ENVIRONMENTAL DAMAGE ESTIMATIONS IN INDUSTRIAL PROCESS CHAINS

Methodology development with a case study on waste incineration and a special focus on human health

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A la Marisa

Die Notwendigkeit zu entscheiden ist stets grösser als das Mass unserer Erkenntnis.
(The necessity to decide is always larger than the degree of cognition.)

Immanuel Kant

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Summary

Environmental damage estimations in industrial process chains need the assessment of environmental impacts in two perspectives: process chain-orientated and site-orientated. For both perspectives specific assessment tools have been developed.

Life Cycle Assessment (LCA) is a fairly new chain-orientated tool to evaluate the environmental performance of products focussing on the entire life cycle of these products: from the extraction of resources via manufacturing and use to the final processing of the disposed product. Through all these stages consumption of resources and releases to air, water and soil are identified and quantified in the Life Cycle Inventory (LCI) analysis. Subsequently, it follows the Life Cycle Impact Assessment (LCIA) phase whose purpose is to assess a product system's life cycle inventory results to better understand their environmental relevance.

Environmental Risk Assessment (ERA) of chemicals is a tool to assess the risk of specific chemicals, which are harmful to man or to the environment under certain circumstances. In comparison to the chain-orientated, site-generic LCIA, the commonly used method for chemicals ERA can be considered to be site-specific, since in general it is applied for the hazard evaluation of a process at one location. The Impact Pathway Analysis (IPA) is a similar method that has been developed for the assessment of environmental damages by the terms of physical impact parameters like for instance cancer cases or restricted activity days, based on the experiences made in the application of ERA. In the IPA usually the physical impact parameters are converted into external costs.

Lacking of common reference for the comparison of different environmental damages involves inevitably a value judgement. One way to convert environmental damage estimates in meaningful results for society is by the means of external costs. By monetisation environmental damages can be introduced in economic balances, allowing a comparison with other types of costs. In this way they could be internalised. However, several monetisation approaches exist and, depending on the socio-economic perspective, individuals may prefer other weighting scheme. It can be said that, in general, weighting schemes approved by international organisation have a high level of acceptance.

In order to overcome the disadvantages of the cited site-generic and site-specific methods, recently site-dependent impact assessment methods have been developed. To be operational the consideration of spatial differentiation in the assessment process should require minimal additional data, i.e. only some general site-parameters. These parameters have to take into account fate, exposure and effect information in the form of indicators that are applicable for classes of emissions sites rather than for specific sites.

Recent developments are leading to advances in the practice of LCIA, but the major limitations and uncertainties cannot be expected to go away in the near future. They only can be made obvious by systematic uncertainty analysis and communicating the results in an adequate manner. It would be an important step further if one encompassing framework for midpoint and endpoints could be developed, including the most important variables of both

types of approaches, thus enabling modelling along the two approaches and comparing the results with each other. This may well imply to integrate other environmental management tools. The level of sophistication might depend upon the type of application and the availability of data. Midpoints are considered to be a point in the cause-effect chain of a particular environmental impact, at which characterisation factors can be calculated to reflect the relative importance of an emission or extraction in a Life Cycle Inventory, prior to the endpoint. Endpoints are the actual impacts or damages that are caused by the environmental loads analysed in the inventory.

In order to allow a high level of site-speciality and accuracy, this thesis differentiates the type of life cycles with respect to their process number and establishes a differentiation between industrial process chains with up to 100 processes and complete product systems with a much higher number of processes. Only little efforts have been made so far to systematically explore the inherent uncertainties, interfaces and possibilities for integration and communication of the mentioned chain-orientated and site-orientated environmental assessment methods in the case of the industrial process chains as defined above. This is what this thesis has in mind. The research question of this thesis is:

- In which way environmental damages of industrial process chains can be estimated the most accurately possible, but still with acceptable efforts and in a communicable way?

That means the objective of this thesis, as outlined in Chapter 1, is to find an adequate trade-off between process chain-orientated and site-orientated environmental impact assessment and to convert environmental damage estimates in meaningful results like environmental costs.

The objective involves the following subsequent questions:

- What are the uncertainties in environmental damage assessment methods?
- What possibilities are there for interfaces and integration of environmental management tools, especially LCA, ERA, IPA and environmental costs?
- How can midpoint and endpoint approaches in LCIA be put in a common framework?

Evidently, all these questions are part of what is called sophistication of environmental management tools and need complex models to be answered.

Chapter 2 familiarises the reader to the basic concepts, tools and technical elements behind environmental damage estimations in industrial process chains. The portrayed methods are critically reviewed. The so-called toolbox for environmental management is explained. Its application-dependency in a process chain perspective is shown, a guide is developed. It is recommended to be careful in the selection of a tool for a determined environmental problem, since the outcome of a study is in somehow predominated by the analytical environmental management tool used.

In general, all analytical environmental impact assessment methods include an effect analysis that indicates the severity of the consequences due to the pollution. Crucial elements are the dose-response and exposure-response functions. Moreover, each analysis can include weighting of the relation between different types of effects. One weighting option consists of costs that are an important aspect in environmental management. A special focus is given on external environmental costs by monetisation. The relation of monetisation to other weighting schemes is demonstrated.

Next the LCA methodology is further described. On the subject of LCIA first the general framework is presented and then several methods, especially those with a single environmental performance index, are discussed. New advances include LCIA damage-orientated or endpoint approaches. Finally, the latest developments in LCIA are characterised as sophistication, which is considered to be the ability to provide very accurate and comprehensive reports that reflect the potential impact of the stressors to help decision-making in each particular case.

In the following section, Environmental Risk Assessment and Impact Pathway Analysis are described. While the conventional ERA method focuses on the evaluation of the probability of hazard occurrence, the more recently developed IPA offers a framework for damage estimations. The application of the ERA methods in this study needs complex technical elements like Gaussian dispersion and the multimedia fate and exposure models that therefore are described here. Moreover, an introduction is given to the integrated impact assessment software model EcoSense, which is used in the IPA.

Finally, the last section of Chapter 2 analyses the character of life cycles with respect to their process number and establishes a differentiation between industrial process chains with less than 100 processes and complete product systems. The relevance of this differentiation for the way of assessing environmental impacts is demonstrated.

Chapter 3 introduces the reader to the case study on waste incineration with a special focus on human health impacts. Based on a common set of fundamental data the results have been obtained for applications of the commonly used methods LCA, ERA and IPA to the Municipal Solid Waste Incinerator (MSWI) of Tarragona and its process chain for two situations: with and without an advanced gas treatment system. Moreover, a modular model is briefly described that allows to generate a scenario of a MSWI with the additional control device to eliminate NOx called DeNOx. As part of the case chapter it is verified that this life cycle can be defined as an industrial process chain with around 20 significant processes, and not as a complete product system, e.g. a computer.

With Chapter 4 starts the methodology development. The existence of different types of uncertainties is often mentioned as crucial limitation for a clear interpretation of environmental damage estimations. Therefore, in Chapter 4 a strategy for stochastic uncertainty assessment by Monte Carlo simulation is presented. A framework for such type of assessment is presented for LCI and IPA. The procedure is applied to the LCI of the case study and the damage estimations of the MSWI emissions by IPA on a local scale. The application of the method to IPA shows that the uncertainty and variability calculated are less than obtained in a study with analytical methods. The sensitivity analysis indicates that using toxicological dose-response and epidemiological exposure-response functions the main impact on human health does not stem from the PCDD/Fs and the heavy metals, but particles and NOx.

This parallel application allows to show that the uncertainties in the inventory analysis are significantly less important than those in the damage assessment; especially the use of dispersion models, dose-response and exposure-response functions as well as the weighting schemes implies huge uncertainties.

Chapter 5 provides a mathematical framework and a flowchart that allows spatial differentiation at different levels of detail and proposes an integration of LCA, IPA and environmental costs. It is especially applicable to so-called industrial process chains, i.e. not full life cycles of product systems. Therefore, the established methodology is called "Environmental Damage Estimations for Industrial Process Chains". An important part of this methodology is its mathematical framework that permits to manage the distribution of the environmental load along different life cycle stages and the assignation of the damages in the corresponding region of emission. The core consists of the flowchart. Different options are given to find the adequate trade-off between site-generic and site-specific environmental impact assessments and to convert environmental damage estimates in meaningful results like environmental costs.

The methodology puts midpoint and endpoint approaches in LCIA in a common framework. In this way the direct comparison of midpoint and endpoint indicators results is made possible. The flowchart is divided into several parts, which are in somehow similar to the LCA phases, but which have often a different content. The first phase is Goal and Scope Definition with the choice by the decision-maker of the weighting and aggregation scheme to use, the second phase is LCI and the third is LCIA. Here the similarity with the LCA framework stops; instead of interpretation the next steps are Dominance Analysis and Spatial Differentiation, Fate & Exposure Assessment and Consequence Analysis as well as Damage Profile and Eco-efficiency. In the proposed methodological framework a number of analytical tools like ERA and technical elements like uncertainty assessment by Monte Carlo simulation are used. Optionally the consideration of accidents is foreseen. Finally, the methodology is applied to the results of the LCA study of the waste incinerator process chain, in order to show its applicability.

Chapter 6 applies and further develops a framework for site-dependent impact assessments as a way to find a compromise between site-specific damage endpoint assessments and potential midpoint life cycle indicators. The main reasons for the discordance between the calculated impact potentials and the expectation of actual impacts in LCIA is, on the one hand, the no consideration of the spatial distribution of the receptors and, on the other hand, the lack of information on the point of emission and the corresponding dispersion conditions in the respective medium. To relate these main factors the presented method uses representative generic impact classes corresponding to several receptor distributions and dispersion conditions, based on statistical reasoning. The existing approach was adapted to include more sophisticated models: a detailed dispersion program and a geographic information system. However, it became clear that further research is required on how to statistically define the source characteristics. The existing basic framework has been generalised to cover more receptors. It is applied in detail for human health effects due to airborne emissions in Catalonia; factors are calculated on a district level.

The site-dependent impact assessment method fits perfectly in the framework developed in Chapter 5 and is applied to compare the importance of transports in relation to the waste incineration process that have been identified to be of special interest by the application in Chapter 5. Spatial differentiation in LCIA has shown to be feasible with a reasonable effort and little additional information required in LCI (district name, stack height, information about particle size distribution). The comparison of the results derived by the endpoint impact indicators with those obtained by the midpoint indicator Human Toxicity Potential indicates that the HTP underestimates the environmental impact of the transport processes. This is especially relevant for the most advanced gas treatment technologies, i.e. additionally with DeNOx, where the improved cleaning capacity is compensated more and more by the additional efforts in raw material supply, including supplementary transport.

Chapter 7 presents the overall discussion of the research done. The presented methodological framework is limited at least with respect to two aspects: the inherent uncertainties and its principal applicability to industrial process chains, here defined as chains with 100 or less processes. The most adequate applications of the presented environmental damage estimation system for industrial process chains are end-of-life management and environmental optimisation of the setting of a new plant. It is evident that a lot of environmental problems related to a life-cycle perspective cannot be studied by this framework, at least currently due to insufficient data availability and infeasible workload. Based on this and other reflections, a strategy for generic applications is proposed. This strategy refers to the development of an internationally accepted framework for midpoint and endpoint LCIA approaches.

In the conclusions, Chapter 8, the methodology development is highlighted as main finding of the thesis. Finally, interesting outlooks are offered. Further work on the differentiation of life cycle methods with regard to the number of process in the considered chain is encouraged. It is stated that the basis for the creation of a new generation of integrated waste management models have been established that include the optimisation of the setting of waste treatment plants and the related transport routes. However, still a lot of work is ahead to make this approach fully operational.

Part A:

Introduction

1 OBJECTIVES AND STRUCTURE OF THE STUDY

1.1 PROBLEM-SETTING

In the last decade, several environmental management concepts and tools have been established. A lot of these methods refer to environmental impact assessment in industrial process chains. Environmental damage estimations in industrial process chains need the assessment of environmental impacts in two perspectives: process chain-orientated and site-orientated. For both perspectives specific assessment tools have been developed.

Life Cycle Assessment (LCA) is a fairly new chain-orientated tool (SETAC, 1993; ISO 14040, 1997) to evaluate the environmental performance of products focussing on the entire life cycle of these products: from the extraction of resources, via manufacturing and use, to the final processing of the disposed product. Through all these stages consumption of resources (including energetic sources) and releases to air, water and soil are identified and quantified in the Life Cycle Inventory (LCI) analysis (Castells, 1995; ISO 14041, 1998). Subsequently, it follows the Life Cycle Impact Assessment (LCIA) phase, whose purpose is to assess a product system's life cycle inventory results to better understand their environmental significance (Udo de Haes, 1996). It models and evaluates selected environmental issues, called impact categories, and through the use of category indicators portrays the potential environmental impact of the environmental loads in an aggregated manner. There have been a number of advances made in the evaluation of environmental impacts in Life Cycle Assessment in recent times (Bare et al., 2000):

- Now the framework for Life Cycle Impact Assessment has become standardised by the International Standard Organisation (ISO 14042, 2000), enhancing the comparability and avoiding unnecessary variation between studies.
- The fate of substances is increasingly taken into account, in particular using multimedia fate models.
- The results of different characterisation procedures for the same category are compared amongst each other and they show convergence.
- Better distinctions are being made between scientific information and value choices.

Environmental Risk Assessment (ERA) of chemicals (Fairman et al., 1998) is an environmental management tool used to assess the risk of specific chemicals, which are harmful to man or to the environment, under certain circumstances of use or in certain environmental compartments. In comparison to the chain-orientated, site-generic LCIA, the commonly used chemical-orientated method ERA is site-orientated. Generally, it is applied for the hazard evaluation of a process at one location. The exposure assessment phase in ERA is the process to estimate the probability that adverse effects to the environment may occur as consequence of the exposure to one or more substances.

The Impact Pathway Analysis (IPA) is a similar method (EC, 1995) that has been developed for the assessment of environmental damages by the terms of physical impact parameters, like for instance cancer cases or restricted activity days, based on the experiences made in ERA. In the IPA usually the physical impact parameters are converted into external costs. Integrated assessment models assist to carry out IPA studies. In site-specific methods the source characteristics and the dispersion conditions influence significantly the final results, especially for pollutants with a short residence time (Rabl et al., 1998).

Lacking of common reference for comparison of different environmental damages involves inevitably a value judgement. One way to convert environmental damage estimates in meaningful results for society is by the means of the above-mentioned external costs. By monetisation environmental damages can be introduced in the expressions of economic balances, allowing a comparison with other types of costs (Rabl & Spadaro, 1998). In this way they could be internalised; however, several monetisation approaches exist and, depending on the socio-economic perspective, individuals may prefer another weighting scheme.

The outcomes of environmental impact assessments are often criticised to be not reliable due to the inherent uncertainties in the models used (Weidema, 2000). Due to this problem, slowly different ways of analysing uncertainty are gaining importance in the normally applied methods, but its use is not common practise (Huijbregts et al., 2000a).

In order to overcome the disadvantages of the cited site-generic and site-specific methods, recently site-dependent impact assessment methods have been developed (Potting, 2000; Nigge, 2000). They can be considered as a bridge between ERA and LCIA. This approach signifies a compromise between exactness and feasibility. To be operational the consideration of spatial differentiation in the assessment process should require minimal additional data, i.e. only some general site-parameters. These parameters have to take into account fate, exposure and effect information in the form of indicators that are applicable for classes of emissions sites rather than for specific sites. This idea of generic classes is the basis for the trade-off between the accuracy of site-specific impact assessment and the practicability of spatial differentiation in impact assessments for a process chain or life cycle perspective.

Important aspects of this technique are the definition of the midpoints and the endpoints and their different approaches. Midpoints are considered to be a point in the cause-effect chain of a particular impact category, at which characterisation factors of potential impacts can be calculated to reflect the relative importance of an emission or extraction in a Life Cycle

Inventory, prior to the endpoint (Bare et al., 2000). Endpoints are the actual impacts or damages that are caused by the environmental loads analysed in the inventory.

