2.43E-02
Air	p-ethyl toluene	1.96E-06	1.27E-04	5.09E-02

Air	p-xylene	2.21E-06	1.44E-04	5.74E-02
Air	pentanal	1.62E-06	1.05E-04	4.21E-02
Air	pentane	8.51E-07	5.53E-05	2.21E-02
Air	propane	3.83E-07	2.49E-05	9.95E-03
Air	propane diol	1.02E-06	6.62E-05	2.65E-02
Air	propene	2.38E-06	1.55E-04	6.18E-02
Air	propanoic acide	3.23E-07	2.10E-05	8.39E-03
Air	s-butanol	8.51E-07	5.53E-05	2.21E-02
Air	s-butyl acetate	5.79E-07	3.76E-05	1.50E-02
Air	t-butanol	2.64E-07	1.71E-05	6.86E-03
Air	t-butyl acetate	1.36E-07	8.83E-06	3.53E-03
Air	perchloroethylene	6.21E-08	4.03E-06	1.61E-03
Air	toluene	1.36E-06	8.83E-05	3.53E-02
Air	trans 1,2-dichloroethene	8.43E-07	5.47E-05	2.19E-02
Air	trans 2-butene	2.47E-06	1.60E-04	6.42E-02
Air	trans 2-hexene	2.30E-06	1.49E-04	5.97E-02
Air	trans 2-pentene	2.38E-06	1.55E-04	6.18E-02
Air	trichloroethylene	6.98E-07	4.53E-05	1.81E-02
Air	chloroform	4.94E-08	3.21E-06	1.28E-03
Air	undecane	8.26E-07	5.36E-05	2.15E-02
Air	VOC	6.46E-07	4.19E-05	1.68E-02
Air	xylene	2.21E-06	1.44E-04	5.74E-02

1.1.3 Respiratory effects on humans caused by inorganic substances (H,A)

This impact category replaces more or less the winter smog category. Fate analysis is based on empirical data.
All damage factors are expressed per kg emission. The unit of damage is DALYs.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	8.50E-05	5.52E-03	2.21E+00
Air	dust (PM10)	3.75E-04	2.44E-02	9.74E+00
Air	dust (PM2.5)	7.00E-04	4.55E-02	1.82E+01
Air	TSP	1.10E-04	7.14E-03	2.86E+00
Air	NO	1.37E-04	8.90E-03	3.56E+00
Air	NO2	8.87E-05	5.76E-03	2.30E+00
Air	NOx	8.87E-05	5.76E-03	2.30E+00
Air	NOx (as NO2)	8.87E-05	5.76E-03	2.30E+00
Air	SO2	5.46E-05	3.55E-03	1.42E+00
Air	SO3	4.37E-05	2.84E-03	1.14E+00
Air	SOx	5.46E-05	3.55E-03	1.42E+00
Air	SOx (as SO2)	5.46E-05	3.55E-03	1.42E+00

1.1.4 Damages to human health caused by climate change (H,A)

Damage calculation was performed over a time scale of 200 years. The IPCC equivalence factors have been modified. As damage is not linear dependent on the atmospheric lifetime, a separate damage calculation is made for CO2, CH4 en CH4:

- Gasses with an atmospheric lifetime below 20 years are assumed to behave like methane
- Gasses with an atmospheric lifetime between 20 and 100 years behave like CO2
- Gasses with an atmospheric lifetime oh more than 100 years are assumed to behave like N2O

This means that the IPCC equivalency factor table is split in three groups.
All damage factors are expressed per kg substance. The unit of damage is DALYs.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	methyl chloroform	-4.3E-05	-2.79E-03	-1.12E+00
Air	perfluorethane	2.00E-03	1.30E-01	5.19E+01
Air	trifluoroiodomethane	2.10E-07	1.36E-05	5.45E-03
Air	perfluormethane	1.40E-03	9.09E-02	3.64E+01
Air	CFC-11	2.20E-04	1.43E-02	5.71E+00
Air	CFC-113	6.30E-04	4.09E-02	1.64E+01
Air	CFC-12	1.40E-03	9.09E-02	3.64E+01
Air	carbon dioxide	2.10E-07	1.36E-05	5.45E-03
Air	methylene chloride	1.90E-06	1.23E-04	4.94E-02
Air	HALON-1301	-7.10E-03	-4.61E-01	-1.84E+02
Air	HCFC-123	6.60E-06	4.29E-04	1.71E-01
Air	HCFC-124	8.50E-05	5.52E-03	2.21E+00
Air	HCFC-141b	5.20E-05	3.38E-03	1.35E+00
Air	HCFC-142b	3.40E-04	2.21E-02	8.83E+00
Air	HCFC-22	2.80E-04	1.82E-02	7.27E+00
Air	HFC-125	5.70E-04	3.70E-02	1.48E+01
Air	HFC-134	2.10E-04	1.36E-02	5.45E+00
Air	HFC-134a	2.70E-04	1.75E-02	7.01E+00
Air	HFC-143	6.30E-05	4.09E-03	1.64E+00
Air	HFC-143a	7.80E-04	5.06E-02	2.03E+01
Air	HFC-152a	2.90E-05	1.88E-03	7.53E-01
Air	HFC-227ea	5.90E-04	3.83E-02	1.53E+01
Air	HFC-23	2.60E-03	1.69E-01	6.75E+01
Air	HFC-236fa	1.40E-03	9.09E-02	3.64E+01
Air	HFC-245ca	1.20E-04	7.79E-03	3.12E+00
Air	HFC-32	1.40E-04	9.09E-03	3.64E+00
Air	HFC-41	3.10E-05	2.01E-03	8.05E-01
Air	HFC-4310mee	2.70E-04	1.75E-02	7.01E+00
Air	methane	4.40E-06	2.86E-04	1.14E-01
Air	nitrous oxide	6.90E-05	4.48E-03	1.79E+00
Air	perfluorbutane	1.50E-03	9.74E-02	3.90E+01
Air	perfluorcyclobutane	1.90E-03	1.23E-01	4.94E+01
Air	perfluorhexane	1.60E-03	1.04E-01	4.16E+01
Air	perfluorpentane	1.70E-03	1.10E-01	4.42E+01
Air	perfluorpropane	1.50E-03	9.74E-02	3.90E+01
Air	sulphur hexafluoride	5.30E-03	3.44E-01	1.38E+02
Air	carbontetrachloride	-2.60E-04	-1.69E-02	-6.75E+00
Air	chloroform	8.30E-07	5.39E-05	2.16E-02

1.1.5 Human health effects caused by ionising radiation (H,A)

Fate-, and exposure models are based on studies for the French nuclear industry. All damage factors are based on a release of 1 Bequerel (Bq). The unit of damage is DALYs.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	C-14	2.10E-10	1.36E-08	5.45E-06
Air	Co-58	4.30E-13	2.79E-11	1.12E-08
Air	Co-60	1.60E-11	1.04E-09	4.16E-07
Air	Cs-134	1.20E-11	7.79E-10	3.12E-07
Air	Cs-137	1.30E-11	8.44E-10	3.38E-07
Air	H-3	1.40E-14	9.09E-13	3.64E-10
Air	I-129	9.40E-10	6.10E-08	2.44E-05
Air	I-131	1.60E-13	1.04E-11	4.16E-09
Air	I-133	9.40E-15	6.10E-13	2.44E-10
Air	Kr-85	1.40E-16	9.09E-15	3.64E-12
Air	Pb-210	1.50E-12	9.74E-11	3.90E-08
Air	Po-210	1.50E-12	9.74E-11	3.90E-08
Air	Pu alpha	8.30E-11	5.39E-09	2.16E-06
Air	Pu-238	6.70E-11	4.35E-09	1.74E-06
Air	Ra-226	9.10E-13	5.91E-11	2.36E-08
Air	Rn-222	2.40E-14	1.56E-12	6.23E-10
Air	Th-230	4.50E-11	2.92E-09	1.17E-06
Air	U-234	9.70E-11	6.30E-09	2.52E-06
Air	U-235	2.10E-11	1.36E-09	5.45E-07
Air	U-238	8.20E-12	5.32E-10	2.13E-07
Air	Xe-133	1.40E-16	9.09E-15	3.64E-12
Water	Ag-110m	5.10E-13	3.31E-11	1.32E-08
Water	Co-58	4.10E-14	2.66E-12	1.06E-09
Water	Co-60	4.40E-11	2.86E-09	1.14E-06
Water	Cs-134	1.40E-10	9.09E-09	3.64E-06
Water	Cs-137	1.70E-10	1.10E-08	4.42E-06
Water	H-3	4.50E-16	2.92E-14	1.17E-11

Water	I-131	5.00E-13	3.25E-11	1.30E-08
Water	Mn-54	3.10E-13	2.01E-11	8.05E-09
Water	Ra-226	1.30E-13	8.44E-12	3.38E-09
Water	Sb-124	8.20E-13	5.32E-11	2.13E-08
Water	U-234	2.40E-12	1.56E-10	6.23E-08
Water	U-235	2.30E-12	1.49E-10	5.97E-08
Water	U-238	2.30E-12	1.49E-10	5.97E-08

1.1.6 Human health effects caused by ozone layer depletion (H,A)

All damage factors are expressed per kg release. The unit of damage is DALYs.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.26E-04	8.18E-03	3.27E+00
Air	CFC-11	1.05E-03	6.82E-02	2.73E+01
Air	CFC-113	9.48E-04	6.16E-02	2.46E+01
Air	CFC-114	8.95E-04	5.81E-02	2.32E+01
Air	CFC-115	4.21E-04	2.73E-02	1.09E+01
Air	CFC-12	8.63E-04	5.60E-02	2.24E+01
Air	HALON-1201	1.47E-03	9.55E-02	3.82E+01
Air	HALON-1202	1.32E-03	8.57E-02	3.43E+01
Air	HALON-1211	5.37E-03	3.49E-01	1.39E+02
Air	HALON-1301	1.26E-02	8.18E-01	3.27E+02
Air	HALON-2311	1.47E-04	9.55E-03	3.82E+00
Air	HALON-2401	2.63E-04	1.71E-02	6.83E+00
Air	HALON-2402	7.37E-03	4.79E-01	1.91E+02
Air	HCFC-123	1.47E-05	9.55E-04	3.82E-01
Air	HCFC-124	3.16E-05	2.05E-03	8.21E-01
Air	HCFC-141b	1.05E-04	6.82E-03	2.73E+00
Air	HCFC-142b	5.26E-05	3.42E-03	1.37E+00
Air	HCFC-22	4.21E-05	2.73E-03	1.09E+00
Air	HCFC-225ca	2.11E-05	1.37E-03	5.48E-01
Air	HCFC-225cb	2.11E-05	1.37E-03	5.48E-01
Air	methyl bromide	6.74E-04	4.38E-02	1.75E+01
Air	methyl chloride	2.11E-05	1.37E-03	5.48E-01
Air	carbontetrachloride	1.26E-03	8.18E-02	3.27E+01

1.2 Damage category Ecosystem Quality (H,A)

The Ecosystem Quality damages are specified as PDF*m^2*yr. PDF is short for Potentially Disappeared Fraction of Species. A damage of one means all species disappear from one m^2 during one year, or 10% of all species disappear from 10 m^2 during one year, or 10% of all species disappear from 1 m^2 during 10 years. Within the damage category Ecosystem Quality, special care is needed to avoid double counting when land-use is modelled. See the remarks under these damage categories.

1.2.1 Damage to Ecosystem Quality caused by ecotoxic emissions (H,A)

Fate analysis was done in EUSES. Pesticides that evaporate during application must be counted as air emissions. Pesticides that are accidentally sprayed in surface waters must be counted as water emissions. The remainder must be counted as soil emissions. The damage from pesticides in the agricultural soil as such (root zone) was deliberately excluded to avoid double counting with land-use. This means the damage factors in this list are based on secondary (leaching) emissions from the soil into surface and ground water and evaporation.

All damage factors are expressed per kg release. The unit of damage is PDF*m^2*yr.

Compartment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,2,3-trichlorobenzene	3.51E-02	6.84E-06	2.74E-03
Air	1,2,4-trichlorobenzene	2.54E-02	4.95E-06	1.98E-03
Air	1,3,5-trichlorobenzene	1.29E-01	2.51E-05	1.01E-02
Air	2,4-D	1.46E+00	2.85E-04	1.14E-01
Air	Arsenic	5.92E+02	1.15E-01	4.62E+01
Air	Atrazine	2.09E+02	4.07E-02	1.63E+01
Air	Azinphos-methyl	1.10E+04	2.14E+00	8.58E+02
Air	Bentazon	7.33E+00	1.43E-03	5.72E-01
Air	benzene	2.75E-03	5.36E-07	2.14E-04
Air	benzo(a)pyrene	1.42E+02	2.77E-02	1.11E+01
Air	Carbendazim	2.40E+03	4.68E-01	1.87E+02
Air	Cadmium	9.65E+03	1.88E+00	7.52E+02
Air	Chromium	4.13E+03	8.05E-01	3.22E+02
Air	Copper	1.46E+03	2.85E-01	1.14E+02
Air	di(2-ethylhexyl)phthalate	1.94E-03	3.78E-07	1.51E-04
Air	dibutylphthalate	1.13E-01	2.20E-05	8.81E-03
Air	Dichlorvos	1.61E+00	3.14E-04	1.26E-01
Air	2,3,7,8-TCDD Dioxin	1.32E+05	2.57E+01	1.03E+04
Air	Diquat-dibromide	2.39E+03	4.66E-01	1.86E+02
Air	Diuron	4.43E+03	8.64E-01	3.45E+02
Air	DNOC	8.19E+00	1.60E-03	6.39E-01
Air	fentin acetate	6.77E+02	1.32E-01	5.28E+01
Air	fluoranthene	4.37E-02	8.52E-06	3.41E-03
Air	gamma-HCH (Lindane)	2.16E+00	4.21E-04	1.68E-01
Air	Hexachlorobenzene	3.88E+01	7.56E-03	3.03E+00
Air	Mercury	8.29E+02	1.62E-01	6.46E+01
Air	Malathion	1.17E+02	2.28E-02	9.12E+00
Air	Maneb	3.84E+01	7.49E-03	2.99E+00
Air	Mecoprop	7.79E-02	1.52E-05	6.07E-03
Air	Metabenzthiazuron	3.07E+02	5.98E-02	2.39E+01
Air	metals	2.60E+02	5.07E-02	2.03E+01
Air	Metamitron	3.78E+01	7.37E-03	2.95E+00
Air	Metribuzin	4.92E+02	9.59E-02	3.84E+01
Air	Mevinphos	2.13E+03	4.15E-01	1.66E+02
Air	Monolinuron	1.06E+02	2.07E-02	8.27E+00
Air	Nickel	7.10E+03	1.38E+00	5.54E+02
Air	PAH's	7.80E-04	1.52E-07	6.08E-05
Air	Parathion	6.05E+01	1.18E-02	4.72E+00
Air	Lead	2.54E+03	4.95E-01	1.98E+02
Air	Polychlorobiphenyls	8.07E+01	1.57E-02	6.29E+00
Air	pentachlorophenol	1.33E+01	2.59E-03	1.04E+00
Air	Simazine	1.44E+03	2.81E-01	1.12E+02
Air	Thiram	2.26E+02	4.41E-02	1.76E+01
Air	toluene	2.40E-04	4.68E-08	1.87E-05
Air	Trifluralin	1.09E+00	2.12E-04	8.50E-02
Air	Zinc	2.89E+03	5.63E-01	2.25E+02
Water	1,2,3-trichlorobenzene	1.56E-01	3.04E-05	1.22E-02
Water	1,2,4-trichlorobenzene	1.39E-01	2.71E-05	1.08E-02
Water	1,3,5-trichlorobenzene	2.73E-01	5.32E-05	2.13E-02
Water	2,4-D	7.56E-02	1.47E-05	5.89E-03
Water	Arsenic	1.14E+01	2.22E-03	8.89E-01
Water	Atrazine	5.06E+01	9.86E-03	3.95E+00
Water	Azinphos-methyl	8.87E+02	1.73E-01	6.92E+01
Water	Bentazon	5.81E-02	1.13E-05	4.53E-03
Water	benzene	4.80E-02	9.36E-06	3.74E-03
Water	benzo(a)pyrene	3.68E+01	7.17E-03	2.87E+00
Water	Carbendazim	1.63E+02	3.18E-02	1.27E+01
Water	Cadmium	4.80E+02	9.36E-02	3.74E+01
Water	Chromium	6.87E+01	1.34E-02	5.36E+00
Water	Copper	1.47E+02	2.87E-02	1.15E+01
Water	di(2-ethylhexyl)phthalate	6.37E-01	1.24E-04	4.97E-02
Water	dibutylphthalate	1.62E+00	3.16E-04	1.26E-01
Water	Dichlorvos	1.81E-01	3.53E-05	1.41E-02
Water	dioxins (TEQ)	1.87E+05	3.65E+01	1.46E+04
Water	Diquat-dibromide	1.18E+02	2.30E-02	9.20E+00
Water	Diuron	2.31E+02	4.50E-02	1.80E+01
Water	DNOC	6.73E-01	1.31E-04	5.25E-02
Water	fentin acetate	7.85E+02	1.53E-01	6.12E+01
Water	fluoranthene	3.96E+00	7.72E-04	3.09E-01
Water	gamma-HCH (Lindane)	1.04E+01	2.03E-03	8.11E-01
Water	hexachlorobenzene	4.55E+01	8.87E-03	3.55E+00