In this frame, for instance in the case of acidification, a midpoint indicator is the acidification potential (AP) in the form of SO_2-equivalents, and the endpoints are the actual impacts that are caused by the protons released into the environment, e.g. fish death due to the acidification of lakes and timber loss due to the acidification of forests. For the case of human health effects, they can be expressed on the midpoint level as human toxicity potential (HTP), for instance in benzene-equivalents. On the endpoint level, damages to human health include cancer, cases of morbidity and Years of Life Lost (YOLLs) to express mortality.

These new approaches and the other recent developments are leading to advances in the practice of LCIA, but the major limitations and uncertainties cannot be expected to go away in the near future (Bare et al., 1999). They only can be made obvious by systematic uncertainty analysis and communicating the results in an adequate manner. It would be an important step further if one encompassing framework for midpoints and endpoints could be developed, including the most important variables of both types of approaches, thus enabling modelling along the two approaches and comparing the results with each other. This may well imply to integrate other environmental management tools. The level of sophistication might depend upon the type of application and the availability of data.

The governing dimensions of applications in relation to environmental damage estimations in industrial process chains are site-speciality, time scale and the need for certainty, transparency and documentation. In order to allow a high level of site-speciality and accuracy, this thesis differentiates the type of life cycles with respect to their process number and establishes a differentiation between industrial process chains and complete product systems.

As example of an industrial application, for which usually an ERA and sometimes an IPA and a LCA are carried out, the waste incineration process was studied. The waste incineration process chain is an end-of-life cycle. Hence, it is not a full product system that, however, is studied in a chain-perspective by a conventional LCA; and it is a system with more than one site-specific process that, however, is studied mainly by ERA for the incineration. For this type of applications, no particular methodology that combines environmental damage estimations and the chain-perspective was reported by the literature.

Only little efforts have been made so far to systematically explore the inherent uncertainties, interfaces and possibilities for integration and communication of the mentioned chain-orientated and site-orientated environmental assessment methods in the case of such industrial process chains. This is what this thesis has in mind.

1.2 OBJECTIVES

The research question of this thesis is:

- In which way environmental damages of industrial process chains can be estimated the most accurately possible, but still with acceptable efforts and in a communicable way?

That means that the objective of this thesis is to find an adequate trade-off between process chain-orientated and site-orientated environmental impact assessment and to convert environmental damage estimates in meaningful results like environmental costs.

This key question involves the following subsequent questions:

- What are the uncertainties in environmental damage assessment methods?
- What possibilities are there for interfaces and integration of environmental management tools, especially LCA, ERA, IPA and environmental costs?
- How can midpoint and endpoint approaches in LCIA be put in a common framework?

Evidently, all these questions are part of what is called sophistication of environmental management tools and need complex models to be answered.

1.3 OUTLINE OF THE STUDY

The present study is divided into four parts, which are a short introduction, the current state of the art, the research work carried out, and finally discussions and conclusions. It is composed by a total of 8 chapters.

Part B presents an evaluation of commonly used methods. Chapter 2 familiarises the reader to the basic concepts, tools and technical elements behind environmental damage estimations in industrial process chains. The portrayed methods are critically reviewed. Chapter 3 introduces the reader to the case study on waste incineration. The fundamental data are shown and the results are presented that have been obtained by the application of commonly used methods for the environmental damage estimations in industrial process chains.

The critical review in Chapter 2 opens with an overview of environmental management concepts and tools. Here concepts like sustainable development and life cycle thinking are presented, and the so-called toolbox for environmental management is explained. Its application-dependency in a process chain perspective is shown.

In general, all analytical environmental impact assessment methods include an effect analysis that indicates the severity of the consequences due to the pollution. Crucial elements are the dose-response and exposure-response functions. These functions are represented partly here with a special focus on human health to give an idea of their characteristics. Moreover, each analysis can include weighting of the relation between different types of effects such as cancer and respiratory health effects. One weighting option consists of costs that are an important aspect in environmental management. Different types of costs exist and are outlined here. A special focus is given on externalities; that means on external environmental costs. The monetisation approaches to determine external environmental costs used in this study are brought up. The relation of monetisation to other weighting schemes is demonstrated.

Then the commonly used chain-orientated environmental impact assessment tools are further explored. This is Life Cycle Assessment (LCA) with the two main parts Life Cycle Inventory (LCI) Analysis and Life Cycle Impact Assessment (LCIA). In connection with the section on LCI, the LCA-commercial software used in this work is described. On the subject of LCIA first the general framework is presented and then several methods are explained and discussed; the focus is made on those applied in this report. LCIA was traditionally carried out with impact potentials or so-called midpoint approaches, and in principle it is site-generic. Methods with a single environmental performance index that include weighting across impact categories are often based on these potentials. New advances include LCIA damage-orientated or endpoint approaches. Finally, the latest developments in LCIA are characterised as sophistication, which is considered to be the ability to provide very accurate and comprehensive reports that reflect the potential impact of the stressors to help decision-making in each particular case.

In opposite to these site-generic tools, site-specific environmental impact assessment methods are presented. As main analytical tools for this, Environmental Risk Assessment (ERA) and Impact Pathway Analysis (IPA) are described. While the traditional ERA method focuses on the evaluation of the probability of hazard occurrence, the more recently developed IPA offers a framework for damage estimations. The application of the ERA methods in this study needs complex technical elements like Gaussian dispersion and multimedia fate & exposure models, which therefore are described here. Moreover, an introduction is given to the integrated impact assessment software model EcoSense, which is used in the IPA.

Finally, the last section of Chapter 2 analyses the character of life cycles with respect to their process number and establishes a differentiation between industrial process chains and complete product systems. The relevance of this differentiation for the way of assessing environmental impacts is studied.

In Chapter 3 the reader learns about the case study and the results obtained by the applications of the commonly used methods. These data and the lessons learned by these applications are the basis for all applications of the further methodology development. First, the technical data of the Municipal Solid Waste Incinerator (MSWI) of Tarragona (Spain) and its process chain are presented for two situations: with and without an advanced acid gas treatment system. Then, a modular MSWI model is briefly described that allows to generate a scenario of a MSWI with the additional control device to eliminate NOx called DeNOx. After that, the results of the Excel-based LCA study for the MSWI process chain are shown. The results are compared with those obtained by an application with TEAM and a similar study published in the literature. Moreover, the question of avoided environmental loads due to benefits for the energy generation is discussed. Then it is verified that this life cycle can be defined as an industrial process chain, and not as a complete product system. In the following step, the outcomes of site-specific impact assessments of the MSWI emissions are shown. Initially the necessary geographic information is provided. Then, the cancer risk is represented (obtained by ERA). Next, the avoided external environmental costs after the installation of the advanced acid gas treatment are displayed (obtained by IPA). Finally, this same approach is further applied on a local scale with more detailed data.

Part C outlines the different steps of the methodology development made in the thesis. Several components of it have been published in scientific journals or are currently undergoing the publication process for research articles, e.g. Sonnemann et al. (2000). In general terms, Chapter 4 concentrates on the uncertainty analysis by Monte Carlo simulation; Chapter 5 presents the methodology of environmental damage estimations in industrial process chains; and finally chapter 6 focuses on site-dependent impact assessment by statistically determined generic classes.

Chapter 4 illustrates the uncertainty assessment by Monte Carlo simulation. The existence of different types of uncertainties is often mentioned as crucial limitation for a clear interpretation of environmental damage estimations, especially when carried out with complex software tools. Therefore, a strategy for stochastic uncertainty assessment by Monte Carlo simulation is presented. A framework for such type of assessment is presented for the Life Cycle Inventory and the Impact Pathway Analysis. The procedure is applied to the Life Cycle Inventory of the waste incinerator case study. In the same way the uncertainties are analysed in the application of the Impact Pathway Analysis on a local scale to the incinerator emissions of the case study. This parallel application allows to compare the magnitude of uncertainties in Life Cycle Inventories to those in Impact Pathway Analysis.

Chapter 5 provides a mathematical framework and a flowchart that allows spatial differentiation at different levels of detail and proposes an integration of LCA, IPA and environmental costs. It is especially applicable to so-called industrial process chains, i.e. not full life cycles of product systems; therefore, the established methodology is called "Environmental Damage Estimations for Industrial Process Chains". First, the challenges for such a methodology are identified, and then the strategy to overcome them is presented. An important part of this methodology is its mathematical framework, which core consists of the flowchart. Different options are given to find the adequate trade-off between site-generic and site-specific environmental impact assessment and to convert environmental damage estimates in meaningful results like environmental costs.

Initially, an overview of the whole methodology is given; the methodology puts midpoint and endpoint approaches in LCIA in a common framework. In this way the direct comparison of midpoint and endpoint indicators results is made possible. Then, the flowchart is divided into several parts, which are in somehow similar to the LCA phases, but which have often a different content. The first phase is Goal and Scope Definition, the second Life Cycle Inventory, and the third Life Cycle Impact Assessment. Here the similarity with the LCA framework stops; instead of interpretation the next steps are Dominance Analysis and Spatial Differentiation, Fate & Exposure Assessment and Consequence Analysis as well as Damage Profile and Eco-efficiency. In the proposed methodological framework a number of analytical tools like ERA and technical elements like uncertainty assessment by Monte Carlo simulation, as proposed in chapter 4, are used. The methodology is applied to the results of the LCA study of the waste incinerator process chain, in order to show its applicability. Finally, applications like end-of-life management and others are suggested as of particular interest for the established methodology.

Chapter 6 further develops a framework for site-dependent impact assessments in a life-cycle perspective. Here an approach focused on site-dependent impact assessment indicators as a way to find a compromise between site-specific damage endpoint assessments and potential midpoint life cycle indicators is used. The main reason for the discordance between the calculated impact potentials and the expectation of actual impacts in LCIA is on the one hand the no consideration of the spatial distribution of the receptors, and on the other hand the lack of information on the point of emission and the corresponding dispersion conditions in the respective medium, which are relevant for the occurrence of actual impacts. To relate these main factors, the present method uses representative generic impact classes corresponding to several receptor distributions and dispersion conditions, based on statistical reasoning. The proposed general framework is applied in detail for human health effects due to airborne emissions in the Mediterranean region of Catalonia. Site-dependent impact factors are calculated on a district level. A system to estimate the human health effects for other Spanish regions based on the results for Catalonia is added. Finally, sources of uncertainties are identified and the site-dependent indicators developed for Catalonia are applied to the waste incineration process chain, too. This is done using the methodological framework presented in Chapter 5, with the aim of comparing the importance of transports in relation to the waste incineration process.

Part D closes the thesis with an overall discussion in Chapter 7 as well as the conclusions and the outlook in Chapter 8.

The chapter 7 discusses the limits of the presented methodological framework and tries to evaluate the inherent uncertainties of the methods based on the detailed uncertainty assessments carried out. Limits are evidently the efforts that are necessary nowadays to carry out the developed methodology and, hence, the number of processes in a life cycle, what, however, could quickly change due to the speedy development of the information society. As a current alternative, a strategy for generic applications is presented. This strategy refers to the development of an internationally accepted framework for midpoint and endpoint LCIA approaches. A special attention is given in Chapter 7 to highlighting what is considered to be new findings.

In Chapter 8 the general conclusions are presented with regard to the objectives of the thesis. Finally, interesting outlooks are offered for future research.

Part B:

Evaluation of commonly used methods

2 CRITICAL REVIEW OF COMMONLY USED METHODS

2.1 OVERVIEW OF ENVIRONMENTAL MANAGEMENT CONCEPTS AND TOOLS

Initially the environmental policy with regard to industry had the exclusive objective to maintain the emissions to the different compartments under control. It was thought that the so-called end-of-pipe technologies would be sufficient to limit the environmental impact.

However, through the years, it was observed that this approach was insufficient to stop the progressive degradation of the environment, due to the fact that it is not sufficiently flexible for an industry that has to react quickly to changing social and business conditions.

It is necessary to make a substantial progress; that is to introduce the environmental considerations in all aspects of the industrial management practices: in all phases of production, commercialisation, use and end-of-life of a product.

Based on these reflections, different general objectives have been formulated as programmes that intent to encompass the idea of good environmental management. Especially these are:

Sustainable Development
The term Sustainable Development was first published in the "Brundtland" report and was established in the Agenda 21 at the UN Conference on Environment and Development in Rio de Janeiro in 1992 as the overall aim also for environmental management. Sustainable Development means satisfying the needs of the present generation without compromising the needs of the future. It includes taking into account three aspects:

1. Economic: We need economic growth.
2. Environment: We need to minimise environmental damage, pollution and exhaustion of resource.
3. Social: This is equity; the world's resources should be better shared between the rich and the poor. There should be good labour conditions and the civil rights should be respected.

Eco-efficiency
Eco-efficiency is reached by the delivery of competitively priced good and services that satisfy human needs and bring quality of life, while progressively reducing ecological impact and resource intensity throughout the life cycle, to a level at least in line with the earth's estimated carrying capacity (WBSCD, 1999; Simone & Popoff, 1997).

To improve their eco-efficiency companies can:

- Reduce the material intensity of goods and services
- Reduce the energy intensity of goods and services
- Reduce toxic dispersion
- Enhance materials recyclability
- Maximise sustainable use of renewable resources
- Extend product durability
- Increase the service intensity of goods and services

Pollution prevention
Pollution prevention means source reduction - preventing or reducing waste where it originates (at the source) - including practices that conserve natural resources by reducing or eliminating pollutants through increased efficiency in the use of raw materials, energy, water, and land (US-EPA, 2000a).

2.1.1 CONCEPTS

The presented overall objectives, in line with the strategies of companies to establish environmental strategies, are reflected in various concepts:

Green Chemistry
Green Chemistry consists in the use of chemistry for pollution prevention. That means Green Chemistry has the objective to design chemical substances that respect the environment and that are at the time produced by environmental friendly processes (US-EPA, 2000b).

Cleaner Production
The concept of Cleaner Production was introduced by UNEP-IE (United Nations Environment Programme, Division Industry and Environment) in 1989. Cleaner Production is the continuous use of an integrated and preventive environmental strategy. It is applied to processes, products and services to increase the eco-efficiency and reduce risks to the population and the environment (UNEP DTIE, 2000).

Total Quality Environmental Management
Total Quality Management (TQM) stems from the area of quality management and describes the procedure to follow, according to the quality standard ISO 9000. There are lot of enterprises that have implemented TQM as Ford, Phillips or Motorola (Hansen, 2000).

Life Cycle Thinking

Life Cycle Thinking is a way of addressing environmental issues and opportunities from a system or holistic perspective. In this way of thinking, a product or service system is evaluated with the goal of reducing potential environmental impacts over its entire life cycle, as illustrated in Figure 2.1. Life Cycle Thinking does not generally normalise the results to a functional unit, as is done as part of a Life Cycle Assessment study. The concept of Life Cycle Thinking implies the linking of individual processes to organized chains starting from a specific function. Life Cycle Thinking implies that everyone in the whole chain of a product's life cycle, from cradle to grave, has a responsibility and a role to play, taking into account all the relevant external effects. From the exploitation of the raw material that will constitute a new product, through all the other processes of extraction, refining, manufacturing, use or consumption to its reuse, recycling or disposal, individuals must be aware of the impact that this product means to the environment and try to reduce it as much as possible. The impacts of all life cycles stages need to be considered when taking informed decisions on the production and consumption patterns, policies and management strategies (UNEP, 1999).