Water	Mercury	1.97E+02	3.84E-02	1.54E+01
Water	Malathion	1.64E+02	3.20E-02	1.28E+01
Water	Maneb	6.23E-01	1.21E-04	4.86E-02
Water	Mecoprop	1.35E-02	2.63E-06	1.05E-03
Water	Metabenzthiazuron	1.43E+01	2.79E-03	1.12E+00
Water	Metamitron	3.77E-01	7.35E-05	2.94E-02
Water	Metribuzin	3.18E+00	6.20E-04	2.48E-01
Water	Mevinphos	6.73E+01	1.31E-02	5.25E+00
Water	Monolinuron	1.04E+01	2.03E-03	8.11E-01
Water	Nickel	1.43E+02	2.79E-02	1.12E+01
Water	PAH's	2.10E-03	4.09E-07	1.64E-04
Water	Parathion	2.48E+02	4.83E-02	1.93E+01
Water	Lead	7.39E+00	1.44E-03	5.76E-01
Water	Polychlorobiphenyls	2.58E+02	5.03E-02	2.01E+01
Water	pentachlorophenol	2.51E+01	4.89E-03	1.96E+00
Water	Simazine	6.03E+01	1.18E-02	4.70E+00
Water	Thiram	8.74E+02	1.70E-01	6.81E+01
Water	toluene	1.73E-01	3.37E-05	1.35E-02
Water	Trifluralin	7.80E+01	1.52E-02	6.08E+00
Water	Zinc	1.63E+01	3.18E-03	1.27E+00
Soil	1,2,3-trichlorobenzene (ind.)	2.41E+00	4.70E-04	1.88E-01
Soil	1,2,4-trichlorobenzene (ind.)	2.26E+00	4.41E-04	1.76E-01
Soil	1,3,5-trichlorobenzene (ind.)	1.19E+00	2.32E-04	9.28E-02
Soil	2,4-D (agr.)	1.27E-04	2.48E-08	9.90E-06
Soil	Arsenic (ind.)	6.10E+02	1.19E-01	4.76E+01
Soil	Atrazine (agr.)	1.49E-01	2.90E-05	1.16E-02
Soil	Azinphos-methyl (agr.)	3.55E-01	6.92E-05	2.77E-02
Soil	Bentazon (agr.)	1.66E-02	3.24E-06	1.29E-03
Soil	benzene (ind.)	4.97E-01	9.69E-05	3.88E-02
Soil	benzo(a)pyrene (ind.)	7.25E+03	1.41E+00	5.65E+02
Soil	Carbendazim (agr.)	2.34E+00	4.56E-04	1.82E-01
Soil	Cadmium (agr.)	3.01E+01	5.87E-03	2.35E+00
Soil	Cadmium (ind.)	9.94E+03	1.94E+00	7.75E+02
Soil	Chromium (ind.)	4.24E+03	8.27E-01	3.31E+02
Soil	Copper (ind.)	1.50E+03	2.92E-01	1.17E+02
Soil	di(2-ethylhexyl)phthalate(ind)	2.67E-02	5.20E-06	2.08E-03
Soil	dibutylphthalate (ind.)	1.14E+00	2.22E-04	8.89E-02
Soil	Dichlorvos (agr.)	7.52E-04	1.47E-07	5.86E-05
Soil	2,3,7,8-TCDD Dioxin (ind.)	2.09E+05	4.07E+01	1.63E+04
Soil	Diquat-dibromide (agr.)	6.84E-02	1.33E-05	5.33E-03
Soil	Diuron (agr.)	4.07E-02	7.93E-06	3.17E-03
Soil	DNOC (agr.)	6.17E-03	1.20E-06	4.81E-04
Soil	fentin acetate (agr.)	3.84E-01	7.49E-05	2.99E-02
Soil	fluoranthene (ind.)	8.00E+00	1.56E-03	6.24E-01
Soil	gamma-HCH (Lindane) (agr.)	1.38E+00	2.69E-04	1.08E-01
Soil	hexachlorobenzene (ind.)	9.96E+01	1.94E-02	7.77E+00
Soil	Mercury (ind.)	1.68E+03	3.27E-01	1.31E+02
Soil	Malathion (agr.)	2.79E-02	5.44E-06	2.18E-03
Soil	Maneb (agr.)	2.61E-01	5.09E-05	2.04E-02
Soil	Mecoprop (agr.)	2.79E-06	5.44E-10	2.18E-07
Soil	Metabenzthiazuron (agr.)	3.15E-01	6.14E-05	2.46E-02
Soil	Metamitron (agr.)	2.03E-04	3.96E-08	1.58E-05
Soil	Metribuzin (agr.)	4.91E-02	9.57E-06	3.83E-03
Soil	Mevinphos (agr.)	2.09E-01	4.07E-05	1.63E-02
Soil	Monolinuron (agr.)	4.38E-01	8.54E-05	3.42E-02
Soil	Nickel (ind.)	7.32E+03	1.43E+00	5.71E+02
Soil	Parathion (agr.)	3.24E-02	6.32E-06	2.53E-03
Soil	Lead (ind.)	1.29E+01	2.51E-03	1.01E+00
Soil	PCBs (ind.)	8.35E+02	1.63E-01	6.51E+01
Soil	pentachloorfenol (ind.)	2.51E+01	4.89E-03	1.96E+00
Soil	Simazine (agr.)	3.87E-01	7.54E-05	3.02E-02
Soil	Thiram (agr.)	9.96E-01	1.94E-04	7.77E-02
Soil	toluene (ind.)	6.79E-02	1.32E-05	5.29E-03
Soil	Trifluralin (agr.)	2.07E-02	4.04E-06	1.61E-03
Soil	Zinc (ind.)	2.98E+03	5.81E-01	2.32E+02

1.2.2 Damage to Ecosystem Quality caused by the combined effect of acidification and eutrophication (H,A)

Unfortunately no damage factors for emissions to water and soil could yet be calculated. We suggest to use the damage factors for air as a temporary, but crude solution.
The damage caused by fertilisers that are deliberately applied on agricultural soil is already included in the land-use damage factors, and should not be treated as an emission leading to eutrophication. The fertilisers that evaporate, or that are accidentally sprayed in surface waters should be counted as an emission.

All damage factors are based on kg emissions to air. The unit of damage is PDF*m^2*yr.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	15,57	3,04E-03	1,21E+00
Air	NO	8,789	1,71E-03	6,85E-01
Air	NO2	5,713	1,11E-03	4,45E-01
Air	NOx	5,713	1,11E-03	4,45E-01
Air	NOx (as NO2)	5,713	1,11E-03	4,45E-01
Air	SO2	1,041	2,03E-04	8,12E-02
Air	SO3	0,8323	1,62E-04	6,49E-02
Air	SOx	1,041	2,03E-04	8,12E-02
Air	SOx (as SO2)	1,041	2,03E-04	8,12E-02

1.2.3 Damage to Ecosystem Quality caused by land occupation and land conversion (H,A)

It is important to separate two cases:

- Land occupation
- Land conversion

The damage factors for occupation are per area [m^2] times the duration of the occupation [yr]. The effect of restoration of the area type to it's natural condition is not included here, but in the land conversion damage factors. Occupation is seen as a damage, because the area is prevented from restoring to its natural area. Typical examples of land occupation are:
Building new houses in an existing urban area, using a factory in an industrial area, agricultural production in an existing agricultural area. In most cases land is used that has already been converted long ago. In such cases conversion should not be considered.

The damage factors for conversions are per area [m^2]. Conversion factors should only be used if it is clear that a process results in the conversion of one area type into another. Examples are: mining operations in natural areas, expanding agricultural areas at the expense of natural areas, and dumping waste. The difference with the factors for occupation is the inclusion of a restoration time that has been set to 30 years as default. Other restoration times can be easily calculated by dividing the damage factors by 30 and multiplying them with the intended restoration time.

Conversions between agricultural and urban area types can also be modelled by subtracting the damage factors, but, as the damage factors can have considerable uncertainties, the result is unreliable. We suggest to use conversion data only for cases where natural areas are converted into non-natural area types.

1.2.3.1 COMPATIBILITY WITH ESU DATABASE

The ESU database, produced at the ETH Zurich, is one of the few large databases that has consistently included land-use data. Unfortunately no distinction is made between conversion and occupation, the two are always combined. This means a restoration time is always included, and this restoration time cannot be separated in a elegant way. In order to be able to use this large database damage factors: land-use II-III, land use II-IV, *land-use III-IV and land-use IV-IV have been estimated using the following (rather crude) assumptions:*

- *ESU land-use type II can be interpreted as near to natural area*
- *ESU land-use type III can be interpreted as green urban or rail areas. These are the not very intensively used areas*
- *ESU land-use type IV can be interpreted as continuos urban land*
- *ESU assumes a 5 year restoration time between type IV and III. In many cases an occupation time for industrial activities of 25 or 30 years is used. As a result the restoration time results in an overestimation of 20% for land-use II-IV. In the figure presented here the damage factor is thus lowered by 20%.*
- *After the conversion from Land-use II-IV the ESU database uses the factors II-III for the restoration time between type II and III. As we do not want to include these, in general they should be omitted. Unfortunately for processes like the production of hydropower this class is used in a different way and should thus be included*

Using the ESU database is thus not very straightforward, but with the factors presented here a reasonable first order approximation can be obtained, except for instance for processes that involve agricultural production and hydropower.

1.2.3.2 DOUBLE COUNTING

The damage factors are based on empirical observations of the number of plant species per area type. In such observations all effects of the area type are included. This means that also the effects of emissions are included. To avoid double counting these emissions, please observe the guidelines for pesticides and eutrophication.

The unit of damage of land occupation is PDF*m^2*yr.

Land-occupation	Damage factor	Normalised damage factor	Weighted damage factor
land use II-III	0.51	9.94E-05	3.98E-02
land use II-IV	0.96	1.87E-04	7.49E-02
land use III-IV	0.96	1.87E-04	7.49E-02
land use IV-IV	1.15	2.24E-04	8.97E-02
Occup. as Contin. urban land	1.15	2.24E-04	8.97E-02
Occup. as Convent. arable land	1.15	2.24E-04	8.97E-02
Occup. as Discont. urban land	0.96	1.87E-04	7.49E-02
Occup. as Forest land	0.11	2.14E-05	8.58E-03
Occup. as Green urban land	0.84	1.64E-04	6.55E-02
Occup. as Industrial area	0.84	1.64E-04	6.55E-02
Occup. as Intens. meadow land	1.13	2.20E-04	8.81E-02
Occup. as Organic arable land	1.09	2.12E-04	8.50E-02
Occup. as organic meadow land	1.02	1.99E-04	7.95E-02
Occup. as rail/ road area	0.84	1.64E-04	6.55E-02
Occup. as Integrated arable land	1.15	2.24E-04	8.97E-02
Occup. as less intens.meadow land	1.02	1.99E-04	7.95E-02

The unit of damage of land conversion is PDF*m^2.

Land conversion	Damage factor	Normalised damage factor	Weighted damage factor
Conv. to Continuous urban land	34.53	6.73E-03	2.69E+00
Conv. to Convent. arable land	34.38	6.70E-03	2.68E+00
Conv. to Discontinuous urban	28.73	5.60E-03	2.24E+00
Conv. to Green urban	25.16	4.90E-03	1.96E+00
Conv. to Industrial area	25.16	4.90E-03	1.96E+00
Conv. to Integr. arable land	34.38	6.70E-03	2.68E+00
Conv. to Intensive meadow	34.02	6.63E-03	2.65E+00
Conv. to Less intensive meadow	30.62	5.97E-03	2.39E+00
Conv. to Organic arable land	32.73	6.38E-03	2.55E+00
Conv. to Organic meadow	30.62	5.97E-03	2.39E+00
Conv. to rail/ road area	25.16	4.90E-03	1.96E+00

1.3 Damage category Resources (H,A)

The damages to resources are specified as MJ surplus energy. A damage of 1 means that due to a certain extraction further extraction of this resources in the future will require one additional MJ of energy, due to the lower resource concentration, or other unfavourable characteristics of the remaining reserves. The point in future has been chosen as the time at which 5 times the cumulative extraction of the resource before 1990 is extracted. The factor 5 is chosen arbitrarily, but after normalisation this has no further significance.

1.3.1 Damage to Resources caused by extraction of minerals (H,A)

The damage factors are expressed per kg of extracted metal or ore:

- “in ore” refers to the metal content in the ore, so 1kg iron (in ore) means one kg of pure iron
- “ore” refers to the ore. An average metal content is assumed to calculate these figures.

The unit of damage is MJ surplus energy per kg extracted material.

Minerals	Damage factor	Normalised damage factor	Weighted damage factor
aluminium (in ore)	2.38	2.83E-04	5.66E-02
bauxite	0.5	5.95E-05	1.19E-02
chromium (in ore)	0.9165	1.09E-04	2.18E-02
chromium (ore)	0.275	3.27E-05	6.54E-03
copper (in ore)	36.7	4.36E-03	8.73E-01
copper (ore)	0.415	4.93E-05	9.87E-03
iron (in ore)	0.051	6.06E-06	1.21E-03
iron (ore)	0.029	3.45E-06	6.90E-04
lead (in ore)	7.35	8.74E-04	1.75E-01
lead (ore)	0.368	4.38E-05	8.75E-03
manganese (in ore)	0.313	3.72E-05	7.44E-03
manganese (ore)	0.141	1.68E-05	3.35E-03
mercury (in ore)	165.5	1.97E-02	3.94E+00
molybdene (in ore)	41	4.88E-03	9.75E-01
molybdenum (ore)	0.041	4.88E-06	9.75E-04
nickel (in ore)	23.75	2.82E-03	5.65E-01
nickel (ore)	0.356	4.23E-05	8.47E-03
tin (in ore)	600	7.13E-02	1.43E+01
tin (ore)	0.06	7.13E-06	1.43E-03
tungsten (ore)	0.927	1.10E-04	2.20E-02
zinc (in ore)	4.09	4.86E-04	9.73E-02
zinc (ore)	0.164	1.95E-05	3.90E-03

1.3.2 Damage to Resources caused by extraction of fossil fuels (H,A)

The damage factors are expressing MJ surplus energy per kg of extracted fuel, or per m^3 of extracted gas, or per MJ extracted energy.

The unit of damage is MJ surplus energy.

Fossil fuels	Unit	Damage factor	Normalised damage factor	Weighted damage factor
coal	kg	0.252	3.00E-05	5.99E-03
coal ETH	kg	0.155	1.84E-05	3.69E-03
crude gas	kg	4.2	4.99E-04	9.99E-02
crude oil	kg	5.9	7.02E-04	1.40E-01
crude oil (feedstock)	kg	5.9	7.02E-04	1.40E-01
crude oil (resource)	MJ	1.44E-01	1.71E-05	3.42E-03
crude oil ETH	kg	6.13	7.29E-04	1.46E-01
crude oil IDEMAT	kg	6.15	7.31E-04	1.46E-01
energy from coal	MJ	8.59E-03	1.02E-06	2.04E-04
energy from natural gas	MJ	1.50E-01	1.78E-05	3.57E-03
energy from oil	MJ	0.144	1.71E-05	3.42E-03
hard coal (resource)	MJ	8.59E-03	1.02E-06	2.04E-04
natural gas	kg	4.55	5.41E-04	1.08E-01
natural gas (feedstock)	m3	5.25	6.24E-04	1.25E-01
natural gas (resource)	MJ	1.50E-01	1.78E-05	3.57E-03
natural gas (vol)	m3	5.49	6.53E-04	1.31E-01
natural gas ETH	m3	5.25	6.24E-04	1.25E-01
oil	kg	6.05	7.19E-04	1.44E-01

The energy content of the fuels listed above are:

Energy Content of Fossil fuels		[MJ / unit]
natural gas ETH	35	MJ / m3
crude oil IDEMAT	42.7	MJ / kg
coal ETH	18	MJ / kg
natural gas (feedstock)	35	MJ / m3
crude oil (feedstock)	41	MJ / kg
crude oil ETH	42.6	MJ / kg
natural gas (vol)	36.6	MJ / m3
coal	29.3	MJ / kg
crude oil	41	MJ / kg
natural gas	30.3	MJ / kg
oil	42	MJ / kg
crude gas	28	MJ / kg

2 Damage factors in the egalitarian perspective (E,E)

This annex lists the Eco-indicator 99 damage factors for the substance lists that can be found in most popular LCA databases. In this case the egalitarian perspective is used, combined with the matching weighting factors. Next to the damage factors two columns are added with the normalised and weighted damages. The normalisation factors and the weights are specified below.