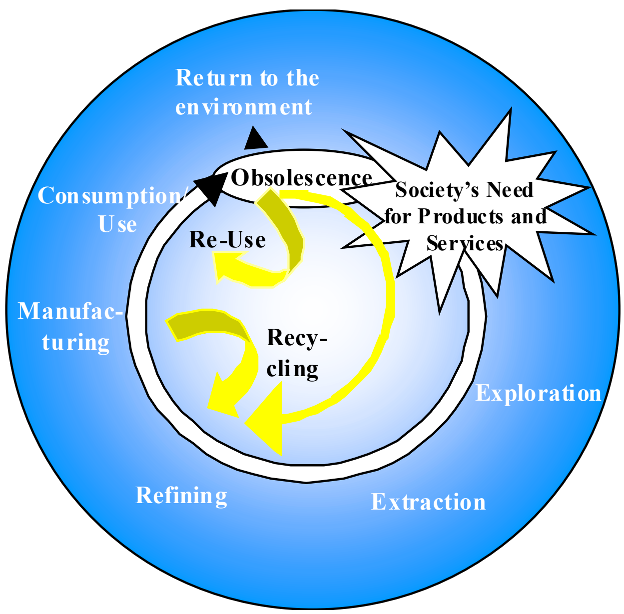


Figure 2.1: The life cycle of a product system (UNEP/ SETAC, 2001)

Design for Environment (DfE)

The Design for Environment allows to address the environmental problems due to a product in the design phase; that means that the environmental variable is considered as one requirement more of a product, additionally to the other conventional design objectives: costs, utility, functionality, security, etc. The basic idea is to fabricate the product with the smallest environmental load possible along its life cycle (Fiksel, 1997; Brezet & Hemel (1997); Rodrigo & Castells, 2002). It includes dematerialisation (DM) and detoxification (DT); that means the reduction of the quantity and the toxicity of the materials used in the manufacturing of the product.

Industrial Ecology

The Industrial Ecology means an approximation of the industrial systems to natural systems. It is about analysing systematically the material and energy flows of the industrial systems with the objective to minimise the generation of waste and environmental impacts (Graedel, 1994).

2.1.2 TOOLS

The presented concepts that have been developed to give an orientation for the environmental management are quite abstract and for its transfer into action tools are necessary that make the environmental aspects more concrete, taken into account the economical, social and technological information. The working group on conceptually related methods in the Society of Environmental Toxicology and Chemistry (SETAC) has distinguished the following types of tools (De Smet et al., 1996):

- Political instruments
- Procedural tool
- Analytical tools

In general, the tools need technical elements, like dispersion and other pollutant fate models as well as damage functions and weighting schemes, and available data, e.g. about emissions and resource consumption as well as the technical specifications and the geographic situation, to be carried out. The application of these tools provides consistent environmental information that facilitates an adequate decision-making towards the sustainable development. In Figure 2.2 an overview of the conceptually related methods in environmental management is presented.

Based on the idea of the interaction between these different concepts and tools, a concerted action named CHAINET took place from 1997 to 1999 under the Environment and Climate programme of the European Union. The mission of this action was to promote the common use of the different tools and to facilitate the interchange of information between the relevant stakeholders (CHAINET, 1998).

For environmental damage estimations in industrial process chains the relevant methods are the analytical tools:

- *Cost-Benefit Analysis* (CBA) and *Cost-Effectiveness Analysis* (CEA) are techno-economic tools to support decision-making towards sustainability. They refer to environmental costs, a topic that will be put in plain words in the next section 2.2.
- *Life Cycle Assessment* (LCA) is a tool standardised according to ISO series 14040 for product-orientated environmental impact assessment and will be further explained in section 2.4.
- *Environmental Risk Assessment* and *Impact Pathway Analysis* (ERA and IPA) are the tools that in general are used for the impact analysis in site-specific environmental impact assessment; these tools will be described more in detail in section 2.4.

- The *Process Simulation* and the related re-engineering is an important tool for the improvement of industrial processes. It allows to foresee the effects on the environment of changes in the process design before its realisation.

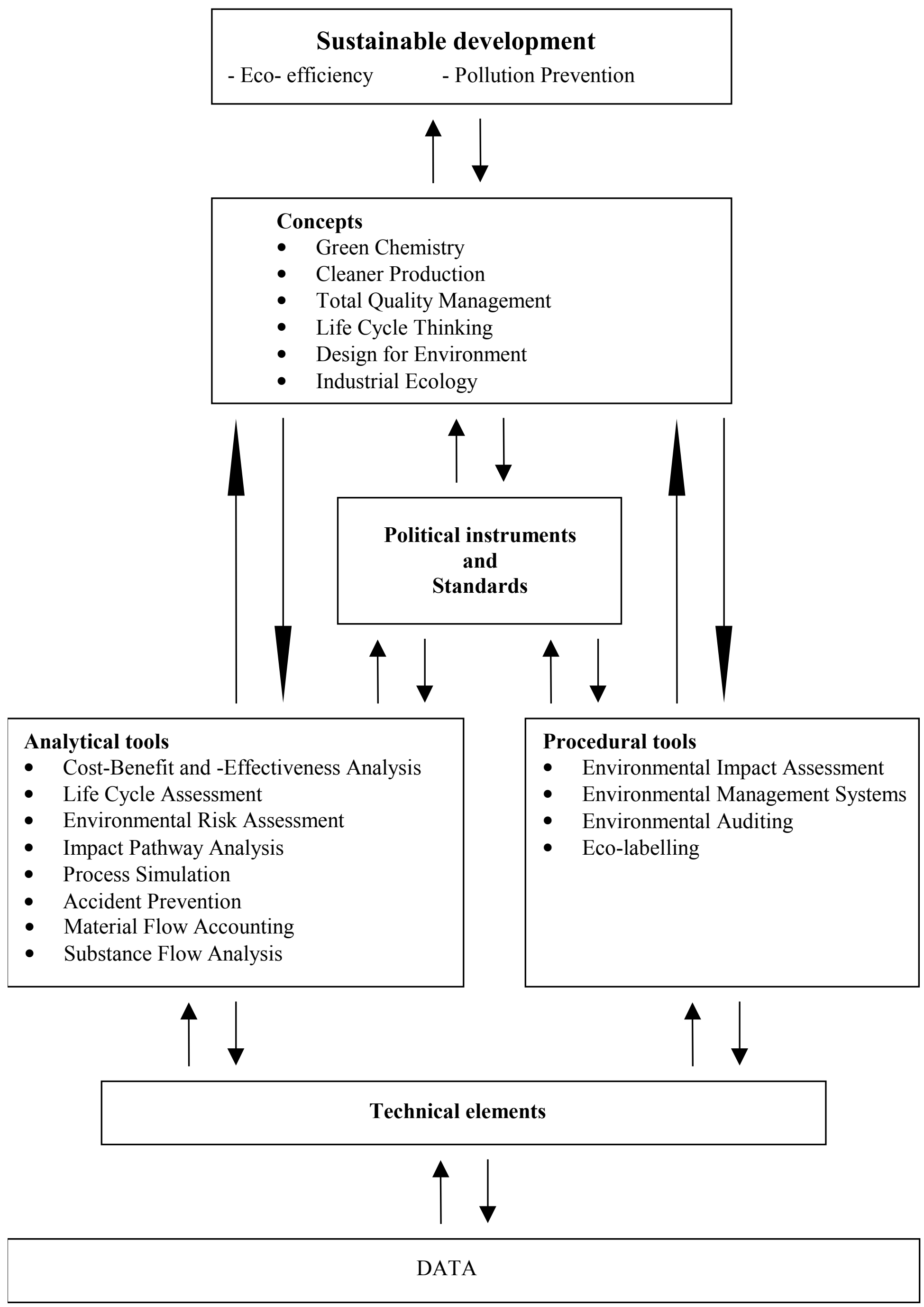


Figure 2.2: Conceptually related methods in environmental management (adapted from De Smet et al., 1996):

- As part of the establishments of national statistical accounts the *Input-Output Analysis* (IOA) has been developed in the 1930s. One of the main uses of input-output analysis is to display all flows of goods and services within an economy, simultaneously illustrating the connection between producers and consumers and the interdependence of industries. Starting in the beginning of the nineties, environmental applications have been developed. Nowadays, this macro-economic method is frequently applied in environmental analysis (Proops et al., 1993).
- For the *Accident Prevention* it is necessary to determine the environmental risk that signifies the installation and operation of an industrial process, due to undesirable events. Undesirable events are caused by unforeseen emissions of pollutants due to accidental reasons. A comprehensive method has been developed to analyse systematically the undesired events or accidents (AICHE, 1985).
- *Material Flow Accounting* and *Substance Flow Analysis* are tools for the study of the pollutant behaviour in a determined region. Material Flow Accounting tries to quantify the extraction, production, transformation, consumption, recycling and disposal of materials in a specific region. The Substance Flow Analysis is about the material balance for one chemical substance in one region (Bringezu et al., 1997).

Important procedural tools are:

- According to legal definitions like the European Council Directive 85/337/CEE, modified by the Council Directive 97/11/CE related to the assessment of the negative effects of determined public and private project on the environment, *Environmental Impact Assessment* (EIA) is the procedure to carry out studies with technical analysis that allows to assess the effects that the realisation of a certain project would cause on the environmental. The idea behind Environmental Impact Assessment is to get an objective judgement of the consequences due to the impacts generated by the realisation of a determined activity. The main part of such an evaluation is the environmental impact study (Coneza Fdez., 1997).
- The *Environmental Management and Audit* (EMA) scheme is a EU-based system based on the Council Regulation (EEC) 1836/93 for the continuous improvement of the environmental aspects in companies. Internationally it corresponds in many features to ISO 14001 (1996), which however does not have the same recognition of the environmental authorities (Zharen, 1995).
- The idea of *Eco-labelling* (ELG) is to guarantee the environmental quality of certain properties or characteristics of the products that obtained the eco-label (Alfonso & Krämer, 1996). In this way better information on green products should be provided to the consumers and the design for environment should be promoted (EC, 1997). A EU Eco-Label Scheme is laid down in the Council Regulation (EEC) 880/92 (EC) 1980/2000.

2.1.3 APPLICATION-DEPENDENCY IN A PROCESS CHAIN-PERSPECTIVE

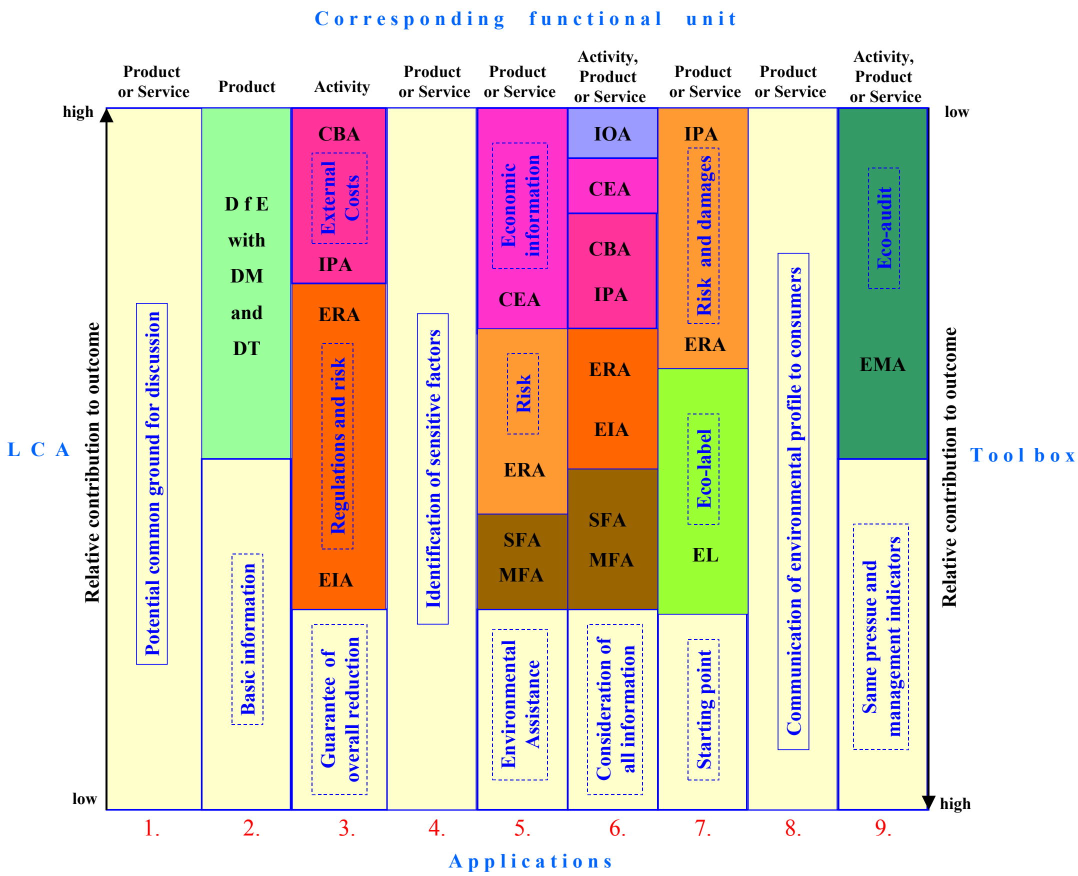
According to Wenzel (1998) the governing dimensions for applications of LCA are:

- Site-speciality
- Time scale
- Need for certainty, transparency and documentation.

Possible applications can be positioned in relation to these governing dimensions. In line with this, Sonnemann et al. (1999a) have examined in which cases LCA it is an integrated element of another concept and for which goals other tools of environmental management should accompany it. By the proposed guide in the form of a matrix (Figure 2.3) the environmental practitioner should be enabled neither to overestimate the possibilities of LCA nor to be discouraged to use it because of its inherent limitations.

1. *Education and communication*. LCA supplies a potential common ground or basis for discussion and communication. All groups in the society need to understand their individual responsibilities for improvements.
2. *Product development and improvement*. The concept used in the field of environmental friendly product (re)design and development is called design for the environment. LCA provides the information to support it.
3. *Production technology assessment*. LCA helps to ensure that overall reductions are achieved and pollutants are not shifted elsewhere in the life cycle, but for the assessment of the actual impacts of the technology other tools are needed.
4. *Improving environmental program*. LCA can be particularly effective at identifying sensitive factors such as the number of times a reusable must be returned, possible energy recovery etc.
5. *Strategic planning for a company's product or service line*. LCA can assist the strategic planning process, especially when coupled with other tools providing economic and risk information.
6. *Public policy planning and legislation*. LCA studies can be used to assure that all relevant environmental information is considered. Because of LCA's restriction to potential impacts the results should be integrated with data form other tools.
7. *Environmental friendly purchasing support*. LCA is obviously a starting point for eco-labelling, but the loading and resource indicators are clearly not a true eco-balance. LCA information should be complemented by data form other tools.
8. *Marketing strategies*. By LCA it is possible to develop an environmental profile of a product or service, which can be communicated to the consumers.
9. *Environmental performance and liability evaluation*. The combination of an environmental management system with the LCA is an interesting topic for the future. For this, it is necessary to use the same pressure and management indicators.

Hence, it can be said that the life cycle thinking is the right concept to evaluate the environmental impacts of a functional unit (product, service or activity) but LCA is often not the only tool that is to consider. The practical guide (Figure 2.3) developed in the present study by Sonnemann et al. (1999a) for environmental decision-makers helps to select the tool that corresponds to a particular application.



1. Education and communication.
2. Product development and improvement.
3. Production technology assessment.
4. Improving environmental program.
5. Strategic planning for a company's product or service line.
6. Public policy planning and legislation.
7. Environmental friendly purchasing support.
8. Marketing strategies.
9. Environmental performance and liability evaluation.