	Normalisation	Weights
Human health	1.55E-02	300
Ecosystem Quality	5.13E+03	500
Resources	5.94E+03	200

Below the impact categories are listed per damage category.

2.1 Damage category Human Health (E,E)

The human health damages are specified in DALYs. This is short for Disability Adjusted Life Years. A damage of 1 means one life year of one individual is lost, or one person suffers four year from a disability with a weight of 0.25.

2.1.1 Carcinogenic effects on humans (E,E)

For the fate and exposure it is important to distinguish emissions to soil between emissions in industrial (ind.) or agricultural (agr.) soil. All emissions of pesticides are assumed to occur in agricultural soil, all other emissions are assumed to occur in industrial (or urban) soil. No direct emissions are assumed to occur in natural soil.
Fate factors are calculated with EUSES. Substances from IARC substances groups 1, 2a, 2b and 3 are included.

All damage factors are expressed per kg emission. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1,2-tetrachlorethane	3.72E-05	2.40E-03	7.20E-01
Air	1,1,2,2-tetrachlorethane	2.86E-04	1.85E-02	5.54E+00
Air	1,1,2-trichlorethane	1.10E-05	7.10E-04	2.13E-01
Air	1,1-dichloroethene	3.43E-06	2.21E-04	6.64E-02
Air	1,2-dibromoethane	2.60E-04	1.68E-02	5.03E+00
Air	1,2-dichloroethane	2.98E-05	1.92E-03	5.77E-01
Air	1,3-butadiene	1.58E-05	1.02E-03	3.06E-01
Air	1,4-dioxane	1.39E-07	8.97E-06	2.69E-03
Air	2,4,6-trichlorophenol	2.05E-06	1.32E-04	3.97E-02
Air	3-methylcholanthrene	1.67E-01	1.08E+01	3.23E+03
Air	acetaldehyde	2.16E-07	1.39E-05	4.18E-03
Air	acrylonitrile	1.69E-05	1.09E-03	3.27E-01
Air	Aldrin	1.93E-01	1.25E+01	3.74E+03
Air	alpha-hexachlorocyclohexan	3.00E-04	1.94E-02	5.81E+00
Air	Arsenic	2.46E-02	1.59E+00	4.76E+02
Air	Bis(chloromethyl)ether	7.48E-03	4.83E-01	1.45E+02
Air	benzene	2.50E-06	1.61E-04	4.84E-02
Air	benzo(a)anthracene	5.86E-02	3.78E+00	1.13E+03
Air	benzo(a)pyrene	3.98E-03	2.57E-01	7.70E+01
Air	benzotrichloride	6.60E-03	4.26E-01	1.28E+02
Air	benzylchloride	1.04E-05	6.71E-04	2.01E-01
Air	beta-chlorocyclohexan	9.99E-05	6.45E-03	1.93E+00
Air	bis(2-chlorethyl)ether	4.03E-05	2.60E-03	7.80E-01
Air	bromodichloromethane	8.76E-06	5.65E-04	1.70E-01
Air	Cadmium	1.35E-01	8.71E+00	2.61E+03
Air	Chromium (VI)	1.75E+00	1.13E+02	3.39E+04
Air	di(2-ethylhexyl)phthalate	3.38E-05	2.18E-03	6.54E-01
Air	dibenz(a)anthracene	3.10E+01	2.00E+03	6.00E+05
Air	dibutylphthalate	3.43E-03	2.21E-01	6.64E+01
Air	dichloromethane	4.36E-07	2.81E-05	8.44E-03
Air	Dichlorvos	3.15E-05	2.03E-03	6.10E-01
Air	Dieldrin	2.70E+01	1.74E+03	5.23E+05
Air	2,3,7,8-TCDD Dioxin	1.79E+02	1.15E+04	3.46E+06
Air	epichlorohydrin	3.02E-07	1.95E-05	5.85E-03
Air	ethylene oxide	1.83E-04	1.18E-02	3.54E+00
Air	formaldehyde	9.91E-07	6.39E-05	1.92E-02
Air	gamma-HCH (Lindane)	3.49E-04	2.25E-02	6.75E+00
Air	Hexachlorobenzene	8.25E-02	5.32E+00	1.60E+03
Air	hexachlorobutadiene	4.30E-05	2.77E-03	8.32E-01
Air	hexachloroethane	1.99E-05	1.28E-03	3.85E-01
Air	metals	5.20E-03	3.35E-01	1.01E+02
Air	methyl chloride	1.83E-05	1.18E-03	3.54E-01
Air	Nickel	2.35E-02	1.52E+00	4.55E+02
Air	Nickel-refinery-dust	4.74E-02	3.06E+00	9.17E+02
Air	Nickel-subsulfide	9.48E-02	6.12E+00	1.83E+03
Air	PAH's	1.70E-04	1.10E-02	3.29E+00
Air	particles diesel soot	9.78E-06	6.31E-04	1.89E-01
Air	Polychlorobiphenyls	1.97E-03	1.27E-01	3.81E+01
Air	pentachlorophenol	7.21E-03	4.65E-01	1.40E+02
Air	propyleneoxide	1.17E-05	7.55E-04	2.26E-01
Air	styrene	2.44E-08	1.57E-06	4.72E-04
Air	perchloroethylene	4.82E-07	3.11E-05	9.33E-03
Air	carbontetrachloride	8.38E-04	5.41E-02	1.62E+01
Air	trichloroethylene	7.95E-08	5.13E-06	1.54E-03
Air	chloroform	2.63E-05	1.70E-03	5.09E-01
Air	Trifluralin	1.10E-07	7.10E-06	2.13E-03
Air	vinyl chloride	2.09E-07	1.35E-05	4.05E-03
Water	1,1,1,2-tetrachlorethane	3.66E-05	2.36E-03	7.08E-01
Water	1,1,2,2-tetrachlorethane	2.78E-04	1.79E-02	5.38E+00
Water	1,1,2-trichlorethane	1.23E-05	7.94E-04	2.38E-01
Water	1,1-dichloroethene	5.88E-05	3.79E-03	1.14E+00
Water	1,2-dibromoethane	1.24E-03	8.00E-02	2.40E+01
Water	1,2-dichloroethane	2.98E-05	1.92E-03	5.77E-01
Water	1,3-butadiene	3.37E-04	2.17E-02	6.52E+00
Water	1,4-dioxane	9.21E-07	5.94E-05	1.78E-02
Water	2,4,6-trichlorophenol	1.05E-05	6.77E-04	2.03E-01
Water	3-methylcholanthrene	3.72E+01	2.40E+03	7.20E+05
Water	acetaldehyde	9.23E-07	5.95E-05	1.79E-02
Water	acrylonitrile	4.16E-05	2.68E-03	8.05E-01
Water	Aldrin	6.78E+00	4.37E+02	1.31E+05
Water	alpha-hexachlorocyclohexan	6.85E-03	4.42E-01	1.33E+02
Water	Arsenic	6.57E-02	4.24E+00	1.27E+03
Water	Bis(chloromethyl)ether	1.54E-02	9.94E-01	2.98E+02
Water	benzene	4.12E-06	2.66E-04	7.97E-02
Water	benzo(a)anthracene	6.58E-01	4.25E+01	1.27E+04
Water	benzo(a)pyrene	2.99	1.93E+02	5.79E+04
Water	benzotrichloride	9.46E-03	6.10E-01	1.83E+02
Water	benzylchloride	1.98E-05	1.28E-03	3.83E-01
Water	beta-chlorocyclohexan	5.75E-03	3.71E-01	1.11E+02
Water	bis(2-chlorethyl)ether	1.61E-04	1.04E-02	3.12E+00
Water	bromodichloromethane	9.36E-06	6.04E-04	1.81E-01
Water	Cadmium	7.12E-02	4.59E+00	1.38E+03
Water	Chromium (VI)	3.43E-01	2.21E+01	6.64E+03
Water	di(2-ethylhexyl)phthalate	6.64E-04	4.28E-02	1.29E+01
Water	dibenz(a)anthracene	4.07E+01	2.63E+03	7.88E+05
Water	dibutylphthalate	5.34E-02	3.45E+00	1.03E+03
Water	dichloromethane	4.79E-07	3.09E-05	9.27E-03
Water	Dichlorvos	1.17E-05	7.55E-04	2.26E-01
Water	Dieldrin	9.75E+01	6.29E+03	1.89E+06
Water	dioxins (TEQ)	2.02E+03	1.30E+05	3.91E+07
Water	epichloorhydrin	9.90E-07	6.39E-05	1.92E-02
Water	ethylene oxide	1.39E-04	8.97E-03	2.69E+00
Water	formaldehyde	4.97E-06	3.21E-04	9.62E-02
Water	gamma-HCH (Lindane)	4.16E-03	2.68E-01	8.05E+01
Water	hexachlorobenzene	1.25E-01	8.06E+00	2.42E+03
Water	hexachlorobutadiene	1.08E-04	6.97E-03	2.09E+00
Water	hexachloroethane	2.12E-05	1.37E-03	4.10E-01
Water	methyl chloride	1.78E-05	1.15E-03	3.45E-01
Water	Nickel	3.11E-02	2.01E+00	6.02E+02
Water	Nickel-subsulfide	1.00E-02	6.45E-01	1.94E+02

Water	Nickel-refinery-dust	5.02E-03	3.24E-01	9.72E+01
Water	PAH's	2.60E-03	1.68E-01	5.03E+01
Water	Polychlorobiphenyls	3.91E-02	2.52E+00	7.57E+02
Water	pentachlorophenol	2.29E-02	1.48E+00	4.43E+02
Water	propylene oxide	1.74E-05	1.12E-03	3.37E-01
Water	styrene	1.22E-06	7.87E-05	2.36E-02
Water	perchloroethylene	4.72E-07	3.05E-05	9.14E-03
Water	carbontetrachloride	8.29E-04	5.35E-02	1.60E+01
Water	trichloroethylene	7.97E-08	5.14E-06	1.54E-03
Water	chloroform	2.60E-05	1.68E-03	5.03E-01
Water	Trifluralin	7.93E-05	5.12E-03	1.53E+00
Water	vinyl chloride	2.84E-07	1.83E-05	5.50E-03
Soil	1,1,1,2-tetrachlorethane(ind.)	1.09E-03	7.03E-02	2.11E+01
Soil	1,1,2,2-tetrachlorethane(ind.)	7.54E-03	4.86E-01	1.46E+02
Soil	1,1,2-trichlorethane (ind.)	1.24E-04	8.00E-03	2.40E+00
Soil	1,1-dichloroethene (ind.)	5.57E-06	3.59E-04	1.08E-01
Soil	1,2-dibromoethane (ind.)	3.81E-03	2.46E-01	7.37E+01
Soil	1,2-dichloroethane (ind.)	4.58E-04	2.95E-02	8.86E+00
Soil	1,3-butadiene (ind.)	1.20E-05	7.74E-04	2.32E-01
Soil	1,4-dioxane (ind.)	3.10E-07	2.00E-05	6.00E-03
Soil	2,4,6-trichlorophenol (ind.)	2.76E-06	1.78E-04	5.34E-02
Soil	3-methylcholanthrene (ind.)	7.85E-01	5.06E+01	1.52E+04
Soil	acetaldehyde (ind.)	4.77E-07	3.08E-05	9.23E-03
Soil	acrylonitrile (ind.)	7.01E-05	4.52E-03	1.36E+00
Soil	Aldrin (agr.)	3.21E+01	2.07E+03	6.21E+05
Soil	alpha-hexachlorocyclohexan (agr.)	2.32E-02	1.50E+00	4.49E+02
Soil	Arsenic (ind.)	1.32E-02	8.52E-01	2.55E+02
Soil	Bis(chloromethyl)ether (ind.)	1.68E-02	1.08E+00	3.25E+02
Soil	benzene (ind.)	1.33E-05	8.58E-04	2.57E-01
Soil	benzo(a)anthracene (ind.)	1.60E-01	1.03E+01	3.10E+03
Soil	benzo(a)pyrene (ind.)	2.06E-03	1.33E-01	3.99E+01
Soil	benzotrichloride (ind.)	1.32E-01	8.52E+00	2.55E+03
Soil	benzylchloride (ind.)	4.16E-05	2.68E-03	8.05E-01
Soil	beta-chlorocyclohexan (agr.)	7.36E-03	4.75E-01	1.42E+02
Soil	bis(2-chloretyl)ether (ind.)	8.29E-05	5.35E-03	1.60E+00
Soil	bromodichloromethane (ind.)	7.82E-05	5.05E-03	1.51E+00
Soil	Cadmium (ind.)	3.98E-03	2.57E-01	7.70E+01
Soil	Chromium (ind.)	2.71E-01	1.75E+01	5.25E+03
Soil	di(2-ethylhexyl)phthalate(ind)	3.18E-07	2.05E-05	6.15E-03
Soil	dibenz(a)anthracene (ind.)	2.44E+01	1.57E+03	4.72E+05
Soil	dibutylphthalate (ind.)	6.00E-06	3.87E-04	1.16E-01
Soil	dichloromethane (ind.)	5.99E-06	3.86E-04	1.16E-01
Soil	Dichlorvos (agr.)	2.25E-05	1.45E-03	4.35E-01
Soil	Dieldrin (agr.)	4.17E+02	2.69E+04	8.07E+06
Soil	2,3,7,8-TCDD Dioxin (ind.)	7.06	4.55E+02	1.37E+05
Soil	epichloorhydrin (ind.)	1.30E-06	8.39E-05	2.52E-02
Soil	ethyleenoxide (ind.)	2.38E-03	1.54E-01	4.61E+01
Soil	formaldehyde (ind.)	1.83E-06	1.18E-04	3.54E-02
Soil	gamma-HCH (Lindane) (agr.)	8.64E-03	5.57E-01	1.67E+02
Soil	hexachlorobenzene (ind.)	1.47E-01	9.48E+00	2.85E+03
Soil	hexachlorobutadiene (ind.)	8.56E-04	5.52E-02	1.66E+01
Soil	hexachloroethane (ind.)	5.26E-04	3.39E-02	1.02E+01
Soil	methylchloride (ind.)	5.58E-04	3.60E-02	1.08E+01
Soil	Nickel (ind.)	3.94E-03	2.54E-01	7.63E+01
Soil	Nickel-refinery-dust (ind.)	6.37E-03	4.11E-01	1.23E+02
Soil	Nickel-subsulfide (ind.)	1.27E-02	8.19E-01	2.46E+02
Soil	PCBs (ind.)	2.04E-02	1.32E+00	3.95E+02
Soil	pentachloorfenol (ind.)	1.26E-05	8.13E-04	2.44E-01
Soil	propyleenoxide (ind.)	1.40E-04	9.03E-03	2.71E+00
Soil	styrene (ind.)	2.09E-08	1.35E-06	4.05E-04
Soil	perchloroethylene (ind.)	6.00E-06	3.87E-04	1.16E-01
Soil	carbontetrachloride (ind.)	3.99E-02	2.57E+00	7.72E+02
Soil	trichlorethene (ind.)	3.22E-07	2.08E-05	6.23E-03
Soil	chloroform (ind.)	4.12E-06	2.66E-04	7.97E-02
Soil	Trifluralin (agr.)	6.89E-05	4.45E-03	1.33E+00
Soil	vinylchloride (ind.)	7.67E-07	4.95E-05	1.48E-02
Soil	Trifluralin (agr.)	6.89E-05	4.45E-03	1.33E+00

2.1.2 Respiratory effects on humans caused by organic substances (E,E)

This impact category replaces more or less the summer smog category. Fate analysis is based on empirical data.
All damage factors are expressed per kg emitted substance. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.96E-08	1.26E-06	3.79E-04
Air	1,2,3-trimethyl benzene	2.72E-06	1.75E-04	5.26E-02
Air	1,2,4-trimethyl benzene	2.72E-06	1.75E-04	5.26E-02
Air	1,3,5-trimethyl benzene	2.98E-06	1.92E-04	5.77E-02
Air	1,3-butadiene	1.87E-06	1.21E-04	3.62E-02
Air	1-butene	2.30E-06	1.48E-04	4.45E-02
Air	1-butoxy propanol	9.36E-07	6.04E-05	1.81E-02
Air	1-hexene	1.87E-06	1.21E-04	3.62E-02
Air	1-methoxy 2-propanol	7.91E-07	5.10E-05	1.53E-02
Air	1-pentene	2.13E-06	1.37E-04	4.12E-02
Air	2,2-dimethyl butane	5.19E-07	3.35E-05	1.00E-02
Air	2,3-dimethyl butane	1.19E-06	7.68E-05	2.30E-02
Air	2-butoxy ethanol	9.36E-07	6.04E-05	1.81E-02
Air	2-ethoxy ethanol	8.34E-07	5.38E-05	1.61E-02
Air	2-hexanone	1.19E-06	7.68E-05	2.30E-02
Air	2-methoxy ethanol	6.47E-07	4.17E-05	1.25E-02
Air	2-methyl 1-butanol	8.51E-07	5.49E-05	1.65E-02
Air	2-methyl 1-butene	1.70E-06	1.10E-04	3.29E-02
Air	2-methyl 2-butanol	3.06E-07	1.97E-05	5.92E-03
Air	2-methyl 2-butene	1.79E-06	1.15E-04	3.46E-02
Air	2-methyl hexane	8.51E-07	5.49E-05	1.65E-02
Air	2-methyl pentane	9.36E-07	6.04E-05	1.81E-02
Air	2-pentanone	1.19E-06	7.68E-05	2.30E-02
Air	3,5-diethyl toluene	2.81E-06	1.81E-04	5.44E-02
Air	3,5-dimethyl ethyl benzene	2.81E-06	1.81E-04	5.44E-02
Air	3-hexanone	1.28E-06	8.26E-05	2.48E-02
Air	3-methyl 1-butanol	8.51E-07	5.49E-05	1.65E-02
Air	3-methyl 1-butene	1.45E-06	9.35E-05	2.81E-02
Air	3-methyl 2-butanol	7.91E-07	5.10E-05	1.53E-02
Air	3-methyl hexane	7.83E-07	5.05E-05	1.52E-02
Air	3-methyl pentane	1.02E-06	6.58E-05	1.97E-02
Air	3-pentanol	9.36E-07	6.04E-05	1.81E-02
Air	3-pentanone	8.51E-07	5.49E-05	1.65E-02
Air	acetaldehyde	1.36E-06	8.77E-05	2.63E-02
Air	acetic acid	2.13E-07	1.37E-05	4.12E-03
Air	acetone	2.04E-07	1.32E-05	3.95E-03
Air	propionaldehyde	1.70E-06	1.10E-04	3.29E-02
Air	alcohols	7.60E-07	4.90E-05	1.47E-02
Air	aldehydes	1.40E-06	9.03E-05	2.71E-02
Air	alkanes	7.50E-07	4.84E-05	1.45E-02
Air	alkenes	2.10E-06	1.35E-04	4.06E-02
Air	benzene	4.68E-07	3.02E-05	9.06E-03
Air	butane	7.57E-07	4.88E-05	1.47E-02
Air	butanol	1.36E-06	8.77E-05	2.63E-02
Air	butene	2.47E-06	1.59E-04	4.78E-02
Air	cis 1,2-dichloroethene	9.36E-07	6.04E-05	1.81E-02
Air	cis 2-butene	2.47E-06	1.59E-04	4.78E-02
Air	cis 2-hexene	2.30E-06	1.48E-04	4.45E-02
Air	cis 2-pentene	2.38E-06	1.54E-04	4.61E-02
Air	CxHy aromatic	2.10E-06	1.35E-04	4.06E-02
Air	CxHy chloro	3.50E-07	2.26E-05	6.77E-03
Air	CxHy halogenated	3.50E-07	2.26E-05	6.77E-03
Air	cyclohexane	6.21E-07	4.01E-05	1.20E-02
Air	cyclohexanol	9.36E-07	6.04E-05	1.81E-02
Air	cyclohexanone	6.47E-07	4.17E-05	1.25E-02
Air	decane	8.26E-07	5.33E-05	1.60E-02

Air	di-i-propyl ether	1.02E-06	6.58E-05	1.97E-02
Air	diacetone alcohol	5.62E-07	3.63E-05	1.09E-02
Air	dichloromethane	1.45E-07	9.35E-06	2.81E-03
Air	diethyl ether	1.02E-06	6.58E-05	1.97E-02
Air	dimethyl ether	3.74E-07	2.41E-05	7.24E-03
Air	dodecane	7.66E-07	4.94E-05	1.48E-02
Air	esters	3.70E-07	2.39E-05	7.16E-03
Air	ethane	2.64E-07	1.70E-05	5.11E-03
Air	ethane diol	8.26E-07	5.33E-05	1.60E-02
Air	ethanol	8.34E-07	5.38E-05	1.61E-02
Air	ethene	2.13E-06	1.37E-04	4.12E-02
Air	ethers	7.40E-07	4.77E-05	1.43E-02
Air	ethyl t-butyl ether	4.60E-07	2.97E-05	8.90E-03
Air	ethylacetate	4.60E-07	2.97E-05	8.90E-03
Air	ethylbenzene	1.53E-06	9.87E-05	2.96E-02
Air	acetylene	1.87E-07	1.21E-05	3.62E-03
Air	formaldehyde	1.11E-06	7.16E-05	2.15E-02
Air	formic acid	6.89E-08	4.45E-06	1.33E-03
Air	heptane	1.11E-06	7.16E-05	2.15E-02
Air	hexane	1.02E-06	6.58E-05	1.97E-02
Air	i-butane	6.64E-07	4.28E-05	1.29E-02
Air	i-butanol	8.09E-07	5.22E-05	1.57E-02
Air	i-butyraldehyde	1.11E-06	7.16E-05	2.15E-02
Air	i-pentane	8.51E-07	5.49E-05	1.65E-02
Air	i-propanol	2.98E-07	1.92E-05	5.77E-03
Air	i-propyl acetate	4.60E-07	2.97E-05	8.90E-03
Air	i-propyl benzene	1.11E-06	7.16E-05	2.15E-02
Air	isoprene	2.38E-06	1.54E-04	4.61E-02
Air	ketones	8.70E-07	5.61E-05	1.68E-02
Air	m-ethyl toluene	2.21E-06	1.43E-04	4.28E-02
Air	m-xylene	2.38E-06	1.54E-04	4.61E-02
Air	methane	1.28E-08	8.26E-07	2.48E-04
Air	methanol	2.81E-07	1.81E-05	5.44E-03
Air	methyl acetate	1.02E-07	6.58E-06	1.97E-03
Air	methyl chloride	1.11E-08	7.16E-07	2.15E-04
Air	methyl ethyl ketone	8.09E-07	5.22E-05	1.57E-02
Air	methyl formate	7.15E-08	4.61E-06	1.38E-03
Air	methyl i-butyl ketone	1.02E-06	6.58E-05	1.97E-02
Air	methyl i-propyl ketone	7.83E-07	5.05E-05	1.52E-02
Air	methyl propene	1.36E-06	8.77E-05	2.63E-02
Air	methyl t-butyl ether	3.32E-07	2.14E-05	6.43E-03
Air	methyl t-butyl ketone	6.98E-07	4.50E-05	1.35E-02
Air	n-butanol	1.36E-06	8.77E-05	2.63E-02
Air	n-butyl acetate	5.19E-07	3.35E-05	1.00E-02
Air	n-butyraldehyde	1.70E-06	1.10E-04	3.29E-02
Air	n-propanol	1.19E-06	7.68E-05	2.30E-02
Air	n-propyl acetate	6.21E-07	4.01E-05	1.20E-02
Air	n-propyl benzene	1.36E-06	8.77E-05	2.63E-02
Air	neopentane	3.74E-07	2.41E-05	7.24E-03
Air	NMVOC	1.28E-06	8.26E-05	2.48E-02
Air	nonane	8.51E-07	5.49E-05	1.65E-02
Air	o-ethyl toluene	1.96E-06	1.26E-04	3.79E-02
Air	o-xylene	2.30E-06	1.48E-04	4.45E-02
Air	octane	9.36E-07	6.04E-05	1.81E-02
Air	p-ethyl toluene	1.96E-06	1.26E-04	3.79E-02
Air	p-xylene	2.21E-06	1.43E-04	4.28E-02
Air	pentanal	1.62E-06	1.05E-04	3.14E-02
Air	pentane	8.51E-07	5.49E-05	1.65E-02
Air	propane	3.83E-07	2.47E-05	7.41E-03
Air	propane diol	1.02E-06	6.58E-05	1.97E-02
Air	propene	2.38E-06	1.54E-04	4.61E-02
Air	propanoic acide	3.23E-07	2.08E-05	6.25E-03
Air	s-butanol	8.51E-07	5.49E-05	1.65E-02
Air	s-butyl acetate	5.79E-07	3.74E-05	1.12E-02
Air	t-butanol	2.64E-07	1.70E-05	5.11E-03
Air	t-butyl acetate	1.36E-07	8.77E-06	2.63E-03
Air	perchloroethylene	6.21E-08	4.01E-06	1.20E-03
Air	toluene	1.36E-06	8.77E-05	2.63E-02
Air	trans 1,2-dichloroethene	8.43E-07	5.44E-05	1.63E-02
Air	trans 2-butene	2.47E-06	1.59E-04	4.78E-02
Air	trans 2-hexene	2.30E-06	1.48E-04	4.45E-02
Air	trans 2-pentene	2.38E-06	1.54E-04	4.61E-02
Air	trichloroethylene	6.98E-07	4.50E-05	1.35E-02
Air	chloroform	4.94E-08	3.19E-06	9.56E-04
Air	undecane	8.26E-07	5.33E-05	1.60E-02
Air	VOC	6.46E-07	4.17E-05	1.25E-02
Air	xylene	2.21E-06	1.43E-04	4.28E-02

2.1.3 Respiratory effects on humans caused by inorganic substances (E,E)

This impact category replaces more or less the winter smog category.
Fate analysis is based on empirical data.
All damage factors are expressed per kg emission. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	8.50E-05	5.48E-03	1.65E+00
Air	CO	7.31E-07	4.72E-05	1.41E-02
Air	dust (PM10)	3.75E-04	2.42E-02	7.26E+00
Air	dust (PM2.5)	7.00E-04	4.52E-02	1.35E+01
Air	TSP	1.10E-04	7.10E-03	2.13E+00
Air	NO	1.37E-04	8.84E-03	2.65E+00
Air	NO2	8.91E-05	5.75E-03	1.72E+00
Air	NOx	8.91E-05	5.75E-03	1.72E+00
Air	NOx (as NO2)	8.91E-05	5.75E-03	1.72E+00
Air	SO2	5.46E-05	3.52E-03	1.06E+00
Air	SO3	4.37E-05	2.82E-03	8.46E-01
Air	SOx	5.46E-05	3.52E-03	1.06E+00
Air	SOx (as SO2)	5.46E-05	3.52E-03	1.06E+00

2.1.4 Damages to human health caused by climate change (E,E)

Damage calculation was performed over a time scale of 200 years.
The IPCC equivalence factors have been modified. As damage is not linear dependent on the atmospheric lifetime, a separate damage calculation is made for CO2, CH4 en CH4:

- Gasses with an atmospheric lifetime below 20 years are assumed to behave like methane
- Gasses with an atmospheric lifetime between 20 and 100 years behave like CO2
- Gasses with an atmospheric lifetime oh more than 100 years are assumed to behave like N2O

This means that the IPCC equivalency factor table is split in three groups.
All damage factors are expressed per kg substance. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	methyl chloroform	-4.3E-05	-2.77E-03	-8.32E-01
Air	perfluorethane	2.00E-03	1.29E-01	3.87E+01
Air	trifluoroiodomethane	2.10E-07	1.35E-05	4.06E-03
Air	perfluormethane	1.40E-03	9.03E-02	2.71E+01
Air	CFC-11	2.20E-04	1.42E-02	4.26E+00
Air	CFC-113	6.30E-04	4.06E-02	1.22E+01
Air	CFC-12	1.40E-03	9.03E-02	2.71E+01
Air	carbon dioxide	2.10E-07	1.35E-05	4.06E-03
Air	methylene chloride	1.90E-06	1.23E-04	3.68E-02
Air	HALON-1301	-7.10E-03	-4.58E-01	-1.37E+02
Air	HCFC-123	6.60E-06	4.26E-04	1.28E-01

Air	HCFC-124	8.50E-05	5.48E-03	1.65E+00
Air	HCFC-141b	5.20E-05	3.35E-03	1.01E+00
Air	HCFC-142b	3.40E-04	2.19E-02	6.58E+00
Air	HCFC-22	2.80E-04	1.81E-02	5.42E+00
Air	HFC-125	5.80E-04	3.74E-02	1.12E+01
Air	HFC-134	2.10E-04	1.35E-02	4.06E+00
Air	HFC-134a	2.70E-04	1.74E-02	5.23E+00
Air	HFC-143	6.30E-05	4.06E-03	1.22E+00
Air	HFC-143a	7.80E-04	5.03E-02	1.51E+01
Air	HFC-152a	2.90E-05	1.87E-03	5.61E-01
Air	HFC-227ea	6.00E-04	3.87E-02	1.16E+01
Air	HFC-23	2.60E-03	1.68E-01	5.03E+01
Air	HFC-236fa	1.40E-03	9.03E-02	2.71E+01
Air	HFC-245ca	1.20E-04	7.74E-03	2.32E+00
Air	HFC-32	1.40E-04	9.03E-03	2.71E+00
Air	HFC-41	3.10E-05	2.00E-03	6.00E-01
Air	HFC-4310mee	2.70E-04	1.74E-02	5.23E+00
Air	methane	4.40E-06	2.84E-04	8.52E-02
Air	nitrous oxide	6.90E-05	4.45E-03	1.34E+00
Air	perfluorbutane	1.60E-03	1.03E-01	3.10E+01
Air	perfluorcyclobutane	1.90E-03	1.23E-01	3.68E+01
Air	perfluorhexane	1.60E-03	1.03E-01	3.10E+01
Air	perfluorpentane	1.70E-03	1.10E-01	3.29E+01
Air	perfluorpropane	1.60E-03	1.03E-01	3.10E+01
Air	sulphur hexafluoride	5.30E-03	3.42E-01	1.03E+02
Air	carbontetrachloride	-2.60E-04	-1.68E-02	-5.03E+00
Air	chloroform	8.40E-07	5.42E-05	1.63E-02

2.1.5 Human health effects caused by ionising radiation (E,E)

Fate-, and exposure models are based on studies for the French nuclear industry. All damage factors are based on a release of 1 Bequerel (Bq). The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	C-14	2.10E-10	1.35E-08	4.06E-06
Air	Co-58	4.30E-13	2.77E-11	8.32E-09
Air	Co-60	1.60E-11	1.03E-09	3.10E-07
Air	Cs-134	1.20E-11	7.74E-10	2.32E-07
Air	Cs-137	1.30E-11	8.39E-10	2.52E-07
Air	H-3	1.40E-14	9.03E-13	2.71E-10
Air	I-129	9.40E-10	6.06E-08	1.82E-05
Air	I-131	1.60E-13	1.03E-11	3.10E-09
Air	I-133	9.40E-15	6.06E-13	1.82E-10
Air	Kr-85	1.40E-16	9.03E-15	2.71E-12
Air	Pb-210	1.50E-12	9.68E-11	2.90E-08
Air	Po-210	1.50E-12	9.68E-11	2.90E-08
Air	Pu alpha	8.30E-11	5.35E-09	1.61E-06
Air	Pu-238	6.70E-11	4.32E-09	1.30E-06
Air	Ra-226	9.10E-13	5.87E-11	1.76E-08
Air	Rn-222	2.40E-14	1.55E-12	4.65E-10
Air	Th-230	4.50E-11	2.90E-09	8.71E-07
Air	U-234	9.70E-11	6.26E-09	1.88E-06
Air	U-235	2.10E-11	1.35E-09	4.06E-07
Air	U-238	8.20E-12	5.29E-10	1.59E-07
Air	Xe-133	1.40E-16	9.03E-15	2.71E-12
Water	Ag-110m	5.10E-13	3.29E-11	9.87E-09
Water	Co-58	4.10E-14	2.65E-12	7.94E-10
Water	Co-60	4.40E-11	2.84E-09	8.52E-07
Water	Cs-134	1.40E-10	9.03E-09	2.71E-06
Water	Cs-137	1.70E-10	1.10E-08	3.29E-06
Water	H-3	4.50E-16	2.90E-14	8.71E-12
Water	I-131	5.00E-13	3.23E-11	9.68E-09
Water	Mn-54	3.10E-13	2.00E-11	6.00E-09
Water	Ra-226	1.30E-13	8.39E-12	2.52E-09
Water	Sb-124	8.20E-13	5.29E-11	1.59E-08
Water	U-234	2.40E-12	1.55E-10	4.65E-08
Water	U-235	2.30E-12	1.48E-10	4.45E-08
Water	U-238	2.30E-12	1.48E-10	4.45E-08

2.1.6 Human health effects caused by ozone layer depletion (E,E)

All damage factors are expressed per kg release. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.26E-04	8.13E-03	2.44E+00
Air	CFC-11	1.05E-03	6.77E-02	2.03E+01
Air	CFC-113	9.48E-04	6.12E-02	1.83E+01
Air	CFC-114	8.95E-04	5.77E-02	1.73E+01
Air	CFC-115	4.21E-04	2.72E-02	8.15E+00
Air	CFC-12	8.63E-04	5.57E-02	1.67E+01
Air	HALON-1201	1.47E-03	9.48E-02	2.85E+01
Air	HALON-1202	1.32E-03	8.52E-02	2.55E+01
Air	HALON-1211	5.37E-03	3.46E-01	1.04E+02
Air	HALON-1301	1.26E-02	8.13E-01	2.44E+02
Air	HALON-2311	1.47E-04	9.48E-03	2.85E+00
Air	HALON-2401	2.63E-04	1.70E-02	5.09E+00
Air	HALON-2402	7.37E-03	4.75E-01	1.43E+02
Air	HCFC-123	1.47E-05	9.48E-04	2.85E-01
Air	HCFC-124	3.16E-05	2.04E-03	6.12E-01
Air	HCFC-141b	1.05E-04	6.77E-03	2.03E+00
Air	HCFC-142b	5.26E-05	3.39E-03	1.02E+00
Air	HCFC-22	4.21E-05	2.72E-03	8.15E-01
Air	HCFC-225ca	2.11E-05	1.36E-03	4.08E-01
Air	HCFC-225cb	2.11E-05	1.36E-03	4.08E-01
Air	methyl bromide	6.74E-04	4.35E-02	1.30E+01
Air	methyl chloride	2.11E-05	1.36E-03	4.08E-01
Air	carbontetrachloride	1.26E-03	8.13E-02	2.44E+01

2.2 Damage category Ecosystem Quality (E,E)

The Ecosystem Quality damages are specified as PDF*m^2*yr. PDF is short for Potentially Disappeared Fraction of Species. A damage of one means all species disappear from one m^2 during one year, or 10% of all species disappear from 10 m^2 during one year, or 10% of all species disappear from 1 m^2 during 10 years. Within the damage category Ecosystem Quality special care is needed to avoid double counting when land-use is modelled; see the remarks under these damage categories.