Legend

CBA: Cost-Benefit Analysis	**EIA**: Environmental Impact Assessment	**IOA**: Input-Output Analysis
CEA: Cost-Effectiveness Analysis	**ELG**: Eco-labelling	**IPA**: Impact Pathway Analysis
DfE: Design for Environment	**ERA**: Environmental Risk Assessment	**LCA**: Life Cycle Assessment
DM: Dematerialisation	**EMA**: Environmental Management and Audit	**MFA**: Material Flow Accounting
DT: Detoxification		**SFA**: Substance Flow Analysis

Figure 2.3: Matrix guide for the inclusion of the life cycle concept in environmental management practises (Sonnemann et al., 1999a)

2.2 CONSEQUENCE OR EFFECT ANALYSIS PARTICULARLY FOR HUMAN HEALTH EFFECTS

2.2.1 EXPOSURE-RESPONSE AND DOSE-RESPONSE FUNCTIONS

Common to all of the assessment conducted on impacts of pollutants emitted is the need for modelling the dispersion of pollutants, and the use of dose-response (D-R) and exposure-response (E-R) functions, also called sometimes damage functions in the effect analysis. These terms are used somewhat loosely in this context. As what we are talking about in the Impact Pathway Analysis (IPA), see section 2.4.4, is the response of a given exposure of pollutant in terms of atmospheric concentration rather than an ingested dose as in Environmental Risk Assessment (ERA) see section 2.4.1, the term dose-response that stems from ERA is comparable to the term exposure-response in IPA. The special focus in this chapter is on human health effects, however some general principles are also valid for other types of impacts. Here only a few other examples will be mentioned.

These functions are based on toxicological dose-orientated and epidemiological exposure-orientated studies. The Figure 2.4 intends to illustrate the difference between the toxicological and the epidemiological approach. While the toxicological approach is based on bioassays or animal test that allow to determine dose-response functions, the epidemiological approach uses empirical studies where correlations are established between exposure situations and observed physical impacts. In this way exposure-response functions, which allow a prevision of physical impacts depending on the exposure concentration, are calculated. Epidemiological studies focus more on macropollutants responsible for respiratory effects as SO_2, NOx and particles. The dose-response functions permit the determination of risk due to the accumulation of pollutants in the human organism. The risk is a way to foresee the probability of physical impacts. Bioassays are the fundament to obtain toxicological information for micropollutants, i.e. heavy metals and the huge number of organic compounds like PCDD/Fs or PAH. Potential cancer risk factors are detemined by both approaches. Both types of damage functions are important sources of uncertainties; especially the question of extrapolation to lower doses and the correlations made implies insecurity. Based on the uncertainty analysis in chapter 4, in our study the uncertainties due to these functions will be compared to other sources of uncertainties in environmental damage estimations in industrial process chains.

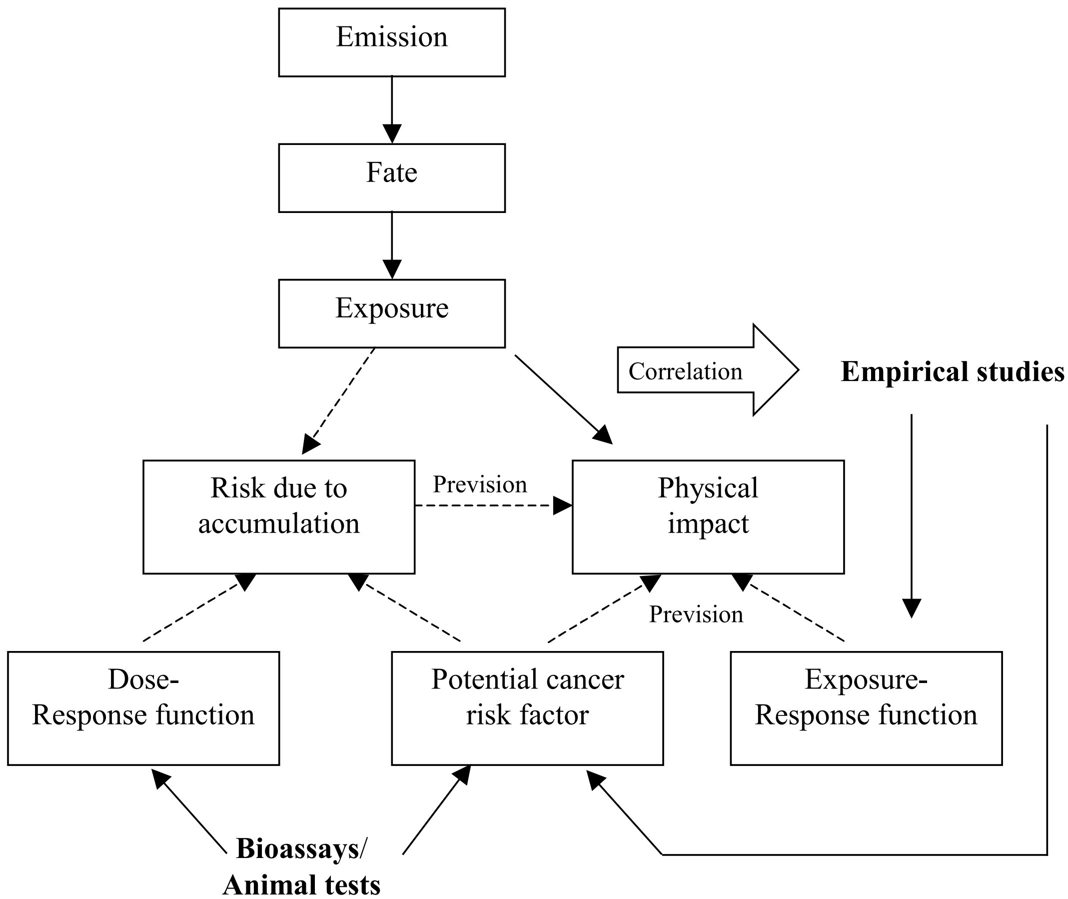


Figure 2.4: Comparison of the toxicological approach by bioassays/ animal tests and the epidemiological approach using empirical studies to determine damage factors for human health impacts*

Dose-response functions

The characterisation of the risk is done by using effect factors for the different pollutants. The risk of carcinogenic pollutants is estimated with a factor of potential cancer (expression (2.1)), the risk of non-carcinogenic pollutants is characterised by a reference dose (Figure 2.5). The reference dose (RfD) is estimated for all pollutants and all ways of exposure (STQ, 1999).

$$\text{Cancer Risk}=\text{Dose}_{\text{day, average}}\ [\text{mg}/(\text{kg}\times\text{day})]\times\text{Cancer Factor}\ [\text{mg}/(\text{kg}\times\text{day})]^{-1} \quad (2.1)$$

* In principle, also exposure-response functions can be used for thre calculation of risk. However, in the impact pathway analysis they are used directly for damages estimations.

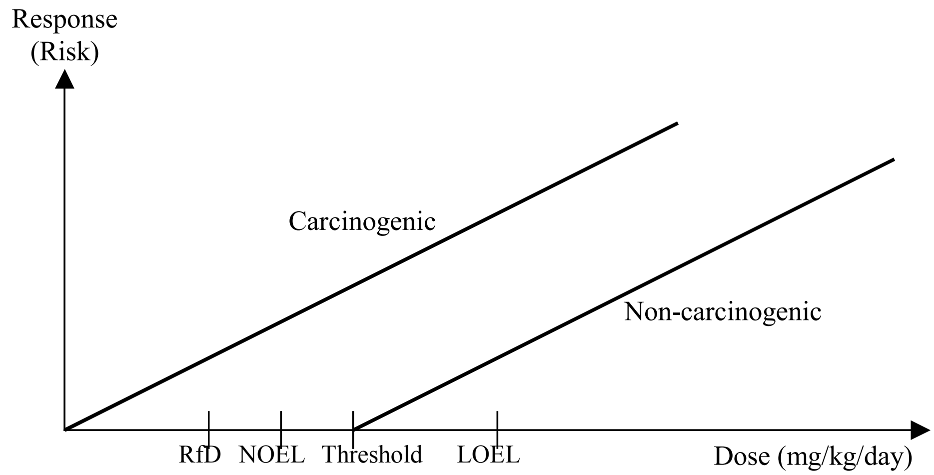


Figure 2.5: Dose-response functions for carcinogenic and non-carcinogenic pollutants (Masters, 1991). This is a schematic presentation; generally these curves are not linear.

A controversial point in the discussion about carcinogenic effects is the estimation of the D-R functions for different pollutants. Animal tests are used administering various doses to observe an effect of the toxin. The doses in these tests are higher then those existing in the environment. The discussion is about the possibility of an extrapolation from the higher concentration in the animal test to the lower concentration in the environment and the possible effects. Many mathematical models exist to extrapolate to the lower level, but there is a lot of uncertainty. With the main aim of protecting human health, the US-EPA chooses the safer way to estimate the risk with an additional correction factor, which means an overestimation of risk (Olsen et al., 2000).

For the non-carcinogenic substances a threshold exists. In order to get knowledge of this threshold it is necessary to conduct experiments with animals by changing the dose. The minimal dose where appears a special effect is called LOEL (Lowest observed effect level). The maximum dose without any special effect to the test animal represents the NOEL (No Observable Effect Level) (Olsen et al., 2000).

The Reference Dose (RfD) is defined as an estimate of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime (Barnes & Dourson, 1988). However, it should not be concluded that all doses below a reference dose, or concentration, are acceptable. Despite this caution, most non-cancer health studies adopt the reference dose as an adequate standard to be met. Residual risks associated with doses at, or below, such standards are not commonly estimated.

As indicated in expression (2.2), the RfD is derived by dividing the NOEL (or LOEL) by uncertainty factors and a modifying factor. Separate adjustment factors are specified for each of several extrapolations, e.g. from average to sensitive individuals, from animal studies to humans, from sub-chronic to chronic exposure durations, and to account for the quality/ breadth of the database. A factor of 10 is usually the default value for the uncertainty factors. Values less than 10 are sometimes used when sufficient data and justifications are available. The LOEL is only used in the absence of a NOEL, and an additional adjustment factor (UF_L) is required in this instance to compensate for the lack of a NOEL estimate. For instance, the use of a large number of animals in a study may enhance the certainty in the NOEL, resulting in the use of a modifying factor less than 1 but greater than 0 (Beck et al., 1994).

$$RfD = NOEL \text{ (or LOEL)} / (UF_H * UF_A * UF_S * UF_L * UF_D) \quad (2.2)$$

where RfD = Reference Dose [mg/kg-day]
NOEL = No Observable Effect Level [mg/kg-day]
LOEL = Lowest Observable Effect Level [mg/kg-day]
UF = Uncertainty factor [-]
H = variation in sensitivity in the human population (intraspecies variability)
A = for extrapolating animal data to humans (interspecies variability)
S = for extrapolating chronic toxicity from sub-chronic studies
L = uncertainty in extrapolating from a LOEL rather than from a NOEL
D = reflecting the adequacy of the total database (Dourson et al., 1992, Evans & Baird, 1998; Barton et al., 1998)

The purpose of the adjustment factors is often misinterpreted as being exclusively focused on adding a margin of safety to RfD calculations. In fact, the factors also serve as an adjustment for what are thought to be real and systematic (Crettaz et al., 2002).

Altogether the values of the RfD represent a dose of approximately 1000 times below the value where it is not possible to observe an adverse effect (NOEL) in the animals. The RfD has the unit milligram of a diary intake of the toxic, which is absorbed in the body, divided through the body weight (Olsen et al., 2000).

It has to be said that the RfDs have been criticised to be inappropriate for LCIA purposes (Crettaz et al., 2000). The US-EPA uses deterministic factors, usually factors of ten, to extrapolate RfDs from measures such as the NOEL. The factors may include an inconsistent margin of safety across different chemicals, contributing to RfDs varying in terms of their conservatism. Probabilistically accounting for the variability of extrapolation factors, Baird et al. (1996) analysed the RfDs reported in the US-EPA's Integrated Risk Information System database (IRIS, 1996). Of 231 RfDs evaluated, 56% were below the 5th percentile in corresponding uncertainty distributions, 44% were between the 5th and 15th percentile and 3 were above the 15th percentile. Such a conservatively biased approach is consistent with the objectives of many risk-screening assessments. However, inconsistent levels of conservatism can undermine the objectives of an LCA where in general residual risks below regulatory requirements are estimated. A procedure based upon central estimates with measures of associated uncertainty is preferable (Krewitt et al., 2002).

Table 2.1 shows the RfD values used in the ERA case study presented in chapter 3 for some heavy metals and dioxins and furans (PCDD/Fs). For the carcinogenic substances the potential cancer factors are presented, a difference is made between the oral exposure and the exposure by inhalation.

Table 2.1: Example of RfD values and potential factors used in the ERA case study (IRIS, 1996)

	RfD [mg/day/kg]	Cancer factor Oral [kg*day/mg]	Cancer factor Inhalation [kg*day/mg]
As	3.0E-04	1.75	50
Cd	5.0E-04		6.3
Cr	1.0E+00		42
Ni	2.0E-02		1.19
Pb	6.0E-03		
Hg	3.0E-04		
Sn	6.0E-01		
Zn	3.0E-01		
PCDD/Fs	4.0E-09	1.56E+05	1.16E+05

For a general environmental health risk assessment of an unspecified mix of emitted pollutants it is necessary to separate the population in different groups depending on the sensibility of the persons, for instance babies, children, adults, adults above 65 years, persons with asthma, etc. Especially for the assessment of non-carcinogenic pollutants with the effect of chronic illness a separation is important. In the case of carcinogenic pollutants it is generally sufficient to separate the population in the two groups of adults and children (until a certain age) to consider the different physical conditions (e.g. breathing, surface of the body, etc.) and the different habits of living (e.g. the playing of the children outside on the meadow).

Crump (1984) proposed the benchmark dose method as an alternative to the NOEL approach; the method has been extended by Kimmel & Gaylor (1988). This alternative is supported in the US-EPA's Benchmark Dose Technical Guidance (US-EPA 1996a). In their guidelines the US-EPA proposes to use the benchmark dose (BMD10) as a basis for deriving the reference dose.

WHO (1987) introduced a model that since then has been known as the unit lifetime risk or unit risk concept. This model was already in use at that time within US-EPA under the name of "average relative risk model" (US-EPA, 1993). The WHO model can be characterised by the unit risk factor for inhalation, which is an estimate of the probability that an average individual will develop cancer when exposed to a pollutant at an ambient concentration of 1 μg/ m3 for the individuals life (70 years).

Exposure-response functions

From studies of former smog pollution episodes (e.g. London smog in the 1950^{ies}), it is known that very high ambient pollution concentration are associated to adverse health effects on the same day or on subsequent days. In the last 20 years numerous well-conducted epidemiological[1] as well as experimental[2] studies have confirmed this correlation between exposure to pollutants and the occurrence of health and environmental damages. Hence, they allow to establish a direct link between ambient concentration of certain pollutants and effects on different receptor endpoints, as indicated in Pilkington et al. (1997). A similar survey is also presented in Beer & Ricci (1999). Under the assumption of linearity of incremental damage with incremental exposure, slope factors (SF) could be defined. Figure 2.6 shows some possible forms of E-R functions as they have been found. E-R functions exist for human health effects, damages to material and crops, and the harm to ecosystems. The function with fertiliser effect means that for low doses no negative impacts occurs, but a positive one. That is the case for nitrogen deposition. However, sufficient epidemiological data may not exist to address human health effects caused by the majority of chemicals. Therefore, the epidemiological approach has to be combined with bioassays. Moreover, it has to be taken into account that epidemiological data are criticised by providing insights that may be limited to the identification of correlations. A correlation does not necessarily imply a causal relationship; in the case of human data from clinical studies and trials the causal association is considered to be higher by Crettaz et al. (2002).