2.2.1 Damage to Ecosystem Quality caused by ecotoxic emissions (E,E)

Fate analysis was done in EUSES.
Pesticides that evaporate during application must be counted as air emissions. Pesticides that are accidentally sprayed in surface waters must be counted as water emissions. The remainder must be counted as soil emissions. The damage from pesticides in the agricultural soil as such (root zone) was deliberately excluded to avoid double counting with land-use. This means the damage factors in this list are based on secondary (leaching) emissions from the soil into surface and ground water and evaporation.

All damage factors are expressed per kg release. The unit of damage is PDF*m^2*yr.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,2,3-trichlorobenzene	3.51E-02	6.84E-06	3.42E-03
Air	1,2,4-trichlorobenzene	2.54E-02	4.95E-06	2.48E-03
Air	1,3,5-trichlorobenzene	1.29E-01	2.51E-05	1.26E-02
Air	2,4-D	1.46E+00	2.85E-04	1.42E-01
Air	Arsenic	5.92E+02	1.15E-01	5.77E+01

Air	Atrazine	2.09E+02	4.07E-02	2.04E+01
Air	Azinphos-methyl	1.10E+04	2.14E+00	1.07E+03
Air	Bentazon	7.33E+00	1.43E-03	7.14E-01
Air	benzene	2.75E-03	5.36E-07	2.68E-04
Air	benzo(a)pyrene	1.42E+02	2.77E-02	1.38E+01
Air	Carbendazim	2.40E+03	4.68E-01	2.34E+02
Air	Cadmium	9.65E+03	1.88E+00	9.41E+02
Air	Chromium	4.13E+03	8.05E-01	4.03E+02
Air	Copper	1.46E+03	2.85E-01	1.42E+02
Air	di(2-ethylhexyl)phthalate	1.94E-03	3.78E-07	1.89E-04
Air	dibutylphthalate	1.13E-01	2.20E-05	1.10E-02
Air	Dichlorvos	1.61E+00	3.14E-04	1.57E-01
Air	2,3,7,8-TCDD Dioxin	1.32E+05	2.57E+01	1.29E+04
Air	Diquat-dibromide	2.39E+03	4.66E-01	2.33E+02
Air	Diuron	4.43E+03	8.64E-01	4.32E+02
Air	DNOC	8.19E+00	1.60E-03	7.98E-01
Air	fentin acetate	6.77E+02	1.32E-01	6.60E+01
Air	fluoranthene	4.37E-02	8.52E-06	4.26E-03
Air	gamma-HCH (Lindane)	2.16E+00	4.21E-04	2.11E-01
Air	Hexachlorobenzene	3.88E+01	7.56E-03	3.78E+00
Air	Mercury	8.29E+02	1.62E-01	8.08E+01
Air	Malathion	1.17E+02	2.28E-02	1.14E+01
Air	Maneb	3.84E+01	7.49E-03	3.74E+00
Air	Mecoprop	7.79E-02	1.52E-05	7.59E-03
Air	Metabenzthiazuron	3.07E+02	5.98E-02	2.99E+01
Air	metals	2.60E+02	5.07E-02	2.53E+01
Air	Metamitron	3.78E+01	7.37E-03	3.68E+00
Air	Metribuzin	4.92E+02	9.59E-02	4.80E+01
Air	Mevinphos	2.13E+03	4.15E-01	2.08E+02
Air	Monolinuron	1.06E+02	2.07E-02	1.03E+01
Air	Nickel	7.10E+03	1.38E+00	6.92E+02
Air	PAH's	7.80E-04	1.52E-07	7.60E-05
Air	Parathion	6.05E+01	1.18E-02	5.90E+00
Air	Lead	2.54E+03	4.95E-01	2.48E+02
Air	Polychlorobiphenyls	8.07E+01	1.57E-02	7.87E+00
Air	pentachlorophenol	1.33E+01	2.59E-03	1.30E+00
Air	Simazine	1.44E+03	2.81E-01	1.40E+02
Air	Thiram	2.26E+02	4.41E-02	2.20E+01
Air	toluene	2.40E-04	4.68E-08	2.34E-05
Air	Trifluralin	1.09E+00	2.12E-04	1.06E-01
Air	Zinc	2.89E+03	5.63E-01	2.82E+02
Water	1,2,3-trichlorobenzene	1.56E-01	3.04E-05	1.52E-02
Water	1,2,4-trichlorobenzene	1.39E-01	2.71E-05	1.35E-02
Water	1,3,5-trichlorobenzene	2.73E-01	5.32E-05	2.66E-02
Water	2,4-D	7.56E-02	1.47E-05	7.37E-03
Water	Arsenic	1.14E+01	2.22E-03	1.11E+00
Water	Atrazine	5.06E+01	9.86E-03	4.93E+00
Water	Azinphos-methyl	8.87E+02	1.73E-01	8.65E+01
Water	Bentazon	5.81E-02	1.13E-05	5.66E-03
Water	benzene	4.80E-02	9.36E-06	4.68E-03
Water	benzo(a)pyrene	3.68E+01	7.17E-03	3.59E+00
Water	Carbendazim	1.63E+02	3.18E-02	1.59E+01
Water	Cadmium	4.80E+02	9.36E-02	4.68E+01
Water	Chromium	6.87E+01	1.34E-02	6.70E+00
Water	Copper	1.47E+02	2.87E-02	1.43E+01
Water	di(2-ethylhexyl)phthalate	6.37E-01	1.24E-04	6.21E-02
Water	dibutylphthalate	1.62E+00	3.16E-04	1.58E-01
Water	Dichlorvos	1.81E-01	3.53E-05	1.76E-02
Water	dioxins (TEQ)	1.87E+05	3.65E+01	1.82E+04
Water	Diquat-dibromide	1.18E+02	2.30E-02	1.15E+01
Water	Diuron	2.31E+02	4.50E-02	2.25E+01
Water	DNOC	6.73E-01	1.31E-04	6.56E-02
Water	fentin acetate	7.85E+02	1.53E-01	7.65E+01
Water	fluoranthene	3.96E+00	7.72E-04	3.86E-01
Water	gamma-HCH (Lindane)	1.04E+01	2.03E-03	1.01E+00
Water	hexachlorobenzene	4.55E+01	8.87E-03	4.43E+00
Water	Mercury	1.97E+02	3.84E-02	1.92E+01
Water	Malathion	1.64E+02	3.20E-02	1.60E+01
Water	Maneb	6.23E-01	1.21E-04	6.07E-02
Water	Mecoprop	1.35E-02	2.63E-06	1.32E-03
Water	Metabenzthiazuron	1.43E+01	2.79E-03	1.39E+00
Water	Metamitron	3.77E-01	7.35E-05	3.67E-02
Water	Metribuzin	3.18E+00	6.20E-04	3.10E-01
Water	Mevinphos	6.73E+01	1.31E-02	6.56E+00
Water	Monolinuron	1.04E+01	2.03E-03	1.01E+00
Water	Nickel	1.43E+02	2.79E-02	1.39E+01
Water	PAH's	2.10E-03	4.09E-07	2.05E-04
Water	Parathion	2.48E+02	4.83E-02	2.42E+01
Water	Lead	7.39E+00	1.44E-03	7.20E-01
Water	Polychlorobiphenyls	2.58E+02	5.03E-02	2.51E+01
Water	pentachlorophenol	2.51E+01	4.89E-03	2.45E+00
Water	Simazine	6.03E+01	1.18E-02	5.88E+00
Water	Thiram	8.74E+02	1.70E-01	8.52E+01
Water	toluene	1.73E-01	3.37E-05	1.69E-02
Water	Trifluralin	7.80E+01	1.52E-02	7.60E+00
Water	Zinc	1.63E+01	3.18E-03	1.59E+00
Soil	1,2,3-trichlorobenzene (ind.)	2.41E+00	4.70E-04	2.35E-01
Soil	1,2,4-trichlorobenzene (ind.)	2.26E+00	4.41E-04	2.20E-01
Soil	1,3,5-trichlorobenzene (ind.)	1.19E+00	2.32E-04	1.16E-01
Soil	2,4-D (agr.)	1.27E-04	2.48E-08	1.24E-05
Soil	Arsenic (ind.)	6.10E+02	1.19E-01	5.95E+01
Soil	Atrazine (agr.)	1.49E-01	2.90E-05	1.45E-02
Soil	Azinphos-methyl (agr.)	3.55E-01	6.92E-05	3.46E-02
Soil	Bentazon (agr.)	1.66E-02	3.24E-06	1.62E-03
Soil	benzene (ind.)	4.97E-01	9.69E-05	4.84E-02
Soil	benzo(a)pyrene (ind.)	7.25E+03	1.41E+00	7.07E+02
Soil	Carbendazim (agr.)	2.34E+00	4.56E-04	2.28E-01
Soil	Cadmium (agr.)	3.01E+01	5.87E-03	2.93E+00
Soil	Cadmium (ind.)	9.94E+03	1.94E+00	9.69E+02
Soil	Chromium (ind.)	4.24E+03	8.27E-01	4.13E+02
Soil	Copper (ind.)	1.50E+03	2.92E-01	1.46E+02
Soil	di(2-ethylhexyl)phthalate(ind)	2.67E-02	5.20E-06	2.60E-03
Soil	dibutylphthalate (ind.)	1.14E+00	2.22E-04	1.11E-01
Soil	Dichlorvos (agr.)	7.52E-04	1.47E-07	7.33E-05
Soil	2,3,7,8-TCDD Dioxin (ind.)	2.09E+05	4.07E+01	2.04E+04
Soil	Diquat-dibromide (agr.)	6.84E-02	1.33E-05	6.67E-03
Soil	Diuron (agr.)	4.07E-02	7.93E-06	3.97E-03
Soil	DNOC (agr.)	6.17E-03	1.20E-06	6.01E-04
Soil	fentin acetate (agr.)	3.84E-01	7.49E-05	3.74E-02
Soil	fluoranthene (ind.)	8.00E+00	1.56E-03	7.80E-01
Soil	gamma-HCH (Lindane) (agr.)	1.38E+00	2.69E-04	1.35E-01
Soil	hexachlorobenzene (ind.)	9.96E+01	1.94E-02	9.71E+00
Soil	Mercury (ind.)	1.68E+03	3.27E-01	1.64E+02
Soil	Malathion (agr.)	2.79E-02	5.44E-06	2.72E-03
Soil	Maneb (agr.)	2.61E-01	5.09E-05	2.54E-02
Soil	Mecoprop (agr.)	2.79E-06	5.44E-10	2.72E-07
Soil	Metabenzthiazuron (agr.)	3.15E-01	6.14E-05	3.07E-02
Soil	Metamitron (agr.)	2.03E-04	3.96E-08	1.98E-05
Soil	Metribuzin (agr.)	4.91E-02	9.57E-06	4.79E-03
Soil	Mevinphos (agr.)	2.09E-01	4.07E-05	2.04E-02
Soil	Monolinuron (agr.)	4.38E-01	8.54E-05	4.27E-02
Soil	Nickel (ind.)	7.32E+03	1.43E+00	7.13E+02
Soil	Parathion (agr.)	3.24E-02	6.32E-06	3.16E-03
Soil	Lead (ind.)	1.29E+01	2.51E-03	1.26E+00
Soil	PCBs (ind.)	8.35E+02	1.63E-01	8.14E+01
Soil	pentachloorfenol (ind.)	2.51E+01	4.89E-03	2.45E+00
Soil	Simazine (agr.)	3.87E-01	7.54E-05	3.77E-02
Soil	Thiram (agr.)	9.96E-01	1.94E-04	9.71E-02
Soil	toluene (ind.)	6.79E-02	1.32E-05	6.62E-03
Soil	Trifluralin (agr.)	2.07E-02	4.04E-06	2.02E-03
Soil	Zinc (ind.)	2.98E+03	5.81E-01	2.90E+02

2.2.2 Damage to Ecosystem Quality caused by the combined effect of acidification and eutrophication (E,E)

Unfortunately no damage factors for emissions to water and soil could yet be calculated. We suggest to use the damage factors for air as a temporary, but crude solution.

The damage caused by fertilisers that are deliberately applied on agricultural soil is already included in the land-use damage factors, and should not be treated as an emission leading to eutrophication. The fertilisers that evaporate, or that are accidentally sprayed in surface waters should be counted as an emission.

All damage factors are based on kg emissions to air. The unit of damage is PDF*m^2*yr.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	15.57	3.04E-03	1.52E+00
Air	NO	8.789	1.71E-03	8.57E-01
Air	NO2	5.713	1.11E-03	5.57E-01
Air	NOx	5.713	1.11E-03	5.57E-01
Air	NOx (as NO2)	5.713	1.11E-03	5.57E-01
Air	SO2	1.041	2.03E-04	1.01E-01
Air	SO3	0.8323	1.62E-04	8.11E-02
Air	SOx	1.041	2.03E-04	1.01E-01
Air	SOx (as SO2)	1.041	2.03E-04	1.01E-01

2.2.3 Damage to ecosystem quality caused by land occupation and land conversion (E,E)

It is important to separate two cases:

- Land occupation
- Land conversion

The damage factors for occupation are per area [m^2] times the duration of the occupation [yr]. The effect of restoration of the area type to it's natural condition is not included here, but in the land conversion damage factors. Occupation is seen as a damage, because the area is prevented from restoring to its natural area. Typical examples of land occupation are:
Building new houses in an existing urban area, using a factory in an industrial area, agricultural production in an existing agricultural area. In most cases land is used that has already been converted long ago. In such cases conversion should not be considered.

The damage factors for conversions are per area [m^2]. Conversion factors should only be used if it is clear that a process results in the conversion of one area type into another. Examples are: mining operations in natural areas, expanding agricultural areas at the expense of natural areas, and dumping waste. The difference with the factors for occupation is the inclusion of a restoration time that has been set to 30 years as default. Other restoration times can be easily calculated by dividing the damage factors by 30 and multiplying them with the intended restoration time.

Conversions between agricultural and urban area types can also be modelled by subtracting the damage factors, but, as the damage factors can have considerable uncertainties, the result is unreliable. We suggest to use conversion data only for cases where natural areas are converted into non-natural area types.

2.2.3.1 COMPATIBILITY WITH ESU DATABASE

The ESU database, produced at the ETH Zurich, is one of the few large databases that has consistently included land-use data. Unfortunately no distinction is made between conversion and occupation, the two are always combined. This means a restoration time is always included, and this restoration time cannot be separated in a elegant way. In order to be able to use this large database damage factors: land-use II-III, land use II-IV, land-use III-IV and land-use IV-IV have been estimated using the following (rather crude) assumptions:

- *ESU land-use type II can be interpreted as near to natural area*
- *ESU land-use type III can be interpreted as green urban or rail areas. These are the not very intensively used areas*
- *ESU land-use type IV can be interpreted as continuos urban land*
- *ESU assumes a 5 year restoration time between type IV and III. In many cases an occupation time for industrial activities of 25 or 30 years is used. As a result the restoration time results in an overestimation of 20% for land-use II-IV. In the figure presented here the damage factor is thus lowered by 20%.*
- *After the conversion from Land-use II-IV the ESU database uses the factors II-III for the restoration time between type II and III. As we do not want to include these, in general they should be omitted. Unfortunately for processes like the production of hydropower this class is used in a different way and should thus be included*

Using the ESU database is thus not very straightforward, but with the factors presented here a reasonable first order approximation can be obtained, except for instance for processes that involve agricultural production and hydropower.

2.2.3.2 DOUBLE COUNTING

The damage factors are based on empirical observations of the number of plant species per area type. In such observations all effects of the area type are included. This means that also the effects of emissions are included. To avoid double counting these emissions, please observe the guidelines for pesticides and eutrophication.