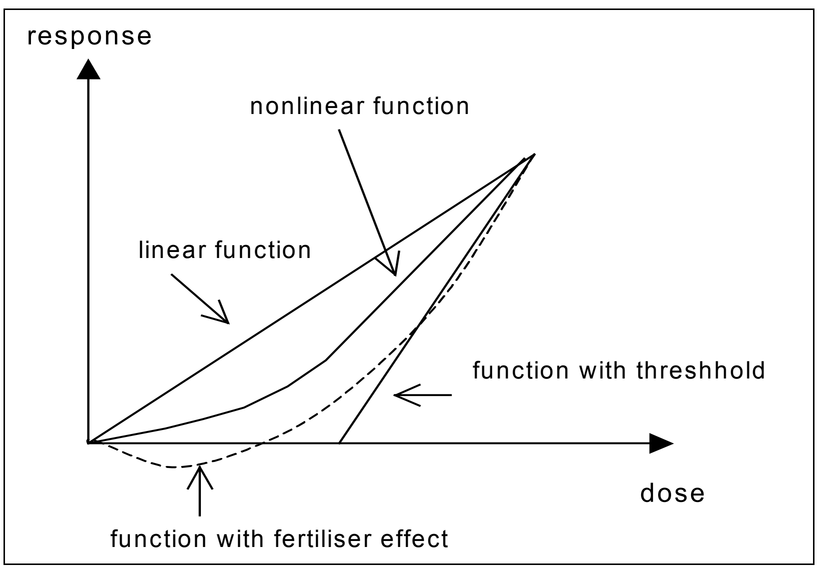


Figure 2.6: Possible forms of exposure-response functions (adapted from European Commission, 1995)

[1] epidemiological studies are statistical methods where causal coherences between ambient pollutant concentration and the occurrence of illness cases have been thorougly investigated

[2] experimental studies are laboritory studies with animals, whereby for the transfer of the obtaied results from animal to human safety factors are used

Human Health
According to European Commission (1995) and Mayerhofer et al. (1998) the E-R functions for human health can be subdivided into the seriousness of their effects:

Morbidity
This effect concerns the following primary and secondary pollutants: NO_x, SO_2, NH_3, CO, nitrate aerosol, sulphate aerosol and PM10. Possible impacts are among others hospital visits, bronchodilator use and chronic cough.

Cancer
This effect is caused by micropollutants like heavy metals, e.g. As, Cd, Cr, and Ni, as well as organic compounds, e.g. PCDD/Fs, PAH etc. Common for all of these substances is their high persistency in the environment and their transport from one media to another. The general form of the exposure-response function is as follows in expression (2.3) (EC, 2000) for all cancer effects. Table 2.2 lists the considered damage factors and puts them in relation to each other. Important differences can be observed. The factor of 70 annualises lifetime risk (assuming an average longevity of 70 years).

$$\text{No. additional cancers} = \Delta \text{ Concentration } [\mu g/m^3] * \text{damage factor} * \text{Population} /70 \quad (2.3)$$

Table 2.2: Overview of different damage factors for cancer (based on IER (1998); Hofstetter, 1998)

Pollutant	Cancer kind	Source	Damage factor [µg/m³]	Relative difference	Minimum	IPA* Case Study	Maximum
As	Lung, Skin, Liver	IRIS unit risk	2.0E-04	100%	2.0E-04	2.0E-04	
		LAI unit risk	4.0E-03	2000%			4.0E-03
		WHO unit risk	1.5E-03	750%			
Cd	Lung, Prostate	IRIS unit risk	1.8E-04	100%	1.8E-04	1.8E-04	
		LAI unit risk	1.2E-02	120%			1.2E-02
Ni	Lung, Nasal, Cavity	IRIS unit risk	4.0E-03	1053%		4.0E-03	4.0E-03
		WHO unit risk	3.8E-04	100%	3.8E-04		
PCDD/Fs**	Tissue, Lymphatic, Hematopoietic	LAI unit risk	1.4E+00	100%			

* Impact Pathway Analysis
** For damage factor [ng/m³]

Mortality
The effect of mortality can be expressed by fatal cases or Years of Life Lost (YOLL). However, since in recent years researchers have moved from studies based on fatal cases to Years of Life Lost (YOLL) (EC, 1995; EC 2000), also in this study YOLL is used as endpoint. The Table 2.3 is introduced to illustrate the highly important E-R functions that use YOLL as endpoint.

For example the additional YOLL by Sulphates proposed by Pope are calculated in the following way:

0.0012 YOLL/(μg/m^3) * 0.00082 μg/m^3)/TWh * 423,000 pers.* 0.75 adults/pers.= 0.312 YOLL/ TWh

where Δc = 0,00082 (μg/m^3)/TWh of electricity produced (concentration increase/ functional unit)
Pop. = 423,000 pers. (Population expressed as persons in region considered)
adults = 0.75 adults/pers. (Percentage of population which age above 18 years)

Table 2.3: Mortality functions expressed in YOLL for concentrations expressed in μg/m^3 (IER, 1998)

Receptor	Category	Reference	Pollutant	Formula
***adults**: percentage of population with age above 18*				
	'chronic' YOLL	Pope	PM10 / Nitrates	0.00072 * Δc * Pop. * adults
		Pope	Sulphates	0.0012 * Δc * Pop. * adults
	'acute' YOLL	London/Athens	SO_2	0.0719 * Δc * Pop. * b_m /100 * adults
		Köln/Amsterdam	PM10 / Nitrates	0.04 * Δc * Pop. * b_m /100 * adults
		Sunyer	Sulphates	0.0677 * Δc * Pop. * b_m /100 * adults
		Barcelona/ London	NO_x	0.0648 * Δc * Pop. * b_m /100 * adults

b_m: baseline mortality London, Köln, Athens, Amsterdam, Barcelona: mortality data of these cities

Crops and Material (man-made environment)
This part addresses the assessment of impacts due to the environmental damages on crops and material, as what in the present is defined as the AoP man-made environment. There are two basic pathways through which plants can be harmed by SO_2. The first is through foliar uptake of pollutants, and the second through effects of acid deposition on the soil. With damages to material are meant the damages that refer to surfaces, especially in buildings, bridges and cars due to acidic deposition. However, not included are cultural damages, as e.g. to ancient cathedrals, due to their intrinsic value as mankind's patrimony.

Ecosystems (natural environment)
This chapter addresses the assessment of impacts that affect ecosystems. Damage functions exist for acidic deposition on natural and semi-natural terrestrial ecosystems. That means it accounts for the impact corresponding to the sub-AoP biodiversity and natural landscapes. Currently the most widely applicable approach for the analysis of pollutant effects on terrestrial ecosystems is the critical load/ level approach. The following definitions of the terms critical load and levels were given by UN-ECE (1991).

Critical load: *"The highest deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on ecosystems structure and function according to the present knowledge."*

Critical levels: *"The concentration of pollutants in the atmosphere above which direct adverse effects on receptors, such as plants, ecosystems or material may occur according to the present knowledge."*

Critical loads have been defined for several pollutants and ecosystems. However, they cannot be used directly to assess damages per se, rather they simply identify the areas where damages are likely to occur. In the present study the Relative Exceedance Weighted (REW) ecosystem area approach is used to assess the environmental impact. The contribution of a specific source of pollutants to the exceedance of critical loads for ecosystems is analysed by taking into account pre-defined background conditions. The relative exceedance factor f_{RE} is the contribution of the concentration increase Δc due to the emission to the height of exceedance of the critical load divided by the critical load C_{CL} itself, see expression (2.4)). The relative exceedance weighted ecosystem area indicator REW is expressed in km^2 and is obtained by the multiplication of the relative exceedance factor by the ecosystem exceeded area A_{EE} where the critical load is exceeded by the pollutant, see expression (2.5) (IER, 1998).

$$f_{RE} = \Delta c / C_{CL} \quad (2.4)$$

$$REW = f_{RE} * A_{EE} \quad (2.5)$$

For example the additional REW by NOx are calculated in the following way for imaginative values of $\Delta c/$, C_{CL} and A_{EE}:

$f_{RE} = 0.00082\ (\mu g/m^3)/TWh / 100\ \mu g/m^3 = 0.0000082/\ TWh$

$REW = 0.0000082/\ TWh * 10{,}000\ km^2 = 0.082\ km^2/\ TWh$

where $\Delta c = 0.00082\ (\mu g/m^3)/TWh$ of electricity produced (concentration increase/ functional unit)
$C_{CL} = 100\ \mu g/m^3$ (Concentration corresponding to critical load)
$A_{EE} = 10{,}000\ km^2$ (Ecosystem exceeded area)

Comparison of the dose- and exposure-response functions in the different methods

Within the different methods presented in this study, dose- and exposure-response functions from diverse environmental authorities are used. Due to that fact, the risk, damage or characterisation factors evaluate distinctly the relation of the pollutants to each other. The Figure 2.7 gives an impression of the importance of that subjective element for the example of four assessed pollutants in the three impact assessment methods further studied in this work: Environmental Risk Assessment (ERA), Impact Pathway Analysis (IPA) and the Human Toxicity Potential (HTP) as an example of a Life Cycle Impact Assessment (LCIA) method. See sections 2.3 and 2.4 for further details about these methods.

The figure shows clearly that the methods depend strongly on the selection of their assessment factors. The comparison was carried out for PCDD/Fs and the heavy metals As, Cd and Ni. For a better overview a logarithm scaling was used. The indicated values express the relation to Arsenic.

In all approaches it is common that PCDD/F has a very high damage potential. The other pollutants are weighted differently. Arsenic is the second strongest pollutant for the assessments in ERA and HTP, in opposition to the IPA; in this approach dominates Nickel while in the other two methods this pollutant has a low damage factor.

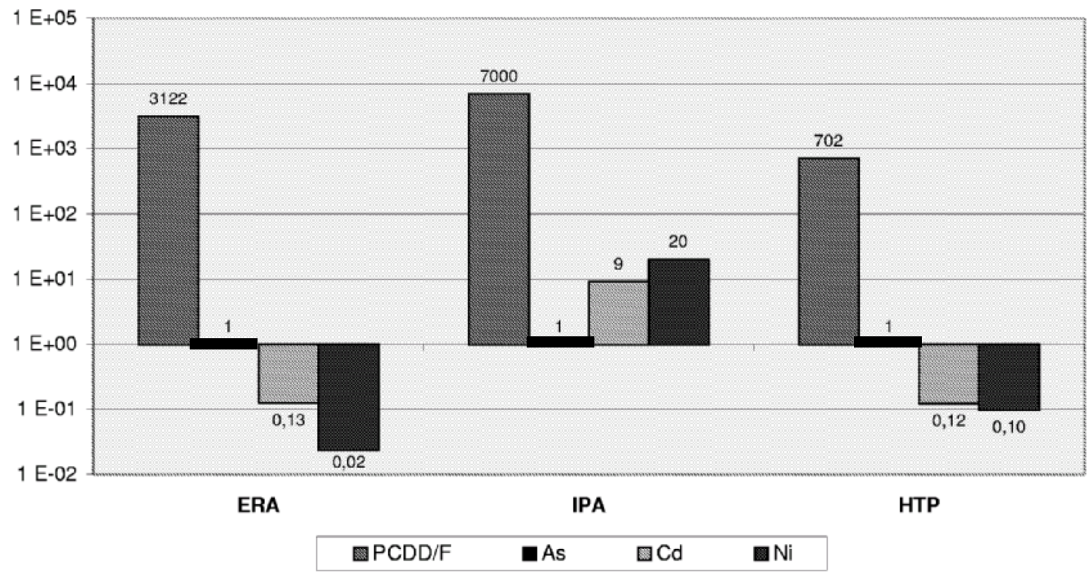


Figure 2.7: Comparison of the dose-response and exposure-response functions in the different methods in relation to arsenic (Schneider, 2001)

2.2.2 ENVIRONMENTAL COSTS

If a company or public administration has to choose between one technological solution and another, money is a very important parameter. Therefore, the techno-economic tools Cost-Benefit and –Effectiveness Analysis have been developed as decision support.

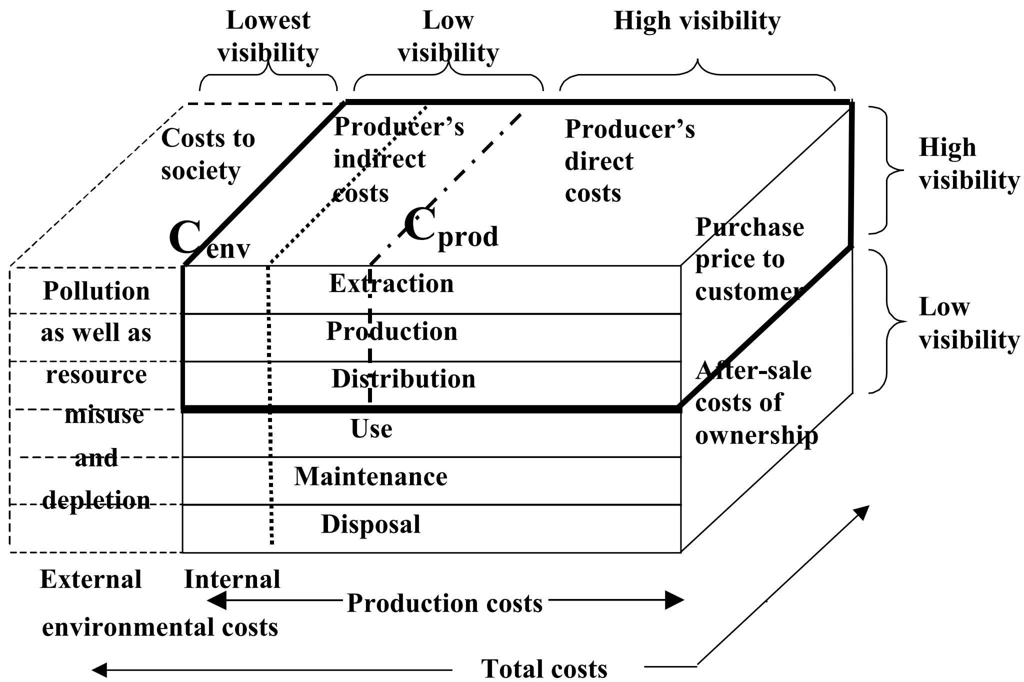
In the Cost-Effectiveness Analysis the abatement costs are compared to the positive effect that this inversion has on the environment. In principle, the alternative with the best performance with regard to its cost-effectiveness is chosen. Abatement costs are those that are necessary to install end-of-pipe pollutant removal equipments or to make the existing processes cleaner. They are internal costs for the company.

The Cost-Benefit Analysis (CBA) has been developed to support long-term decisions from a society point of view, in contrast to the company perspective. The field of application includes especially the evaluation of regulatory measures with a huge influence on the environment and the selection of general public environmental strategies. The Cost-Benefit Analysis intends to convert the cost and benefits of regulatory measures, public environmental strategies, etc. in monetary units (Nas, 1996). The basic principle behind this purpose coming from economic science is to arrange the disequilibria, caused by imperfections of the market, in the economic optimum between public and private interests. Therefore, it is necessary to quantify economically the effects of the analysed plans on the society. These effects can be environmental damages; then they refer to effects on the environment; methods for their monetisation allow to estimate external environmental costs or externalities. They are called external, since they are not considered in conventional accounting methods (Desgupta & Pearce, 1972). The CBA facilitates a way for an efficient management of the resources for the whole society. When the results indicate that, as a consequence of the project, negative effects to third parts, like atmospheric pollution or generation of dangerous wastes, dominate the public administration should make an intervention. The establishment of emission thresholds or taxes related to the activities that provoke the damages are some of the interventions the government can undertake to neutralise the negative effects. The CBA methodology consist of four phases (Nas, 1996):

- Identification of the relevant costs and benefits
- Assignation of monetary values to the costs and benefits
- Comparison of the costs and benefits in the form of monetary units generated along the lifetime of the project
- Final decision about the viability of the project and if appropriated the adaptation of necessary interventions by the public administration

Types of costs generated in the life cycle of a product

Figure 2.8 gives an overview of all the costs generated in the life cycle of a product and its visibility. The total costs are divided into two main types: production costs and environmental costs. The costs with lot of visibility are the direct ones of the producer that are included in the selling price to the client and generated in the phases from extraction to distribution. These are the conventional costs for raw materials, energy and salaries. The costs in the second half of the life cycle until the disposal are less visible; that are the costs related to the ownership after buying the product. The indirect costs of the producer do neither have lot of visibility, they consists of pollution abatement costs, actions to reduce the accident risk at the working place and other measures that are not directly necessarily to manufacture the product. The first part of the environmental costs, the internal environmental costs or abatements costs, belongs in the first half of the life cycle to the producer's indirect costs. They are internal costs from an environmental point of view because the polluter pays them. Moreover, in each phase of the product life cycle costs to society are generated in the form of pollution as well as misuse and depletion of resources. These costs are the second part of environmental costs, the external environmental costs or externalities; they have very little visibility.



C_{env} – Environmental costs C_{prod} – Production costs

Figure 2.8: Types of production and environmental costs and their visibility

Thus, two types of environmental costs can be distinguished:

- *Internal environmental costs or abatement costs* are those a company pays to reduce its environmental loads, at least, under the legal threshold, e.g. the installation and maintenance of gas filters.
- *External environmental costs or externalities* are those emissions and other environmental loads caused to society, e.g. increase of asthma cases; to obtain them the monetisation of environmental damage estimates is necessary.

The internal environmental or indirect costs do no have so much visibility as the direct costs because the companies consider them as an integrated part of the product cost and not separately. However, they can be determined by the costs that the installation and the operation of the pollution reduction technologies generate in the companies. In economical science the fact to make the companies pay the pollution reduction, i.e. the "polluter pays principle", is called internalisation of the social costs. In general, the legislation provides the framework for defining how much emissions are allowed and how much have to be internalised.

Externalities

The conversion of environmental damages in costs is called monetisation. Having in this way the external environmental costs or externalities at our disposal it is possible to internalise these costs and calculate the total cost of a product. Theoretically, this is the price a product should have to be consistent with the market. In a figurative sense, we could consider it as the amount that we have to pay to maintain the planet in equilibrium, apart from the amount we pay to the producer.

More practically it means that with the monetisation environmental damages can be introduced in the equations of economic balances and that the monetisation gives a support to solve the allocation problem of public funds for the protection of life.

However, lacking of common reference for comparison of different impact endpoints involves inevitably a value judgement. Monetisation is just one option; therefore, in the following also critical points have to be mentioned against the monetisation of environmental damages:

- On a more fundamental level there are doubts whether the monetary evaluation of human health and the environment is ethically defendable.
- The assignment of economic values to human health and the environment is not necessarily a guarantee for a sustainable development and they are considered to be insufficient for the prescription of environmental policies.

It is out of the scope of this study to take part for one or the other point of view. It is up to the decision-maker to make the choice. At continuation, the monetisation methods are shortly presented according to the state-of-the-art. Other weighting schemes exist; some examples will be presented later in this chapter.

In principle, two fundamental concepts exist in the science of environmental economics for the monetisation of environmental damages:

- In the *direct measurement of damages* the costs are directly quantified in the market, for example costs of illness (COI).
- In the case of environmental impacts that the individuals consider as damages, but which cannot be measured directly in the market, another perspective is taken. It is considered that the function of the *Willingness-To-Pay* (WTP) for the reduction of the emission is equal to the marginal damage function for the increment of emissions.

For the evaluation of externalities several methods have been proposed (Turner, 1993; Azqueta, 1994; Field, 1995; Endres & Holm-Müller, 1998). In Table 2.4 they are summarised and their advantages and disadvantages are highlighted.

Table 2.4: Comparison of methods for the monetisation of environmental damages

Method	Description	Type of cost	Advantages	Disadvantages
Prevention costs	Inversion of the companies in abatement technologies	Internal and external (market)	Inversions carried out for the environment	The majority does not correspond to the costs that cause the damages.
Mitigation costs (costs of illness and others)	Expenses of the society to avoid environmental damages and to pay for the reparation	External (market), Direct Estimation	Real expenses due to the damages	It does not contain all the known damages, especially those to human life and ecosystems.
Accounting data	Differences in the economic results of a company due to environmental damages	External (market), Direct Estimation	Real economic losses for the companies due to the damages	The economic losses are not always evident; data interpretation is necessary.
Travel costs	Economic evaluation of the behaviour of people to enjoy tourism areas	External (WTP by market)	It gives a possibility to obtain a value based on the real behaviour in the market.	No real costs are measured, only the WTP based on behaviour observation.
Hedonistic Pricing	Analysis of the divergences of prices in the market	External (WTP by market)	The divergences allow to obtain a value based on market data.	The divergences may be caused by various reasons and measure only the WTP.
Contingent Valuation Method (CVM)	Interviews with the objective to determine in monetary terms the willingness to pay or accept an environmental impact	External (WTP by interviews)	The only way to obtain costs for all type of environmental damages for which a market does not exist.	The results depend a lot on the way the questions are made; a difference is observed between WTP and the Willingness-to-Accept (WTA).

While the methods “prevention costs”, “mitigation costs” and “accounting data” are relatively easy to understand, since they are direct measurement of damages, the methods “travel costs”, “hedonic pricing” and “Contingent Valuation Method (CVM)” are more difficult to understand since they use the concept WTP.

The travel cost method is one of the first approaches that the environmental economists use to calculate the demand for recreation sites, e.g. natural parks. It consists in a method that considers the travel costs as an approximate value of the price. Although it is not observed directly that people buy unities of environmental quality, the travel costs are treated as a price people are willed to pay for the experience of environmental recreation. In this way under certain circumstances a demand function, corresponding to the WTP for being in recreation sites, can be calculated.

The hedonic pricing method uses the fact that the price of certain goods in the market depends also on the level of environmental quality, for instance the price of a terrain depends on various factors, such as size, distance to centre, etc., but also on the noise level, the air quality and other environmental characteristics. In the economic literature it was tried to establish a relation between the environmental elements of the terrain prices and those characteristics. The analysis of these divergences in the market price allows to determine the WTP.

The Contingent Valuation Method is based on the concept that if someone wants to know the WTP of the population for a certain environmental characteristic he will have to ask for it. That means the basis of this approach consists of interviews. The method is called contingent valuation because it is intended by the method to make people express how they would act if they were in a determined contingent situation.

With regard to environmental damages the most important concept is the Value of Statistical Life (VSL). The loss of a statistical life is defined as the increment of the number of deaths expressed as one over a certain number of inhabitants. This corresponds to the probability to die by a factor of 1/n, where n corresponds to a certain number of inhabitants as reference group.

The focus of scientists evaluating the statistical life can be distinguished between the WTP approach and the human capital concept, where the statistical life essentially is assessed by the not received salary. The WTP can be determined by CVM or hedonic pricing through the market.

When using VSL to evaluate the death of a person due to environmental damages, the age of the person is not taken into account. Therefore, another principle has been established called Years Of Life Lost (YOLL). It is possible to estimate YOLLs based on VSL if data on the age of the reference group affected by environmental damages is available.

In Table 2.5 European results are presented for the estimation of the VSL. The lowest values are obtained by market-based studies, while the higher values stem from CVM works. These studies are difficult since a lot of factors have to be taken into account. Hence, they have an important inherent uncertainty. More recent studies from the US show a range of the VSL between € 2,3 and 12,4 million (US-EPA, 1999).

Table 2.5 Overview of some European results for VSL estimations, values reported by Ives et al. (1993)*

Method	Study	Year	Million Euro (1990)
Salaries	Melinek	1974	0,7
Salaries	Veljanovski	1978	7,0-9,8
Salaries	Needleman	1976	0,3
Salaries	Marin et al	1982	3,1-3,5
CVM	Melinek	1973	0,4
CVM	Jones-Lee	1976	12,9-16
CVM	Maclean	1979	4,3
CVM	Frankel	1979	4,3-17,5
CVM	Jones-Lee et al	1985	1,1-4,8
CVM	Persson	1989	2,2-2,7
CVM	Maier	1989	2,7
Market	Melinek	1974	0,3-0,7
Market	Ghosh et al	1975	0,7
Market	Jones-Lee	1979	0,8-9,2
Market	Blomquist	1979	0,8-2,9

* Studies have been carried out primarily in UK.

Another important point for the monetisation of environmental damages is the discount rate. Discounting is the practice to give a lower numerical value to the benefits in the future than to those from present. This fact has a lot of consequences when applied to the monetisation of environmental damages because these often occur in the near or even far future.

The damage evaluated today X_0 that will occur in T years is quantified by expression (2.6). That means that for example at a 10 % discount rate 100 € today is comparable to 121 € in two years time. Different stakeholders have broadly discussed the question of the most adequate discount rate.

$$X_t = X_0 / (1+r)^T \qquad (2.6)$$

where r = discount rate

The accurate economic evaluation of environmental damages depends not only on the monetisation method chosen and the discount rate used, but also on the question in how far once determined economic damage values can be transferred from one place to another. It is a difficult aspect to decide which modifications should be done for example in order to use results of US studies in the EU.

In this study the monetisation is done by the following methods for economic valuation of damages used by the European Commission (1995):

- The Direct Estimation of damage costs is the most evident evaluation method. Here the external damage costs that are measurable in the market are taken into consideration. It facilitates the valuation of an important part of the impacts. It allows to obtain an under borderline of the total environmental cost. But there exist other types of costs.

- The WTP method tries to answer the question how much are we prepared to pay to reduce the emissions. The WTP is considered to be an adequate measure of preference. For example, we give daily a price to our live by certain decisions as buying a car with or without an airbag.
- Discounting corresponds to weighting on the level of inter-generational equity. That means that the interests of future generations have to be taken into account. However, since in practice it is not possible to measure the values of future generations, the discount rates applied can be understood as the true social discount rate minus the rate of appreciation of the value itself. This consideration justifies the use of a discount rate below rates observed in capital markets.

With these methods, the types of monetary values obtained are for instance:

- Mediation Costs, i.e. costs of illness
- Productivity Loss/ Company's Accounting Data, i.e. wage loss
- Economic valorisation of a VSL on the basis of expression (2.7)

$$VSL = \Sigma\, WTP / \Delta p \qquad (2.7)$$

where VSL = value of statistical life
WTP = Willingness-to-pay
Δp = change in probability of death

The conventional approach for valuing mortality is based on the estimation of the WTP for a change in the risk of death (Δp), allowing calculation of VSL by dividing the WTP by the change in risk. A meta-analysis of valuation studies from Europe and North America undertaken in ExternE suggests a mean VSL of 3.1 Mio Euro at a 3% discount rate, derived from accident studies according to the ExternE project (Mayerhofer et al., 1998).

As most of the valuation studies are based on a context in which the individuals involved are exposed to an accidental risk leading to a loss of life expectancy of about 30 to 40 years, the transfer of results to the air pollution context is problematic. Increased mortality from air pollution is mainly expected to affect old people in poor health, leading to a loss of life expectancy between some few days (harvesting effect due to a high pollution episode –acute mortality) and some few years (resulting from long-term exposure to increased levels of air pollution – chronic mortality). An alternative valuation approach that seems to better reflect the context of mortality related to air pollution is to value a change in risk in terms of the willingness to pay for life years and to derive a Value of a Life Year Lost (VLYL). As there is only little empirical evidence on the WTP for lost life years, the ExternE study has developed a theoretical framework to calculate the Value of a Life Year Lost from the Value of Statistical Life. Assuming for simplicity reasons that the value of a life year is independent of age, a relationship between the VSL and the VLYL is established (Krewitt et al., 1999).

Rabl et al. (1998) indicate that from this assumption results that a VLYL corresponds to approximately 0.1 Million Euro. In principle, throughout this study a discount rate of 3 % is applied. Based on the uncertainty analysis in chapter 4, in this study we will try to compare the uncertainties due to this valuation step with other sources of uncertainties in environmental damage estimations in industrial process chains.

2.2.3 WEIGHTING SCHEMES FOR THE EVALUATION OF ENVIRONMENTAL DAMAGES

The most fundamental problem in damage estimations is the fact that the final outcome often refers to value choices and thus the weighting scheme of the decision-maker. A single truth simply does not exist as long as value choices are necessary.

Cultural Theory

To deal with this, Hofstetter (1998) has analysed the problem of modelling subjectivity thoroughly and he proposed to use the socio-cultural viability theory (Thompson et al. 1990), shortly called Cultural Theory, to distinguish five extreme value systems that are illustrated in Figure 2.9.

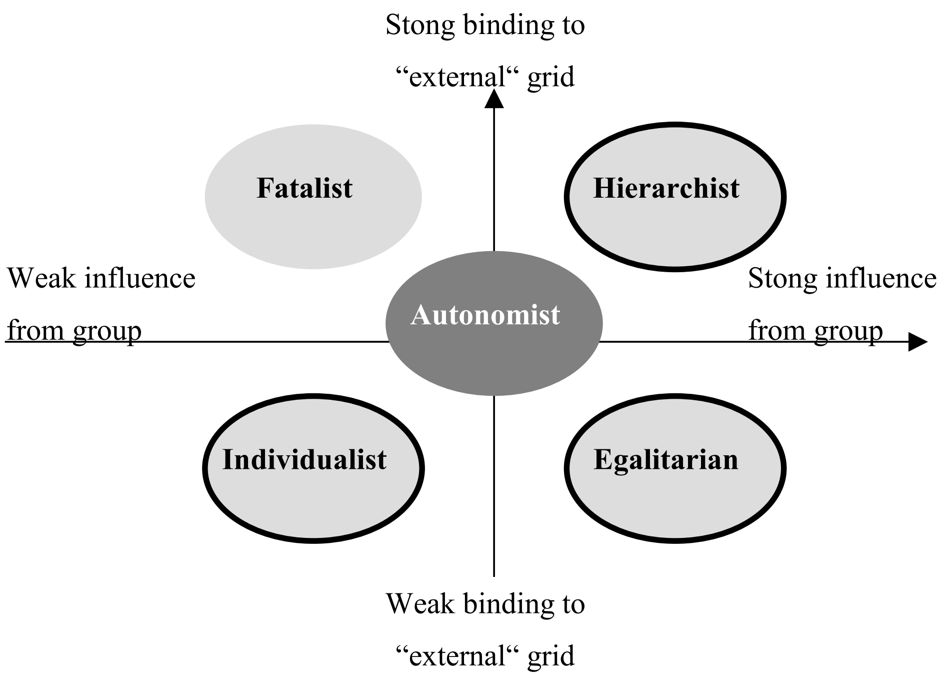


Figure 2.9: The grid-group dependency of the five extreme archetypes distinguished in Cultural Theory (Goedkoop & Spriensma, 1999). The Autonomist has no fixed position in this figure, because the Autonomist does not have social relations and should be seen as floating over the archetypes. Only the hierarchist, egalitarian and individualist perspectives are relevant for decision-making and will be used in the default scenarios which are proposed as extreme cases if no other scenarios based on more specific information are available (Weidema et al., 2001).

Thompson et al. (1990) derive 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 positions 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.

According to Hofstetter (1998) several authors have used these value systems 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 that there are only five types of people. Almost nobody conforms to the viewpoints of a single group in a consistent way. People can switch between different attitudes, dependent on their context. The archetypes are extreme positions not likely to be held by anyone, but they are useful because they delimit the area of possible positions.

The most important characteristics of the five extreme archetypes can be summarised in the following way:

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, 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 extreme archetypes have distinctly different preferences referring to modelling choices that have to be made. Hence, they are relevant for decision-making. The two last 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 socio-cultural viability theory is that we can predict a wide range of basic attitudes and assumptions for the three remaining extreme archetypes: the hierarchist, the individualist and the egalitarian. The Table 2.6 specifies some of the many different characteristics per archetype. Therefore, the LCIA method Eco-indicator 99 uses these three perspectives to facilitate the analysis of the relative contribution of the different damage category indicators to one endpoint.