The unit of damage of land occupation is PDF*m^2*yr.

Land-occupation	Damage factor	Normalised damage factor	Weighted damage factor
land use II-III	0.51	9.94E-05	4.97E-02
land use II-IV	0.96	1.87E-04	9.36E-02
land use III-IV	0.96	1.87E-04	9.36E-02
land use IV-IV	1.15	2.24E-04	1.12E-01
Occup. as Contin. urban land	1.15	2.24E-04	1.12E-01
Occup. as Convent. arable land	1.15	2.24E-04	1.12E-01
Occup. as Discont. urban land	0.96	1.87E-04	9.36E-02
Occup. as Forest land	0.11	2.14E-05	1.07E-02
Occup. as Green urban land	0.84	1.64E-04	8.19E-02
Occup. as Industrial area	0.84	1.64E-04	8.19E-02
Occup. as Intens. meadow land	1.13	2.20E-04	1.10E-01
Occup. as Organic arable land	1.09	2.12E-04	1.06E-01
Occup. as organic meadow land	1.02	1.99E-04	9.94E-02
Occup. as rail/ road area	0.84	1.64E-04	8.19E-02
Occup. as Integrated arable land	1.15	2.24E-04	1.12E-01
Occup. as less intens.meadow land	1.02	1.99E-04	9.94E-02

The unit of damage of land conversion is PDF*m^2.

Land conversion	Damage factor	Normalised damage factor	Weighted damage factor
Conv. to Continuous urban land	34.53	6.73E-03	3.37E+00
Conv. to Convent. arable land	34.38	6.70E-03	3.35E+00
Conv. to Discontinuous urban	28.73	5.60E-03	2.80E+00
Conv. to Green urban	25.16	4.90E-03	2.45E+00
Conv. to Industrial area	25.16	4.90E-03	2.45E+00
Conv. to Integr. arable land	34.38	6.70E-03	3.35E+00
Conv. to Intensive meadow	34.02	6.63E-03	3.32E+00
Conv. to Less intensive meadow	30.62	5.97E-03	2.98E+00
Conv. to Organic arable land	32.73	6.38E-03	3.19E+00
Conv. to Organic meadow	30.62	5.97E-03	2.98E+00
Conv. to rail/ road area	25.16	4.90E-03	2.45E+00

2.3 Damage category Resources (E,E)

The damages to resources are specified as MJ surplus energy. A damage of 1 means that due to a certain extraction further extraction of this resources in the future will require one additional MJ of energy, due to the lower resource concentration, or other unfavourable characteristics of the remaining reserves. The point in future has been chosen as the time at which 5 times the cumulative extraction of the resource before 1990 is extracted. The factor 5 is chosen arbitrarily, but after normalisation this has no further significance.

2.3.1 Damage to Resources caused by extraction of minerals (E,E)

The damage factors are per kg of extracted metal or ore:

- "in ore" refers to the metal content in the ore, so 1kg iron (in ore) means one kg of pure iron
- "ore" refers to the ore. An average metal content is assumed to calculate these figures.

The unit of damage is MJ surplus energy per kg of extracted material

Minerals	Damage factor	Normalised damage factor	Weighted damage factor
aluminium (in ore)	2.38	4.01E-04	8.01E-02
bauxite	0.5	8.42E-05	1.68E-02
chromium (in ore)	0.9165	1.54E-04	3.09E-02
chromium (ore)	0.275	4.63E-05	9.26E-03
copper (in ore)	36.7	6.18E-03	1.24E+00
copper (ore)	0.415	6.99E-05	1.40E-02
iron (in ore)	0.051	8.59E-06	1.72E-03
iron (ore)	0.029	4.88E-06	9.76E-04
lead (in ore)	7.35	1.24E-03	2.47E-01
lead (ore)	0.368	6.20E-05	1.24E-02
manganese (in ore)	0.313	5.27E-05	1.05E-02
manganese (ore)	0.141	2.37E-05	4.75E-03
mercury (in ore)	165.5	2.79E-02	5.57E+00
molybdene (in ore)	41	6.90E-03	1.38E+00
molybdenum (ore)	0.041	6.90E-06	1.38E-03
nickel (in ore)	23.75	4.00E-03	8.00E-01
nickel (ore)	0.356	5.99E-05	1.20E-02
tin (in ore)	600	1.01E-01	2.02E+01
tin (ore)	0.06	1.01E-05	2.02E-03
tungsten (ore)	0.927	1.56E-04	3.12E-02
zinc (in ore)	4.09	6.89E-04	1.38E-01
zinc (ore)	0.164	2.76E-05	5.52E-03

2.3.2 Damage to Resources caused by extraction of fossil fuels (E,E)

The damage factors are expressing MJ surplus energy per kg of extracted fuel, or per m^3 of extracted gas, or per MJ extracted energy.

The unit of damage is MJ surplus energy.

Fossil fuels	Unit	Damage factor	Normalised damage factor	Weighted damage factor
coal	kg	2.04	3.43E-04	6.87E-02
coal ETH	kg	1.25	2.10E-04	4.21E-02
crude gas	kg	2.49	4.19E-04	8.38E-02
crude oil	kg	3.4	5.72E-04	1.14E-01
crude oil (feedstock)	kg	3.4	5.72E-04	1.14E-01
crude oil (resource)	MJ	8.30E-02	1.40E-05	2.79E-03
crude oil ETH	kg	3.54	5.96E-04	1.19E-01
crude oil IDEMAT	kg	3.54	5.96E-04	1.19E-01
energy from coal	MJ	6.96E-02	1.17E-05	2.34E-03
energy from lignite	MJ	6.10E-02	1.03E-05	2.05E-03
energy from natural gas	MJ	8.90E-02	1.50E-05	3.00E-03
energy from oil	MJ	8.30E-02	1.40E-05	2.79E-03
hard coal (resource)	MJ	6.96E-02	1.17E-05	2.34E-03
lignite	kg	0.61	1.03E-04	2.05E-02
lignite (resource)	MJ	6.10E-02	1.03E-05	2.05E-03
lignite ETH	kg	0.488	8.22E-05	1.64E-02
natural gas	kg	2.7	4.55E-04	9.09E-02
natural gas (feedstock)	m3	3.12	5.25E-04	1.05E-01
natural gas (resource)	MJ	0.089	1.50E-05	3.00E-03
natural gas (vol)	m3	3.26	5.49E-04	1.10E-01
natural gas ETH	m3	3.12	5.25E-04	1.05E-01
oil	kg	3.49	5.88E-04	1.18E-01

The energy content of the fuels listed above are:

Energy Content of Fossil fuels		[MJ / unit]
natural gas ETH	35	MJ / m3
crude oil IDEMAT	42.7	MJ / kg
coal ETH	18	MJ / kg
lignite ETH	8	kg
natural gas (feedstock)	35	MJ / m3
lignite	10	kg
crude oil (feedstock)	41	MJ / kg
crude oil ETH	42.6	MJ / kg
natural gas (vol)	36.6	MJ / m3
coal	29.3	MJ / kg
crude oil	41	MJ / kg
natural gas	30.3	MJ / kg
oil	42	MJ / kg
crude gas	28	MJ / kg

3 Damage factors in the individualist perspective (I,I)

This annex lists the Eco-indicator 99 damage factors for the substance lists that can be found in most popular LCA databases. In this case the individualist perspective is used, combined with the matching weighting factors. Next to the damage factors two columns are added with the normalised and weighted damages. The normalisation factors and the weights are specified below.

	Normalisation	Weights
Human health	8.25E-03	550
Ecosystem Quality	4.51E+03	250
Resources	1.48E+02	200

Below the impact categories are listed per damage category.

3.1 Damage category Human Health (I,I)

The human health damages are specified in DALYs. This is short for Disability Adjusted Life Years. In the individualis perspective, DALY calculations are based on age weighting.

3.1.1 Carcinogenic effects on humans (I,I)

For the fate and exposure it is important to distinguish emissions to soil between emissions in industrial (ind.) or agricultural (agr.) soil. All emissions of pesticides are assumed to occur in agricultural soil, all other emissions are assumed to occur in industrial (or urban) soil. No direct emissions are assumed to occur in natural soil.
Fate factors are calculated with EUSES. Substances from IARC substances groups 1are included.

All damage factors are expressed per kg emission. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	Arsenic	1.00E-03	1.21E-01	6.67E+01
Air	Bis(chloromethyl)ether	4.83E-03	5.85E-01	3.22E+02
Air	benzene	1.58E-06	1.92E-04	1.05E-01
Air	Cadmium	1.03E-02	1.25E+00	6.87E+02
Air	Chromium (VI)	1.50E-02	1.82E+00	1.00E+03
Air	metals	2.20E-04	2.67E-02	1.47E+01
Air	Nickel	6.79E-03	8.23E-01	4.53E+02
Air	Nickel-refinery-dust	2.22E-03	2.69E-01	1.48E+02
Air	Nickel-subsulfide	4.44E-03	5.38E-01	2.96E+02
Air	vinyl chloride	1.34E-07	1.62E-05	8.93E-03
Water	Arsenic	3.42E-02	4.15E+00	2.28E+03
Water	Bis(chloromethyl)ether	9.96E-03	1.21E+00	6.64E+02
Water	benzene	2.45E-06	2.97E-04	1.63E-01
Water	Cadmium	3.73E-02	4.52E+00	2.49E+03
Water	Chromium (VI)	1.79E-01	2.17E+01	1.19E+04
Water	Nickel	1.63E-02	1.98E+00	1.09E+03
Water	Nickel-subsulfide	2.65E-02	3.21E+00	1.77E+03
Water	Nickel-refinery-dust	5.29E-02	6.41E+00	3.53E+03
Water	vinyl chloride	1.73E-07	2.10E-05	1.15E-02
Soil	Arsenic (ind.)	7.28E-04	8.82E-02	4.85E+01
Soil	Bis(chloromethyl)ether (ind.)	1.08E-02	1.31E+00	7.20E+02
Soil	benzene (ind.)	8.35E-06	1.01E-03	5.57E-01
Soil	Cadmium (ind.)	6.09E-04	7.38E-02	4.06E+01
Soil	Chromium (ind.)	4.13E-03	5.01E-01	2.75E+02
Soil	Nickel (ind.)	3.27E-04	3.96E-02	2.18E+01
Soil	Nickel-refinery-dust (ind.)	5.31E-04	6.44E-02	3.54E+01
Soil	Nickel-subsulfide (ind.)	1.06E-03	1.28E-01	7.07E+01
Soil	vinylchloride (ind.)	4.92E-07	5.96E-05	3.28E-02

3.1.2 Respiratory effects on humans caused by organic substances (I,I)

This impact category replaces more or less the summer smog category. Fate analysis is based on empirical data.
All damage factors are expressed per kg emitted substance. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.82E-08	2.21E-06	1.21E-03
Air	1,2,3-trimethyl benzene	2.53E-06	3.07E-04	1.69E-01
Air	1,2,4-trimethyl benzene	2.53E-06	3.07E-04	1.69E-01
Air	1,3,5-trimethyl benzene	2.77E-06	3.36E-04	1.85E-01
Air	1,3-butadiene	1.74E-06	2.11E-04	1.16E-01
Air	1-butene	2.14E-06	2.59E-04	1.43E-01
Air	1-butoxy propanol	8.70E-07	1.05E-04	5.80E-02
Air	1-hexene	1.74E-06	2.11E-04	1.16E-01
Air	1-methoxy 2-propanol	7.36E-07	8.92E-05	4.91E-02
Air	1-pentene	1.98E-06	2.40E-04	1.32E-01
Air	2,2-dimethyl butane	4.83E-07	5.85E-05	3.22E-02
Air	2,3-dimethyl butane	1.11E-06	1.35E-04	7.40E-02
Air	2-butoxy ethanol	8.70E-07	1.05E-04	5.80E-02
Air	2-ethoxy ethanol	7.75E-07	9.39E-05	5.17E-02
Air	2-hexanone	1.11E-06	1.35E-04	7.40E-02
Air	2-methoxy ethanol	6.01E-07	7.28E-05	4.01E-02
Air	2-methyl 1-butanol	7.91E-07	9.59E-05	5.27E-02
Air	2-methyl 1-butene	1.58E-06	1.92E-04	1.05E-01
Air	2-methyl 2-butanol	2.85E-07	3.45E-05	1.90E-02
Air	2-methyl 2-butene	1.66E-06	2.01E-04	1.11E-01
Air	2-methyl hexane	7.91E-07	9.59E-05	5.27E-02
Air	2-methyl pentane	8.70E-07	1.05E-04	5.80E-02
Air	2-pentanone	1.11E-06	1.35E-04	7.40E-02
Air	3,5-diethyl toluene	2.61E-06	3.16E-04	1.74E-01
Air	3,5-dimethyl ethyl benzene	2.61E-06	3.16E-04	1.74E-01
Air	3-hexanone	1.19E-06	1.44E-04	7.93E-02
Air	3-methyl 1-butanol	7.91E-07	9.59E-05	5.27E-02
Air	3-methyl 1-butene	1.35E-06	1.64E-04	9.00E-02
Air	3-methyl 2-butanol	7.36E-07	8.92E-05	4.91E-02
Air	3-methyl hexane	7.28E-07	8.82E-05	4.85E-02
Air	3-methyl pentane	9.50E-07	1.15E-04	6.33E-02
Air	3-pentanol	8.70E-07	1.05E-04	5.80E-02
Air	3-pentanone	7.91E-07	9.59E-05	5.27E-02
Air	acetaldehyde	1.27E-06	1.54E-04	8.47E-02
Air	acetic acid	1.98E-07	2.40E-05	1.32E-02
Air	acetone	1.90E-07	2.30E-05	1.27E-02
Air	propionaldehyde	1.58E-06	1.92E-04	1.05E-01
Air	alcohols	7.10E-07	8.61E-05	4.73E-02
Air	aldehydes	1.30E-06	1.58E-04	8.67E-02
Air	alkanes	6.90E-07	8.36E-05	4.60E-02
Air	alkenes	2.00E-06	2.42E-04	1.33E-01
Air	benzene	4.35E-07	5.27E-05	2.90E-02
Air	butane	7.04E-07	8.53E-05	4.69E-02
Air	butanol	1.27E-06	1.54E-04	8.47E-02
Air	butene	2.29E-06	2.78E-04	1.53E-01
Air	cis 1,2-dichloroethene	8.70E-07	1.05E-04	5.80E-02
Air	cis 2-butene	2.29E-06	2.78E-04	1.53E-01
Air	cis 2-hexene	2.14E-06	2.59E-04	1.43E-01
Air	cis 2-pentene	2.22E-06	2.69E-04	1.48E-01
Air	CxHy aromatic	1.90E-06	2.30E-04	1.27E-01
Air	CxHy chloro	3.20E-07	3.88E-05	2.13E-02
Air	CxHy halogenated	3.20E-07	3.88E-05	2.13E-02
Air	cyclohexane	5.78E-07	7.01E-05	3.85E-02
Air	cyclohexanol	8.70E-07	1.05E-04	5.80E-02

Air	cyclohexanone	6.01E-07	7.28E-05	4.01E-02
Air	decane	7.68E-07	9.31E-05	5.12E-02
Air	di-i-propyl ether	9.50E-07	1.15E-04	6.33E-02
Air	diacetone alcohol	5.22E-07	6.33E-05	3.48E-02
Air	dichloromethane	1.35E-07	1.64E-05	9.00E-03
Air	diethyl ether	9.50E-07	1.15E-04	6.33E-02
Air	dimethyl ether	3.48E-07	4.22E-05	2.32E-02
Air	dodecane	7.12E-07	8.63E-05	4.75E-02
Air	esters	3.40E-07	4.12E-05	2.27E-02
Air	ethane	2.45E-07	2.97E-05	1.63E-02
Air	ethane diol	7.68E-07	9.31E-05	5.12E-02
Air	ethanol	7.75E-07	9.39E-05	5.17E-02
Air	ethene	1.98E-06	2.40E-04	1.32E-01
Air	ethers	6.80E-07	8.24E-05	4.53E-02
Air	ethyl t-butyl ether	4.27E-07	5.18E-05	2.85E-02
Air	ethylacetate	4.27E-07	5.18E-05	2.85E-02
Air	ethylbenzene	1.42E-06	1.72E-04	9.47E-02
Air	acetylene	1.74E-07	2.11E-05	1.16E-02
Air	formaldehyde	1.03E-06	1.25E-04	6.87E-02
Air	formic acid	6.41E-08	7.77E-06	4.27E-03
Air	heptane	1.03E-06	1.25E-04	6.87E-02
Air	hexane	9.50E-07	1.15E-04	6.33E-02
Air	i-butane	6.17E-07	7.48E-05	4.11E-02
Air	i-butanol	7.52E-07	9.12E-05	5.01E-02
Air	i-butyraldehyde	1.03E-06	1.25E-04	6.87E-02
Air	i-pentane	7.91E-07	9.59E-05	5.27E-02
Air	i-propanol	2.77E-07	3.36E-05	1.85E-02
Air	i-propyl acetate	4.27E-07	5.18E-05	2.85E-02
Air	i-propyl benzene	1.03E-06	1.25E-04	6.87E-02
Air	isoprene	2.22E-06	2.69E-04	1.48E-01
Air	ketones	8.10E-07	9.82E-05	5.40E-02
Air	m-ethyl toluene	2.06E-06	2.50E-04	1.37E-01
Air	m-xylene	2.22E-06	2.69E-04	1.48E-01
Air	methane	1.19E-08	1.44E-06	7.93E-04
Air	methanol	2.61E-07	3.16E-05	1.74E-02
Air	methyl acetate	9.50E-08	1.15E-05	6.33E-03
Air	methyl chloride	1.03E-08	1.25E-06	6.87E-04
Air	methyl ethyl ketone	7.52E-07	9.12E-05	5.01E-02
Air	methyl formate	6.65E-08	8.06E-06	4.43E-03
Air	methyl i-butyl ketone	9.50E-07	1.15E-04	6.33E-02
Air	methyl i-propyl ketone	7.28E-07	8.82E-05	4.85E-02
Air	methyl propene	1.27E-06	1.54E-04	8.47E-02
Air	methyl t-butyl ether	3.09E-07	3.75E-05	2.06E-02
Air	methyl t-butyl ketone	6.49E-07	7.87E-05	4.33E-02
Air	n-butanol	1.27E-06	1.54E-04	8.47E-02
Air	n-butyl acetate	4.83E-07	5.85E-05	3.22E-02
Air	n-butyraldehyde	1.58E-06	1.92E-04	1.05E-01
Air	n-propanol	1.11E-06	1.35E-04	7.40E-02
Air	n-propyl acetate	5.78E-07	7.01E-05	3.85E-02
Air	n-propyl benzene	1.27E-06	1.54E-04	8.47E-02
Air	neopentane	3.48E-07	4.22E-05	2.32E-02
Air	NMVOC	1.19E-06	1.44E-04	7.93E-02
Air	nonane	7.91E-07	9.59E-05	5.27E-02
Air	o-ethyl toluene	1.82E-06	2.21E-04	1.21E-01
Air	o-xylene	2.14E-06	2.59E-04	1.43E-01
Air	octane	8.70E-07	1.05E-04	5.80E-02
Air	p-ethyl toluene	1.82E-06	2.21E-04	1.21E-01
Air	p-xylene	2.06E-06	2.50E-04	1.37E-01
Air	pentanal	1.50E-06	1.82E-04	1.00E-01
Air	pentane	7.91E-07	9.59E-05	5.27E-02
Air	propane	3.56E-07	4.32E-05	2.37E-02
Air	propane diol	9.50E-07	1.15E-04	6.33E-02
Air	propene	2.22E-06	2.69E-04	1.48E-01
Air	propanoic acide	3.01E-07	3.65E-05	2.01E-02
Air	s-butanol	7.91E-07	9.59E-05	5.27E-02
Air	s-butyl acetate	5.38E-07	6.52E-05	3.59E-02
Air	t-butanol	2.45E-07	2.97E-05	1.63E-02
Air	t-butyl acetate	1.27E-07	1.54E-05	8.47E-03
Air	perchloroethylene	5.78E-08	7.01E-06	3.85E-03
Air	toluene	1.27E-06	1.54E-04	8.47E-02
Air	trans 1,2-dichloroethene	7.83E-07	9.49E-05	5.22E-02
Air	trans 2-butene	2.29E-06	2.78E-04	1.53E-01
Air	trans 2-hexene	2.14E-06	2.59E-04	1.43E-01
Air	trans 2-pentene	2.22E-06	2.69E-04	1.48E-01
Air	trichloroethylene	6.49E-07	7.87E-05	4.33E-02
Air	chloroform	4.59E-08	5.56E-06	3.06E-03
Air	undecane	7.68E-07	9.31E-05	5.12E-02
Air	VOC	6.00E-07	7.27E-05	4.00E-02
Air	xylene	2.06E-06	2.50E-04	1.37E-01

3.1.3 Respiratory effects on humans caused by inorganic substances (I,I)

This impact category replaces more or less the winter smog category. Fate analysis is based on empirical data.
All damage factors are expressed per kg emission. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	5.10E-05	6.18E-03	3.40E+00
Air	dust (PM10)	2.74E-04	3.32E-02	1.83E+01
Air	dust (PM2.5)	5.10E-04	6.18E-02	3.40E+01
Air	TSP	8.03E-05	9.73E-03	5.35E+00
Air	NO	1.83E-06	2.22E-04	1.22E-01
Air	NO2	1.19E-06	1.44E-04	7.93E-02
Air	NOx	1.19E-06	1.44E-04	7.93E-02
Air	NOx (as NO2)	1.19E-06	1.44E-04	7.93E-02
Air	SO2	3.90E-05	4.73E-03	2.60E+00
Air	SO3	3.12E-05	3.78E-03	2.08E+00
Air	SOx	3.90E-05	4.73E-03	2.60E+00
Air	SOx (as SO2)	3.90E-05	4.73E-03	2.60E+00

3.1.4 Damages to human health caused by climate change (I,I)

Damage calculation was performed over a time scale of 200 years.
The IPCC equivalence factors have been modified. As damage is not linear dependent on the atmospheric lifetime, a separate damage calculation is made for CO2, CH4 en CH4:

- Gasses with an atmospheric lifetime below 20 years are assumed to behave like methane
- Gasses with an atmospheric lifetime between 20 and 100 years behave like CO2
- Gasses with an atmospheric lifetime oh more than 100 years are assumed to behave like N2O

This means that the IPCC equivalency factor table is split in three groups.
All damage factors are expressed per kg substance. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	methyl chloroform	-4.2E-05	-5.09E-03	-2.80E+00
Air	perfluorethane	2.00E-03	2.42E-01	1.33E+02
Air	trifluoroiodomethane	2.10E-07	2.55E-05	1.40E-02
Air	perfluormethane	1.40E-03	1.70E-01	9.33E+01
Air	CFC-11	2.20E-04	2.67E-02	1.47E+01
Air	CFC-113	6.20E-04	7.52E-02	4.13E+01
Air	CFC-12	1.30E-03	1.58E-01	8.67E+01
Air	carbon dioxide	2.00E-07	2.42E-05	1.33E-02
Air	methylene chloride	1.90E-06	2.30E-04	1.27E-01
Air	HALON-1301	-7.00E-03	-8.48E-01	-4.67E+02
Air	HCFC-123	6.60E-06	8.00E-04	4.40E-01
Air	HCFC-124	8.50E-05	1.03E-02	5.67E+00
Air	HCFC-141b	5.20E-05	6.30E-03	3.47E+00
Air	HCFC-142b	3.40E-04	4.12E-02	2.27E+01
Air	HCFC-22	2.80E-04	3.39E-02	1.87E+01
Air	HFC-125	5.70E-04	6.91E-02	3.80E+01
Air	HFC-134	2.10E-04	2.55E-02	1.40E+01
Air	HFC-134a	7.70E-04	9.33E-02	5.13E+01
Air	HFC-143	6.20E-05	7.52E-03	4.13E+00

Air	HFC-143a	1.00E-02	1.21E+00	6.67E+02
Air	HFC-152a	2.90E-05	3.52E-03	1.93E+00
Air	HFC-227ea	5.90E-04	7.15E-02	3.93E+01
Air	HFC-23	2.50E-03	3.03E-01	1.67E+02
Air	HFC-236fa	1.40E-03	1.70E-01	9.33E+01
Air	HFC-245ca	1.20E-04	1.45E-02	8.00E+00
Air	HFC-32	1.30E-04	1.58E-02	8.67E+00
Air	HFC-41	3.10E-05	3.76E-03	2.07E+00
Air	HFC-4310mee	2.70E-04	3.27E-02	1.80E+01
Air	methane	4.40E-06	5.33E-04	2.93E-01
Air	nitrous oxide	6.70E-05	8.12E-03	4.47E+00
Air	perfluorbutane	1.50E-03	1.82E-01	1.00E+02
Air	perfluorcyclobutane	1.90E-03	2.30E-01	1.27E+02
Air	perfluorhexane	1.60E-03	1.94E-01	1.07E+02
Air	perfluorpentane	1.60E-03	1.94E-01	1.07E+02
Air	perfluorpropane	1.50E-03	1.82E-01	1.00E+02
Air	sulphur hexafluoride	5.20E-03	6.30E-01	3.47E+02
Air	carbontetrachloride	-2.50E-04	-3.03E-02	-1.67E+01
Air	chloroform	8.30E-07	1.01E-04	5.53E-02

3.1.5 Human health effects caused by ionising radiation (I,I)

Fate-, and exposure models are based on studies for the French nuclear industry. All damage factors are based on a release of 1 Bequerel (Bq). The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	C-14	1.60E-11	1.94E-09	1.07E-06
Air	Co-58	3.60E-13	4.36E-11	2.40E-08
Air	Co-60	1.40E-11	1.70E-09	9.33E-07
Air	Cs-134	1.00E-11	1.21E-09	6.67E-07
Air	Cs-137	1.10E-11	1.33E-09	7.33E-07
Air	H-3	1.20E-14	1.45E-12	8.00E-10
Air	I-129	2.50E-10	3.03E-08	1.67E-05
Air	I-131	1.30E-13	1.58E-11	8.67E-09
Air	I-133	7.90E-15	9.58E-13	5.27E-10
Air	Kr-85	1.20E-16	1.45E-14	8.00E-12
Air	Pb-210	1.30E-12	1.58E-10	8.67E-08
Air	Po-210	1.30E-12	1.58E-10	8.67E-08
Air	Pu alpha	7.00E-11	8.48E-09	4.67E-06
Air	Pu-238	5.70E-11	6.91E-09	3.80E-06
Air	Ra-226	7.60E-13	9.21E-11	5.07E-08
Air	Rn-222	2.00E-14	2.42E-12	1.33E-09
Air	Th-230	3.80E-11	4.61E-09	2.53E-06
Air	U-234	8.20E-11	9.94E-09	5.47E-06
Air	U-235	1.70E-11	2.06E-09	1.13E-06
Air	U-238	6.90E-12	8.36E-10	4.60E-07
Air	Xe-133	1.20E-16	1.45E-14	8.00E-12
Water	Ag-110m	4.20E-13	5.09E-11	2.80E-08
Water	Co-58	3.40E-14	4.12E-12	2.27E-09
Water	Co-60	3.70E-11	4.48E-09	2.47E-06
Water	Cs-134	1.20E-10	1.45E-08	8.00E-06
Water	Cs-137	1.40E-10	1.70E-08	9.33E-06
Water	H-3	3.80E-16	4.61E-14	2.53E-11
Water	I-131	4.20E-13	5.09E-11	2.80E-08
Water	Mn-54	2.60E-13	3.15E-11	1.73E-08
Water	Ra-226	1.10E-13	1.33E-11	7.33E-09
Water	Sb-124	6.90E-13	8.36E-11	4.60E-08
Water	U-234	2.00E-12	2.42E-10	1.33E-07
Water	U-235	2.00E-12	2.42E-10	1.33E-07
Water	U-238	1.90E-12	2.30E-10	1.27E-07

3.1.6 Human health effects caused by ozone layer depletion (I,I)

All damage factors are expressed per kg release. The unit of damage is DALYs.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,1,1-trichloroethane	1.02E-04	1.24E-02	6.80E+00
Air	CFC-11	8.50E-04	1.03E-01	5.67E+01
Air	CFC-113	7.65E-04	9.27E-02	5.10E+01
Air	CFC-114	7.23E-04	8.76E-02	4.82E+01
Air	CFC-115	3.40E-04	4.12E-02	2.27E+01
Air	CFC-12	6.97E-04	8.45E-02	4.65E+01
Air	HALON-1201	1.19E-03	1.44E-01	7.93E+01
Air	HALON-1202	1.06E-03	1.28E-01	7.07E+01
Air	HALON-1211	4.34E-03	5.26E-01	2.89E+02
Air	HALON-1301	1.02E-02	1.24E+00	6.80E+02
Air	HALON-2311	1.19E-04	1.44E-02	7.93E+00
Air	HALON-2401	2.13E-04	2.58E-02	1.42E+01
Air	HALON-2402	5.95E-03	7.21E-01	3.97E+02
Air	HCFC-123	1.19E-05	1.44E-03	7.93E-01
Air	HCFC-124	2.55E-05	3.09E-03	1.70E+00
Air	HCFC-141b	8.50E-05	1.03E-02	5.67E+00
Air	HCFC-142b	4.25E-05	5.15E-03	2.83E+00
Air	HCFC-22	3.40E-05	4.12E-03	2.27E+00
Air	HCFC-225ca	1.70E-05	2.06E-03	1.13E+00
Air	HCFC-225cb	1.70E-05	2.06E-03	1.13E+00
Air	methyl bromide	5.44E-04	6.59E-02	3.63E+01
Air	methyl chloride	1.70E-05	2.06E-03	1.13E+00
Air	carbontetrachloride	1.02E-03	1.24E-01	6.80E+01

3.2 Damage category Ecosystem Quality (I,I)

The Ecosystem Quality damages are specified as PDF*m^2*yr. PDF is short for Potentially Disappeared Fraction of Species. A damage of one means all species disappear from one m^2 during one year, or 10% of all species disappear from 10 m^2 during one year, or 10% of all species disappear from 1 m^2 during 10 years. Within the damage category Ecosystem Quality special care is needed to avoid double counting when land-use is modelled; see the remarks under these damage categories.

3.2.1 Damage to Ecosystem Quality caused by ecotoxic emissions (I,I)

Fate analysis was done in EUSES.
Pesticides that evaporate during application must be counted as air emissions. Pesticides that are accidentally sprayed in surface waters must be counted as water emissions. The remainder must be counted as soil emissions. The damage from pesticides in the agricultural soil as such (root zone) was deliberately excluded to avoid double counting with land-use. This means the damage factors in this list are based on secondary (leaching) emissions from the soil into surface and ground water and evaporation.

All damage factors are expressed per kg release. The unit of damage is PDF*m^2*yr.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	1,2,3-trichlorobenzene	3.51E-02	7.78E-06	1.95E-03
Air	1,2,4-trichlorobenzene	2.54E-02	5.63E-06	1.41E-03
Air	1,3,5-trichlorobenzene	1.29E-01	2.86E-05	7.15E-03
Air	2,4-D	1.46E+00	3.24E-04	8.09E-02
Air	Arsenic	5.00E+01	1.11E-02	2.77E+00
Air	Atrazine	2.09E+02	4.63E-02	1.16E+01
Air	Azinphos-methyl	1.10E+04	2.44E+00	6.10E+02
Air	Bentazon	7.33E+00	1.63E-03	4.06E-01
Air	benzene	2.75E-03	6.10E-07	1.52E-04
Air	benzo(a)pyrene	1.42E+02	3.15E-02	7.87E+00
Air	Carbendazim	2.40E+03	5.32E-01	1.33E+02
Air	Cadmium	2.66E+03	5.90E-01	1.47E+02
Air	Chromium	9.70E+01	2.15E-02	5.38E+00
Air	Copper	1.93E+02	4.28E-02	1.07E+01