Table 2.6: Attitudes corresponding to the three cultural perspectives used in the Eco-indicator 99 (Hofstetter, 1998; Goedkoop & Spriensma, 1999)

Archetype	**Time perspective**	**Manageability**	**Required level of evidence**
Hierarchist	Balance between short and long term	Proper policy can avoid many problems	Consideration is based on consensus.
Individualist	Short time	Technology can avoid many problems	Only proven effects are accepted.
Egalitarian	Very long term	Problems can lead to catastrophe	All possible effects are taken into account.

Monetisation

In somehow the discount rates used in the above-mentioned ExternE project (EC, 1995) can be interpreted in the following way: 0 % discount rate corresponds to the egalitarian perspective, a damage in the far future is as important as a damage nowadays. The 3 % discount rate can be interpreted as the hierarchist perspective, considering the opinion of an economic scientist and recommended by the ExternE project team (EC, 1995). Finally a discount rate of 10 % describes the individualist point of view, a person with this perspective prefers to have what he can in the present; only the very near future is of interest when considering possible damages (Weidema et al., 2001). Another way would be to use discount rates differentiated according to the environmental relevance of the different impacts.

The DALY concept

In order to aggregate different health effects into a single indicator, instead of economic valuation also the DALY concept can be implemented. In the following these concepts as applied to LCIA are explained (Hofstetter, 1998).

DALY stands for Disability Adjusted Life Years (Murray & Lopez, 1996). This indicator aggregates health effects leading to death or illness. Health effects leading to death are described using the YOLL indicator. This indicator includes all fatal health effects such as cancer or death due to respiratory health effects. Respiratory health effects are further divided into acute and chronic death. Acute death means the immediate occurrence of death due to an overdose of a certain pollutant. Chronic death accounts for health effects that lead to a shorter life expectancy. In order to derive the number of life years lost due to a fatal disease, statistics are used, especially from the World Health Organisation (WHO). In these statistics it can be found at what age and with which probability death occurs due to a certain cancer type or respiratory health effect. Combining these statistics and the dose- response and exposure-response functions, it can be calculated how many years of life are lost due to the concentration increase of a certain pollutant.

The DALY concept does not only include the mortality effects but also morbidity. Morbidity describes those health effects that do not lead to immediate death or to a shorter period of life, but which account for a decrease of life quality and for pain and suffering. Cough, asthma or hospitalisations due to different pollutants refer to this indicator. The morbidity health effects are also expressed in years, the so-called Years of Life Disabled (YLD). Value choices have to be made to weight the pain or the suffering during a certain period of time against premature death. Depending on the severity of the illness, suffering and pain the weighting factor for morbidity is between 0 and 1. A weighting factor of 0.5 means that one year of suffering is supposed to be equally severe as half a year of premature death. The DALY indicator is then the result out of the addition of both indicators, with DW being the relative Disability Weight and L representing the duration of the disability, see expression (2.8).

$$DALY = YOLL + YLD = YOLL + DW * L \tag{2.8}$$

Often a pollutant contributes to more than one health effect and a certain health effect can lead to both morbidity and mortality. Cancer, for instance, often leads to a period of suffering and pain before death occurs. Therefore cancer contributes to both YLD and YOLL.

The value of YOLL or YLD does not only depend on the pollutant and the type of disease. As value-choices are necessary for weighting, YLD and YOLL strongly depend on the attitude of the person carrying out the weighting step. Moreover, a year of life lost at the age of 20 or at the age of 60 is not equally appreciated in every socio-economic perspective according to the cultural theory. For instance, one cultural theory discounts years of life lost in the way discounting is done in finances. Therefore, a year of life lost in the future is worth less than a year of life lost today. And another cultural perspective judges every year of life lost equally important independently of the age when it occurs. If one looks at the unit YOLLs, YLDs and DALYs, it can be said that the overall damage for cancer is determined by mortality effects (YOLL) while morbidity effects (YLD) can be neglected. For respiratory health effects however, morbidity plays an important role.

Comparison of monetisation with the DALY concept

If one compares one unit of external costs according to the ExternE project (EC, 1995), as used in this study, with one unit of YOLL and YLD, then it can be seen how different diseases are weighted according to the weighting scheme: physical impact (YOLL and YLD) or economic impact (costs). From the point of view of the economic weighting scheme a high ratio costs/ YOLL or costs/ YLD means that the cost-weighting scheme considers the health effect to be relatively more economically harmful than a health effect with a low respective ratio. From the point of view of the YOLL and YLD weighting scheme it can be said that a high ratio means that the physical impact (expressed in YOLL and YLD) is not very important while for a low ratio the physical impact becomes important.

This can be explained using the following example: if one considers the comparison of costs and YLD for morbidity, it can be seen that the economic value per YLD ranges from 9,600 €/ a (chronic cough) up to 502,326 €/a (respiratory hospital admission). Apparently, the economic approach does not consider chronic cough to be a severe disease in the economic sense while the respiratory hospital admission is assumed to be cost intensive. The underlying thinking is that the COI for chronic cough is less cost intensive because the treatment usually can be done at home and only medicine has to be considered economically. Respiratory hospital admission is considered to be much more cost intensive, what is due to the fact that besides medicine cost-intensive care in hospital is necessary. Moreover, while people suffering from chronic cough still can work, people admitted to hospital lead to a loss of manpower and therefore of money. From the point of view of the YLD approach it can be said that chronic cough is considered a severe disease. While an admission to hospital is over after a certain period of time, chronic cough leads to a life long suffering and pain. This example shows that diseases are valuated differently depending on the weighting scheme. In this example the valuation is opposite for the two weighting schemes. This shows that the choice of the weighting scheme is of high importance for the outcome in environmental damage estimations.

Looking at mortality it can be seen that the ratio for chronic YOLL is much lower than for acute YOLL. The underlying reasoning is that premature death in the future is considered to be less economically severe than acute death (at a discounting rate of 3%).

If one looks at cancer, the ratios costs/ YOLL and costs/ YLD are the same because the economic values cannot be distributed between the fatal (YOLL) and the non-fatal (YLD) effects of cancer. Nose cancer is then considered to be economically the most severe type of cancer.

For all types of diseases it can be said that the ratio costs/ YOLL and costs/ YLD is greater for the hierachist perspective with age-weighting than for the egalitarian perspective without age-weighing. They depend on the time of occurrence of the disease, which determines the difference between the unit YOLL and YLD values. It can be seen that the unit values of YLD and YOLL for cancer are always lower when age-weighting is applied. This is owing to the fact that cancer usually happens at an advanced age.

Selection of dose-response and exposure-response functions

The assessment of the risk, damage or impact potential of a pollutant represents the result of a partly objective and partly subjective evaluation process due to the subjective element in the selection of the dose-response and exposure-response functions. Therefore, another differentiation according to the Cultural Theory is feasible with respect to dose-response and exposure-response functions. There are cultural theories that doubt certain correlations to exist between pollutants and health effects. For example, some cultural perspectives dispute the correlation between sulphate and ischaemic heart disease, see Hofstetter (1998) for further discussion of this topic.

2.3 CHAIN-ORIENTATED ENVIRONMENTAL IMPACT ASSESSMENT

Life Cycle Assessment is the only standardised tool for chain-orientated environmental impact assessment and will be further explained in this section. It has been developed for the environmental evaluation of products.

2.3.1 LIFE CYCLE ASSESSMENT (LCA)

Life Cycle Assessment (LCA) is a tool to evaluate the environmental performance of products (SETAC, 1993; UNEP, 1996). LCA focuses on the entire life cycle of a product: from the extraction of resources and processing of raw material, through the manufacture, distribution, and use of the product, to the final processing of the disposed product (see Figure 2.1). Through all these stages, extractions and consumptions or resources (including energy) and releases to air, water and soil are identified and quantified. Subsequently, the potential contribution of these resource extractions and consumptions, and environmental releases to several important types of environmental impact are assessed and evaluated (Curran, 1996; EEA, 1998).

The technical framework for the Life Cycle Assessment methodology has been standardised by the International Standards Organization (ISO). According to ISO 14040 (1997) LCA consists of four phases, as presented in Figure 2.10:

1. Goal and Scope Definition
2. Inventory Analysis
3. Impact Assessment
4. Interpretation

These phases are not simply followed in a single sequence. It is an iterative process, in which subsequent iterations (rounds) can achieve increasing levels of detail (from screening LCA to full LCA), or lead to changes in the first phase prompted by the results of the last phase. The Life Cycle Assessment has proven to be a valuable tool to document the environmental considerations of product systems that need to be part of decision-making towards sustainability. Products mean here goods and services.

ISO 14040 (1997) provides the general framework of LCA. ISO 14041 (1998) provides guidance for determining the goal and scope of an LCA study, and for conducting a life cycle inventory. ISO 14042 (2000) is about the life cycle impact assessment phase, and ISO 14043 (2000) provides guidance for the interpretation of results from an LCA study. Moreover, technical guidelines exist that illustrate how to apply the standards.

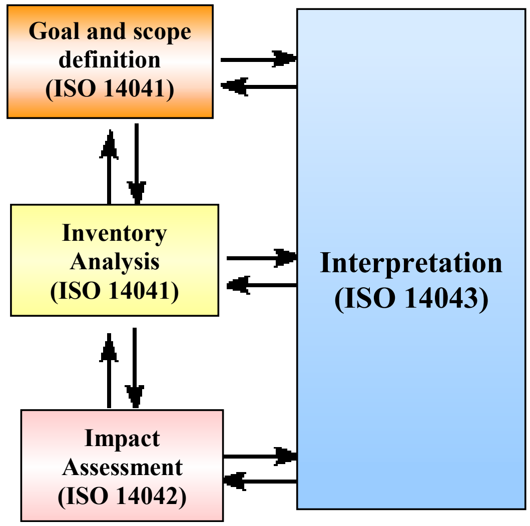


Figure 2.10: The phases of LCA according to ISO 14040 (1997)

The **Goal and Scope Definition** phase is designed to obtain the required specifications for the LCA study: What questions do we want to answer and who is the intended audience? The following steps must be taken:

1. *Defining the purpose of the LCA study*, ending with the definition of the functional unit, which is the quantitative reference for the study.
2. *Defining the scope of the study,* which includes the drawing up of a flowchart of the unit processes that constitute the product system under study, taking into account a first estimation of their inputs from and outputs to the environment (the elementary flows or burdens to the environment).
3. *Defining the data required*, which includes a specification of the data required both for the Inventory Analysis and for the subsequent Impact Assessment phase.

The **Inventory Analysis** collects all data of the unit processes of the product system and relates them to the functional unit of the study. The following steps must be taken:

1. *Data collection*, which includes the specification of all input and output flows of the processes of the product system, both product flows (i.e. flows to other unit processes) and elementary flows (from and to the environment).
2. *Normalisation* to the functional unit, which means that all data collected are quantitatively related to one quantitative output of the product system under study, most typically 1 kg of material is chosen, but often other units like a car or 1 km of mobility are preferable.
3. *Allocation*, which means the distribution of the emissions and resource extractions of a given process over the different functions which such a process, e.g. petroleum refining, may provide.
4. *Data evaluation*, which involves a quality assessment of the data, e.g. by performing sensitivity analysis.

The result of the Inventory Analysis, consisting of the elementary flows related to the functional unit, is often called the "Life Cycle Inventory table".

The **Impact Assessment** phase aims to make the results from the Inventory Analysis more understandable and more manageable in relation to human health, the availability of resources, and the natural environment. To accomplish this, the inventory table will be converted into a smaller number of indicators. The mandatory steps to be taken are:

1. *Selection and definition of impact categories*, which are classes of a selected number of environmental impacts such as global warming or acidification.
2. *Classification*, comprising the assignment of the results from the Inventory Analysis to the relevant impact categories.
3. *Characterisation*, which means the aggregation of the inventory results in terms of adequate factors, so-called characterisation factors, of different types of substances in the impact categories, therefore a common unit is to be defined for each category, the results of the characterisation step are entitled the environmental profile of the product system.

The **Interpretation** phase aims to evaluate the results from either Inventory Analysis or Impact Assessment and to compare them with the goal of the study defined in the first phase. The following steps can be distinguished:

1. *Identification* of the most important results of the Inventory Analysis and of the Impact Assessment.
2. *Evaluation* of the study's outcomes, consisting of a number of the following routines: completeness check, sensitivity analysis, uncertainty analysis and consistency check.
3. *Conclusions, recommendations and reporting*, including a definition of the final outcome; a comparison with the original goal of the study; the drawing up of recommendations; procedures for a critical review, and the final reporting of the results.

The results of the Interpretation may lead to a new iteration round of the study, including a possible adjustment of the original goal.

2.3.2 LIFE CYCLE INVENTORY (LCI) ANALYSIS

According to Castells et al. (1995), in the LCI Analysis the assignment of the environmental loads to the different flows of a process and the realisation of the corresponding balance is carried out by a methodology based on a **vector** that contains all the information about the possible pollution and resource depletion. Every product or flow has a vector associated with the information about all the environmental loads generated along its entire life cycle. This **eco-vector *v*** is a multidimensional vector in which each dimension corresponds to a specific environmental load.

Each mass flow in the process [kg/s] has got an associated eco-vector ***v*** whose elements are expressed in mass unit as kg of pollutant per kg of product or other units like energy [kJ/ kg] or Environmental Load (EL) per mass unit [EL/ kg] that cannot be expressed in mass like radiation or acoustic intensity [W/m^2]. In each case the elements have to be expressed in units that can be accumulated and are adequate to make balances. Expression (2.9) shows a mass eco-vector, $\boldsymbol{v_m}$ in which the environmental loads are grouped by types of environmental impact. For instance in Boustead (1993) and Frischknecht et al. (1996) examples of lists with environmental loads can be found that constitutes elements of the eco-vectors.

$$v_m = \begin{bmatrix} \text{(kg/kg) or (EL/kg)} \\ \text{Renewable raw materials} \\ \text{Not renewable raw materials} \\ \text{Emissions to air} \\ \text{Emissions to water} \\ \text{Solid wastes} \\ \text{Radiation} \\ \text{Other environmental impacts} \end{bmatrix} \tag{2.9}$$

The product of the mass flow M [kg/s] of a process and the corresponding vector $\boldsymbol{v_m}$, give the quantity of pollutants $\boldsymbol{P}$ [kg/s] or [EL/s] generated until the life cycle phase of this process, see expression (2.10). The quantity [EL/s] indicates the environmental load per unit of time not measurable in mass unities as for example the radiation.

$$\mathrm{M}\, v_m = P \tag{2.10}$$

In parallel, for the energy flows an energy eco-vector is defined $\boldsymbol{v_e}$ whose elements are expressed in mass, e.g. in kg of pollutant per kJ or in the units like in the case of the mass eco-vector. The rows of $\boldsymbol{v_e}$ have analogue elements than those of the mass eco-vector, see expression (2.11).

$$v_e = \begin{bmatrix} \text{(kg/kJ) or (EL/kJ)} \\ \text{Renewable raw materials} \\ \text{Not renewable raw materials} \\ \text{Emissions to air} \\ \text{Emissions to water} \\ \text{Solid wastes} \\ \text{Radiation} \\ \text{Other environmental impacts} \end{bmatrix} \tag{2.11}$$

From the product of the energy flow E [kW] and the corresponding vector $\boldsymbol{v_e}$ the flow of the pollutants generated in the energy production is obtained; that is the vector $\boldsymbol{P}$ [kg/s] or [EL/s], see expression (2.12).

$$\mathrm{E}\, \boldsymbol{v_e} = \boldsymbol{P} \tag{2.12}$$

The expressions (2.10) and (2.12) indicate that the environmental loads of the mass and energy flow can be handled together, because the product of the flow and the corresponding vector is always the pollutants flow ***P*** [kg/s] or [EL/s].

Each input to the system has got an associated eco-vector and its content has to be distributed to the output of the system. The balance of each of the elements of the eco-vector has to be closed so that the total in the output of the process is equal to the pollutant quantity that entered the process plus the quantity generated in the process itself.

To enable this balance the output is divided in products and wastes. In the present methodology the waste flows have eco-vectors with negative elements, corresponding to the pollutants they contain. The environmental load of the input and of the waste flows have to be distributed among the products of the process.