Air	di(2-ethylhexyl)phthalate	1.94E-03	4.30E-07	1.08E-04
Air	dibutylphthalate	1.13E-01	2.51E-05	6.26E-03
Air	Dichlorvos	1.61E+00	3.57E-04	8.92E-02
Air	2,3,7,8-TCDD Dioxin	1.32E+05	2.93E+01	7.32E+03
Air	Diquat-dibromide	2.39E+03	5.30E-01	1.32E+02
Air	Diuron	4.43E+03	9.82E-01	2.46E+02
Air	DNOC	8.19E+00	1.82E-03	4.54E-01
Air	Fentin acetate	6.77E+02	1.50E-01	3.75E+01
Air	fluoranthene	4.37E-02	9.69E-06	2.42E-03
Air	gamma-HCH (Lindane)	2.16E+00	4.79E-04	1.20E-01
Air	Hexachlorobenzene	3.88E+01	8.60E-03	2.15E+00
Air	Mercury	4.53E+01	1.00E-02	2.51E+00
Air	Malathion	1.17E+02	2.59E-02	6.49E+00
Air	Maneb	3.84E+01	8.51E-03	2.13E+00
Air	Mecoprop	7.79E-02	1.73E-05	4.32E-03
Air	Metabenzthiazuron	3.07E+02	6.81E-02	1.70E+01
Air	metals	3.40E+01	7.54E-03	1.88E+00
Air	Metamitron	3.78E+01	8.38E-03	2.10E+00
Air	Metribuzin	4.92E+02	1.09E-01	2.73E+01
Air	Mevinphos	2.13E+03	4.72E-01	1.18E+02
Air	Monolinuron	1.06E+02	2.35E-02	5.88E+00
Air	Nickel	9.06E+02	2.01E-01	5.02E+01
Air	PAH's	7.80E-04	1.73E-07	4.32E-05
Air	Parathion	6.05E+01	1.34E-02	3.35E+00
Air	Lead	3.94E+01	8.74E-03	2.18E+00
Air	Polychlorobiphenyls	8.07E+01	1.79E-02	4.47E+00
Air	pentachlorophenol	1.33E+01	2.95E-03	7.37E-01
Air	Simazine	1.44E+03	3.19E-01	7.98E+01
Air	Thiram	2.26E+02	5.01E-02	1.25E+01
Air	toluene	2.40E-04	5.32E-08	1.33E-05
Air	Trifluralin	1.09E+00	2.42E-04	6.04E-02
Air	Zinc	6.61E+02	1.47E-01	3.66E+01
Water	1,2,3-trichlorobenzene	1.56E-01	3.46E-05	8.65E-03
Water	1,2,4-trichlorobenzene	1.39E-01	3.08E-05	7.71E-03
Water	1,3,5-trichlorobenzene	2.73E-01	6.05E-05	1.51E-02
Water	2,4-D	7.56E-02	1.68E-05	4.19E-03
Water	Arsenic	9.14E+00	2.03E-03	5.07E-01
Water	Atrazine	5.06E+01	1.12E-02	2.80E+00
Water	Azinphos-methyl	8.87E+02	1.97E-01	4.92E+01
Water	Bentazon	5.81E-02	1.29E-05	3.22E-03
Water	benzene	4.80E-02	1.06E-05	2.66E-03
Water	benzo(a)pyrene	3.68E+01	8.16E-03	2.04E+00
Water	Carbendazim	1.63E+02	3.61E-02	9.04E+00
Water	Cadmium	3.87E+02	8.58E-02	2.15E+01
Water	Chromium	5.54E+01	1.23E-02	3.07E+00
Water	Copper	1.18E+02	2.62E-02	6.54E+00
Water	di(2-ethylhexyl)phthalate	6.37E-01	1.41E-04	3.53E-02
Water	dibutylphthalate	1.62E+00	3.59E-04	8.98E-02
Water	Dichlorvos	1.81E-01	4.01E-05	1.00E-02
Water	dioxins (TEQ)	1.87E+05	4.15E+01	1.04E+04
Water	Diquat-dibromide	1.18E+02	2.62E-02	6.54E+00
Water	Diuron	2.31E+02	5.12E-02	1.28E+01
Water	DNOC	6.73E-01	1.49E-04	3.73E-02
Water	Fentin acetate	7.85E+02	1.74E-01	4.35E+01
Water	fluoranthene	3.96E+00	8.78E-04	2.20E-01
Water	gamma-HCH (Lindane)	1.04E+01	2.31E-03	5.76E-01
Water	hexachlorobenzene	4.55E+01	1.01E-02	2.52E+00
Water	Mercury	1.93E+01	4.28E-03	1.07E+00
Water	Malathion	1.64E+02	3.64E-02	9.09E+00
Water	Maneb	6.23E-01	1.38E-04	3.45E-02
Water	Mecoprop	1.35E-02	2.99E-06	7.48E-04
Water	Metabenzthiazuron	1.43E+01	3.17E-03	7.93E-01
Water	Metamitron	3.77E-01	8.36E-05	2.09E-02
Water	Metribuzin	3.18E+00	7.05E-04	1.76E-01
Water	Mevinphos	6.73E+01	1.49E-02	3.73E+00
Water	Monolinuron	1.04E+01	2.31E-03	5.76E-01
Water	Nickel	1.16E+02	2.57E-02	6.43E+00
Water	PAH's	2.10E-03	4.66E-07	1.16E-04
Water	Parathion	2.48E+02	5.50E-02	1.37E+01
Water	Lead	5.95E+00	1.32E-03	3.30E-01
Water	Polychlorobiphenyls	2.58E+02	5.72E-02	1.43E+01
Water	pentachlorophenol	2.51E+01	5.57E-03	1.39E+00
Water	Simazine	6.03E+01	1.34E-02	3.34E+00
Water	Thiram	8.74E+02	1.94E-01	4.84E+01
Water	toluene	1.73E-01	3.84E-05	9.59E-03
Water	Trifluralin	7.80E+01	1.73E-02	4.32E+00
Water	Zinc	1.31E+01	2.90E-03	7.26E-01
Soil	1,2,3-trichlorobenzene (ind.)	2.41E+00	5.34E-04	1.34E-01
Soil	1,2,4-trichlorobenzene (ind.)	2.26E+00	5.01E-04	1.25E-01
Soil	1,3,5-trichlorobenzene (ind.)	1.19E+00	2.64E-04	6.60E-02
Soil	2,4-D (agr.)	1.27E-04	2.82E-08	7.04E-06
Soil	Arsenic (ind.)	6.43E+01	1.43E-02	3.56E+00
Soil	Atrazine (agr.)	1.49E-01	3.30E-05	8.26E-03
Soil	Azinphos-methyl (agr.)	3.55E-01	7.87E-05	1.97E-02
Soil	Bentazon (agr.)	1.66E-02	3.68E-06	9.20E-04
Soil	benzene (ind.)	4.97E-01	1.10E-04	2.75E-02
Soil	benzo(a)pyrene (ind.)	7.25E+03	1.61E+00	4.02E+02
Soil	Carbendazim (agr.)	2.34E+00	5.19E-04	1.30E-01
Soil	Cadmium (agr.)	2.35E+00	5.21E-04	1.30E-01
Soil	Cadmium (ind.)	3.35E+03	7.43E-01	1.86E+02
Soil	Chromium (ind.)	1.24E+02	2.75E-02	6.87E+00
Soil	Copper (ind.)	2.44E+02	5.41E-02	1.35E+01
Soil	di(2-ethylhexyl)phthalate(ind)	2.67E-02	5.92E-06	1.48E-03
Soil	dibutylphthalate (ind.)	1.14E+00	2.53E-04	6.32E-02
Soil	Dichlorvos (agr.)	7.52E-04	1.67E-07	4.17E-05
Soil	2,3,7,8-TCDD Dioxin (ind.)	2.09E+05	4.63E+01	1.16E+04
Soil	Diquat-dibromide (agr.)	6.84E-02	1.52E-05	3.79E-03
Soil	Diuron (agr.)	4.07E-02	9.02E-06	2.26E-03
Soil	DNOC (agr.)	6.17E-03	1.37E-06	3.42E-04
Soil	Fentin acetate (agr.)	3.84E-01	8.51E-05	2.13E-02
Soil	fluoranthene (ind.)	8.00E+00	1.77E-03	4.43E-01
Soil	gamma-HCH (Lindane) (agr.)	1.38E+00	3.06E-04	7.65E-02
Soil	hexachlorobenzene (ind.)	9.96E+01	2.21E-02	5.52E+00
Soil	Mercury (ind.)	1.03E+02	2.28E-02	5.71E+00
Soil	Malathion (agr.)	2.79E-02	6.19E-06	1.55E-03
Soil	Maneb (agr.)	2.61E-01	5.79E-05	1.45E-02
Soil	Mecoprop (agr.)	2.79E-06	6.19E-10	1.55E-07
Soil	Metabenzthiazuron (agr.)	3.15E-01	6.98E-05	1.75E-02
Soil	Metamitron (agr.)	2.03E-04	4.50E-08	1.13E-05
Soil	Metribuzin (agr.)	4.91E-02	1.09E-05	2.72E-03
Soil	Mevinphos (agr.)	2.09E-01	4.63E-05	1.16E-02
Soil	Monolinuron (agr.)	4.38E-01	9.71E-05	2.43E-02
Soil	Nickel (ind.)	1.16E+03	2.57E-01	6.43E+01
Soil	Parathion (agr.)	3.24E-02	7.18E-06	1.80E-03
Soil	Lead (ind.)	6.83E-01	1.51E-04	3.79E-02
Soil	PCBs (ind.)	8.35E+02	1.85E-01	4.63E+01
Soil	pentachloorfenol (ind.)	2.51E+01	5.57E-03	1.39E+00
Soil	Simazine (agr.)	3.87E-01	8.58E-05	2.15E-02
Soil	Thiram (agr.)	9.96E-01	2.21E-04	5.52E-02
Soil	toluene (ind.)	6.79E-02	1.51E-05	3.76E-03
Soil	Trifluralin (agr.)	2.07E-02	4.59E-06	1.15E-03
Soil	Zinc (ind.)	8.39E+02	1.86E-01	4.65E+01

3.2.2 Damage to Ecosystem Quality caused by the combined effect of acidification and eutrophication (I,I)

Unfortunately no damage factors for emissions to water and soil could yet be calculated. We suggest to use the damage factors for air as a temporary, but crude solution.
The damage caused by fertilisers that are deliberately applied on agricultural soil is already included in the land-use damage factors, and should not be treated as an emission leading to eutrophication. The fertilisers that evaporate, or that are accidentally sprayed in surface waters should be counted as an emission.

All damage factors are based on kg emissions to air. The unit of damage is PDF*m^2*yr.

Com-part-ment	Substances	Damage factor	Normalised damage factor	Weighted damage factor
Air	ammonia	15.57	3.45E-03	8.63E-01
Air	NO	8.789	1.95E-03	4.87E-01
Air	NO2	5.713	1.27E-03	3.17E-01
Air	NOx	5.713	1.27E-03	3.17E-01

Air	NOx (as NO2)	5.713	1.27E-03	3.17E-01
Air	SO2	1.041	2.31E-04	5.77E-02
Air	SO3	0.8323	1.85E-04	4.61E-02
Air	SOx	1.041	2.31E-04	5.77E-02
Air	SOx (as SO2)	1.041	2.31E-04	5.77E-02

3.2.3 Damage to ecosystem quality caused by land occupation and land conversion (I,I)

It is important to separate two cases:

- Land occupation
- Land conversion

The damage factors for occupation are per area [m^2] times the duration of the occupation [yr]. The effect of restoration of the area type to it's natural condition is not included here, but in the land conversion damage factors. Occupation is seen as a damage, because the area is prevented from restoring to its natural area. Typical examples of land occupation are:
Building new houses in an existing urban area, using a factory in an industrial area, agricultural production in an existing agricultural area. In most cases land is used that has already been converted long ago. In such cases conversion should not be considered.

The damage factors for conversions are per area [m^2]. Conversion factors should only be used if it is clear that a process results in the conversion of one area type into another. Examples are: mining operations in natural areas, expanding agricultural areas at the expense of natural areas, and dumping waste. The difference with the factors for occupation is the inclusion of a restoration time that has been set to 30 years as default. Other restoration times can be easily calculated by dividing the damage factors by 30 and multiplying them with the intended restoration time.

Conversions between agricultural and urban area types can also be modelled by subtracting the damage factors, but, as the damage factors can have considerable uncertainties, the result is unreliable. We suggest to use conversion data only for cases where natural areas are converted into non-natural area types.

3.2.3.1 COMPATIBILITY WITH ESU DATABASE

The ESU database, produced at the ETH Zurich, is one of the few large databases that has consistently included land-use data. Unfortunately no distinction is made between conversion and occupation, the two are always combined. This means a restoration time is always included, and this restoration time cannot be separated in a elegant way. In order to be able to use this large database damage factors: land-use II-III, land use II-IV, land-use III-IV and land-use IV-IV have been estimated using the following (rather crude) assumptions:

- *ESU land-use type II can be interpreted as near to natural area*
- *ESU land-use type III can be interpreted as green urban or rail areas. These are the not very intensively used areas*
- *ESU land-use type IV can be interpreted as continuos urban land*
- *ESU assumes a 5 year restoration time between type IV and III. In many cases an occupation time for industrial activities of 25 or 30 years is used. As a result the restoration time results in an overestimation of 20% for land-use II-IV. In the figure presented here the damage factor is thus lowered by 20%.*
- *After the conversion from Land-use II-IV the ESU database uses the factors II-III for the restoration time between type II and III. As we do not want to include these, in general they should be omitted. Unfortunately for processes like the production of hydropower this class is used in a different way and should thus be included*

Using the ESU database is thus not very straightforward, but with the factors presented here a reasonable first order approximation can be obtained, except for instance for processes that involve agricultural production and hydropower.

3.2.3.2 DOUBLE COUNTING

The damage factors are based on empirical observations of the number of plant species per area type. In such observations all effects of the area type are included. This means that also the effects of emissions are included. To avoid double counting these emissions, please observe the guidelines for pesticides and eutrophication.

The unit of damage of land occupation is PDF*m^2*yr.

Land-occupation	Damage factor	Normalised damage factor	Weighted damage factor
land use II-III	0.51	1.13E-04	2.83E-02
land use II-IV	0.96	2.13E-04	5.32E-02
land use III-IV	0.96	2.13E-04	5.32E-02
land use IV-IV	1.15	2.55E-04	6.37E-02
Occup. as Contin. urban land	1.15	2.55E-04	6.37E-02
Occup. as Convent. arable land	1.15	2.55E-04	6.37E-02
Occup. as Discont. urban land	0.96	2.13E-04	5.32E-02
Occup. as Forest land	0.11	2.44E-05	6.10E-03
Occup. as Green urban land	0.84	1.86E-04	4.66E-02
Occup. as Industrial area	0.84	1.86E-04	4.66E-02
Occup. as Intens. meadow land	1.13	2.51E-04	6.26E-02
Occup. as Organic arable land	1.09	2.42E-04	6.04E-02
Occup. as organic meadow land	1.02	2.26E-04	5.65E-02
Occup. as rail/ road area	0.84	1.86E-04	4.66E-02
Occup. as Integrated arable land	1.15	2.55E-04	6.37E-02
Occup. as less intens.meadow land	1.02	2.26E-04	5.65E-02

The unit of damage of land conversion is PDF*m^2.

Land conversion	Damage factor	Normalised damage factor	Weighted damage factor
Conv. to Continuous urban land	34.53	7.66E-03	1.91E+00
Conv. to Convent. arable land	34.38	7.62E-03	1.91E+00
Conv. to Discontinuous urban	28.73	6.37E-03	1.59E+00
Conv. to Green urban	25.16	5.58E-03	1.39E+00
Conv. to Industrial area	25.16	5.58E-03	1.39E+00
Conv. to Integr. arable land	34.38	7.62E-03	1.91E+00
Conv. to Intensive meadow	34.02	7.54E-03	1.89E+00
Conv. to Less intensive meadow	30.62	6.79E-03	1.70E+00
Conv. to Organic arable land	32.73	7.26E-03	1.81E+00
Conv. to Organic meadow	30.62	6.79E-03	1.70E+00
Conv. to rail/ road area	25.16	5.58E-03	1.39E+00

3.3 Damage category Resources (I,I)

The damages to resources are specified as MJ surplus energy. A damage of 1 means that due to a certain extraction further extraction of this resources in the future will require one additional MJ of energy, due to the lower resource concentration, or other unfavourable characteristics of the remaining reserves. The point in future has been chosen as the time at which 5 times the cumulative extraction of the resource before 1990 is extracted. The factor 5 is chosen arbitrarily, but after normalisation this has no further significance.

3.3.1 Damage to Resources caused by extraction of minerals (I,I)

The damage factors are per kg of extracted metal or ore:

- "in ore" refers to the metal content in the ore, so 1kg iron (in ore) means one kg of pure iron
- "ore" refers to the ore. An average metal content is assumed to calculate these figures.

The unit of damage is MJ surplus energy per kg of extracted material

Minerals	Damage factor	Normalised damage factor	Weighted damage factor
aluminium (in ore)	2.38	1.59E-02	3.17E+00
bauxite	0.5	3.33E-03	6.67E-01
chromium (in ore)	0.9165	6.11E-03	1.22E+00
chromium (ore)	0.275	1.83E-03	3.67E-01
copper (in ore)	36.7	2.45E-01	4.89E+01
copper (ore)	0.415	2.77E-03	5.53E-01
iron (in ore)	0.051	3.40E-04	6.80E-02
iron (ore)	0.029	1.93E-04	3.87E-02
lead (in ore)	7.35	4.90E-02	9.80E+00
lead (ore)	0.368	2.45E-03	4.91E-01
manganese (in ore)	0.313	2.09E-03	4.17E-01
manganese (ore)	0.141	9.40E-04	1.88E-01
mercury (in ore)	165.5	1.10E+00	2.21E+02
molybdene (in ore)	41	2.73E-01	5.47E+01
molybdenum (ore)	0.041	2.73E-04	5.47E-02
nickel (in ore)	23.75	1.60E-01	3.21E+01
nickel (ore)	0.356	2.41E-03	4.81E-01
tin (in ore)	600	4.05E+00	8.11E+02
tin (ore)	0.06	4.05E-04	8.11E-02
tungsten (ore)	0.927	6.26E-03	1.25E+00
zinc (in ore)	4.09	2.76E-02	5.53E+00
zinc (ore)	0.164	1.11E-03	2.22E-01

3.3.2 Damage to Resources caused by extraction of fossil fuels (I,I)

In the individualist perspective, the extraction of fossil resources is not considered to be a problem