In this way, the LCI or the balance of environmental loads of a process is carried out in a similar way to a material balance. A whole product system is divided into units or subsystems and in each one the established system of equations is solved so that eco-vectors for the final outputs or intermediate products are obtained. The solution of the whole system allows to know in detail the origin of the pollution and it can be assigned to each product of a plant.

In the case of discontinuous processes the balances are carried out in a similar way, only changing the basis of the computation. For example, instead of considering the pollutant mass per second, the mass per kg of product is taken.

As an illustration of this algorithm the generic system represented in Figure 2.11 is taken with n inputs of raw materials and energy and n outputs of products and wastes. The resulting balance of the global environmental load is given by expression (2.13).

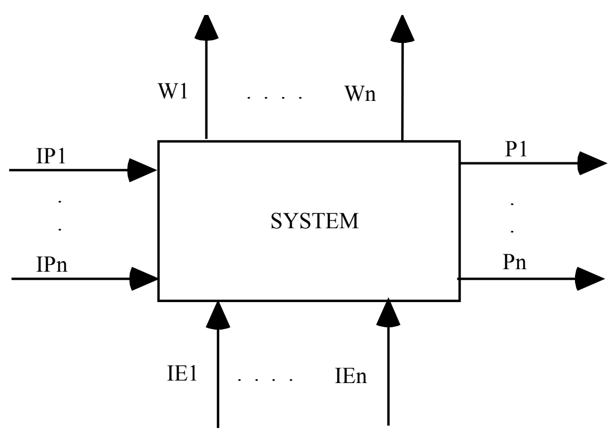


Figure 2.11: Generic product system (E- Energy, I – Intermediate, P- Product, W- Waste)

$$\sum_{i=1}^{n} IPi\, v_m\, IPi + \sum_{i=1}^{n} IEi\, v_e\, IEi - \sum_{i=1}^{n} Wi\, v_e\, Wi = \sum_{i=1}^{n} Pi\, v_e\, Pi \qquad (2.13)$$

where: IPi = mass inputs
IEi = energy inputs
Pi = outputs (products and subproducts)
Wi = wastes
Vm,e = mass and energy eco-vectors of the flows

One part of the uncertainty analysis in chapter 4, will focus on Life Cycle Inventory analysis to make the sources of uncertainty within this approach more transparent and to compare it to other sources of uncertainties in environmental damage estimations in industrial process chains.

2.3.3 LCA SOFTWARE

Based on the LCA methodology described above, in the last decade numerous LCA software tools have been developed. Examples of generic tools are TEAM (Ecobilan group), SimaPro (Pré Consultants), GaBi (IKP), LCAiT (Chalmers) and Umberto (IFEU). WISARD (Ecobilan group) is a specialised LCA software tool that allows the comparison of different waste management scenarios.

Moreover, a countless number of Excel-based tools exist. In this study, one of these Excel-based tools developed within the Environmental Management and Analysis (AGA) Group of the Universitat Rovira i Virgili was applied. The Excel based tool has been developed according to the above-explained algorithm proposed by Castells et al. (1994), using mainly data from Frischknecht et al. (1996).

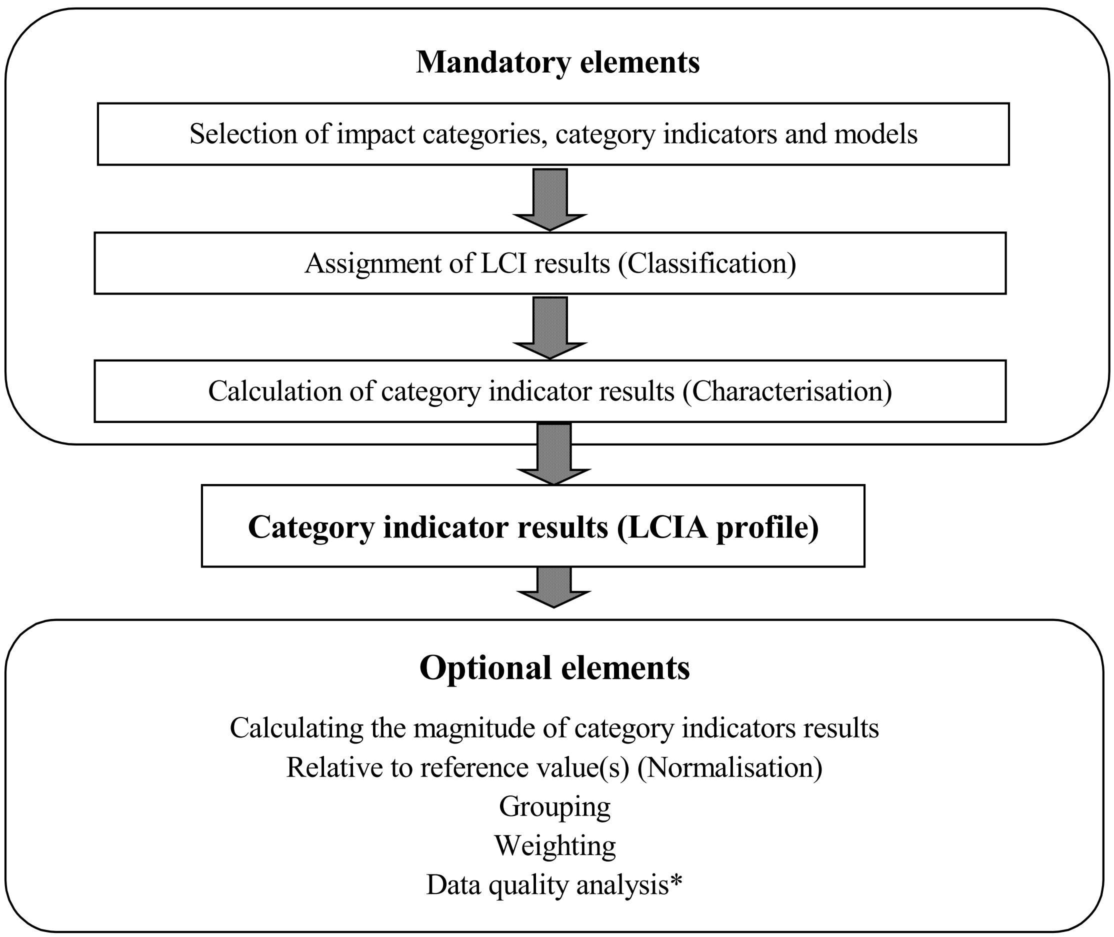
Additionally, in this study the more complex commercial software TEAM™ was used. The acronym TEAM stands for Tools for Environmental Analysis and Management (Ecobilan, 1998). The TEAM software has been delivered with database blocks called DEAM (Data for Environmental Analysis and Management) that includes several publicly available LCI data sets.

2.3.4 LIFE CYCLE IMPACT ASSESSMENT (LCIA) FRAMEWORK

Life Cycle Impact Assessment (LCIA) is the third phase of Life Cycle Assessment described in ISO 14042 (2000) and further outlined in ISO/TR 14047 (2002). The purpose of LCIA is to assess a product system's Life Cycle Inventory to better understand their environmental significance. It also provides information for the interpretation phase.

The LCIA phase, in conjunction with other phases, provides a system wide perspective of environmental and resource issues for product systems. It assigns Life Cycle Inventory results to impact categories. For each impact category the indicators are selected and the category indicator results are calculated. The collection of these results, the LCIA profile, provides information on the environmental relevance of the resource use and emissions associated with the product system. In the same way as LCA, Life Cycle Impact Assessment is a relative approach based on a functional unit.

The general framework of the LCIA phase is composed of several mandatory elements that convert LCI results into indicator results. In addition, there are optional elements for normalisation, grouping or weighting of the indicator results and data quality analysis techniques. The LCIA phase is only one part of a total LCA study and shall be coordinated with other phases of LCA. An overview of the mandatory and optional elements in LCIA is given in Figure 2.12.



* Mandatory in comparative assertions

Figure 2.12: Mandatory and optional elements of LCIA according to ISO 14042 (2000)

Separation of the LCIA phase into different elements is necessary for several reasons:

1. Each LCIA element is distinct and can be clearly defined.
2. The LCA study goal and scope definition phase can consider each element.

3. A quality assessment of the LCIA methods, assumptions and other decisions can be conducted for each LCIA element.
4. LCIA procedures, assumptions, and other operations within each element may be transparent for critical review and reporting.
5. Values and subjectivity – value choices – within each element have to be made transparent for critical review and reporting, if applied.

The mandatory LCIA elements are listed below:

- Selection of impact categories, category indicators, and models.
- Assignment of LCI results (Classification) to the impact category. That is, the data from the inventory table are grouped together into a number of impact categories.
- Calculation of category indicator results (Characterisation). That is, the magnitude of the impacts on the ecological health, human health, or resource depletion for each of the stressor categories are analysed and estimated.

The indicator results for different impact categories together represent the LCIA profile for the product system.

There are optional elements and information that can be used depending on the goal and scope of the LCA study:

- Calculating the magnitude of category indicators results relative to reference value(s) (Normalisation). That is, all impact scores - contribution of a product system to one impact category - are related to a reference situation.
- Grouping; sorting and possibly ranking of the indicators.
- Weighting (across impact categories). That is a quantitative comparison of the seriousness of the different resource consumptions or impact potentials of the product, aiming at covering and possibly aggregating indicators results across impact categories.
- Data quality analysis. That is to better understand the reliability of the LCIA results.

The use of models is necessary to derive the characterisation factors. The applicability of the characterisation factors depends on the accuracy, validity and characteristics of the models used. For most LCA studies no models are needed because existing impact categories, indicators and characterisation factors can be selected.

Models reflect the cause-effect chain (environmental mechanism) by describing the relationship between the LCI results, indicators and if possible category endpoint(s). For each impact category, the following procedure is proposed in ISO 14042 (2000):

- Identification of the category endpoint(s).
- Definition of the indicator for given category endpoint(s).
- Identification of appropriate LCI results that can be assigned to the impact category, taking into account the chosen indicator and identified category endpoint(s).
- Identification of the model and the characterisation factors.

This procedure facilitates an adequate inventory analysis and the identification of the scientific and technical validity, assumptions, value choices and the degree of accuracy of the model. The resulting indicators may vary in precision among impact categories due to the differences between the model and the corresponding actual environmental mechanism. The use of simplifying assumptions and available scientific knowledge influences the accuracy of the indicators.

Figure 2.13 illustrates the relationship between the results of the life cycle inventory analysis, indicators and category endpoint(s) for one impact category for the example of acidification. It shows clearly were a model is needed.

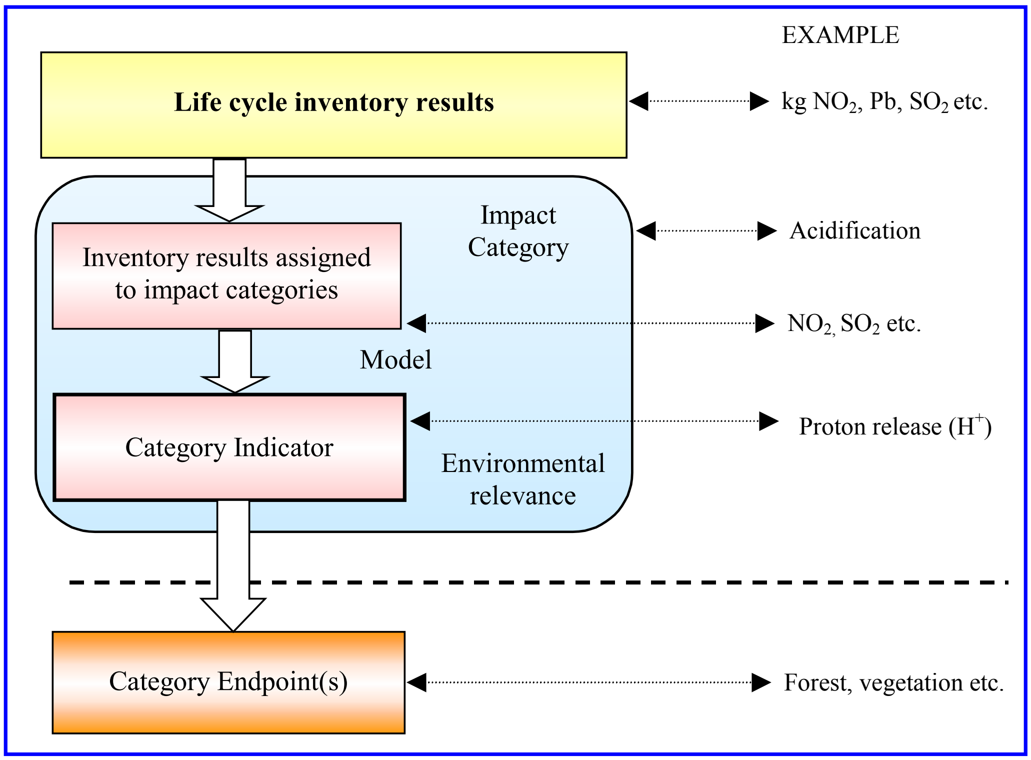


Figure 2.13: The concept of indicators (ISO 14042, 2000)

In the first report of the Second SETAC Working Group on Life Cycle Impact Assessment (Udo de Haes et al., 1999), an Area of Protection (AoP) is defined as a class of category endpoints. In ISO 14042 three of such classes are mentioned, be it in a rather implicit way: human health, natural environment and natural resources. Another term used is the expressive term "safeguard subject", introduced by Steen & Ryding (1992). It is important to note that these two terms exactly convey the same message: they relate to the category endpoints as physical elements, not to the societal values behind. So, following this terminology, the human right to life, or economic welfare cannot be an AoP or a safeguard subject; and neither can be respect for nature or cultural values.

However, the concept of AoPs enables a clear link with the societal values that are the basis for the protection of the endpoints concerned. Table 2.7 gives an overview of the AoPs with underlying societal values, as presented by Udo de Haes & Lindeijer (2001). Since the AoPs are the basis for the determination of relevant endpoints, their definition implies value choices. Thus, there is not one scientifically correct way to define a set of AoPs.

Table 2.7: Assignation of societal values to AoPs (Udo de Haes & Lindeijer, 2001)

Societal values	Human/ man-made	Natural
Intrinsic values	• Human Health • Man-made Environment (landscapes, monuments, works of art)	• Natural Environment (Biodiversity and Natural Landscapes)
Functional values	• Man-made Environment (materials, buildings, crops, livestock)	• Natural Environment (Natural Resources) • Natural Environment (Life Support Functions)

Udo de Haes & Lindeijer (2001) propose to differentiate within the AoP Natural Environment between the sub-AoPs Life Support Functions, Natural Resources and Biodiversity and Natural Landscapes. Life Support Functions concern the major regulating functions of the natural environment, which enable life on earth (both human and non-human). These particularly include the regulation of the earth climate, hydrological cycles, soil fertility and the bio-geo-chemical cycles. Like Man-made Environment (materials, buildings, crops, livestock) and Natural Resources, the Life Support Functions are of functional value for society. From a value perspective, these two are therefore of a fundamentally other nature than the AoPs with intrinsic value to society, such as in particular those connected with human health, with biodiversity and natural langscapes and with works of art, monuments and man-made landscapes. An overview of the classification of AoPs according to societal values is presented in Figure 2.14.

Normally damages to elements within the economy that do not involve environmental processes are excluded from LCIA. In fact, these types of impact are part of the product system itself. A product system therefore not only fulfils a function, but also can lead to internal damage within the product system itself without any involvement of processes in the environment. An example concerns material damage caused by car accidents. In principle, LCA can include also the analysis of these types of impact, but in general these are considered to be additional to the environmental impacts that are part of the scope of an environmental management tool.

An important number of LCIA methods have been built up. In this study they are distinguished in impact potentials, single index approaches and damage-orientated methods.

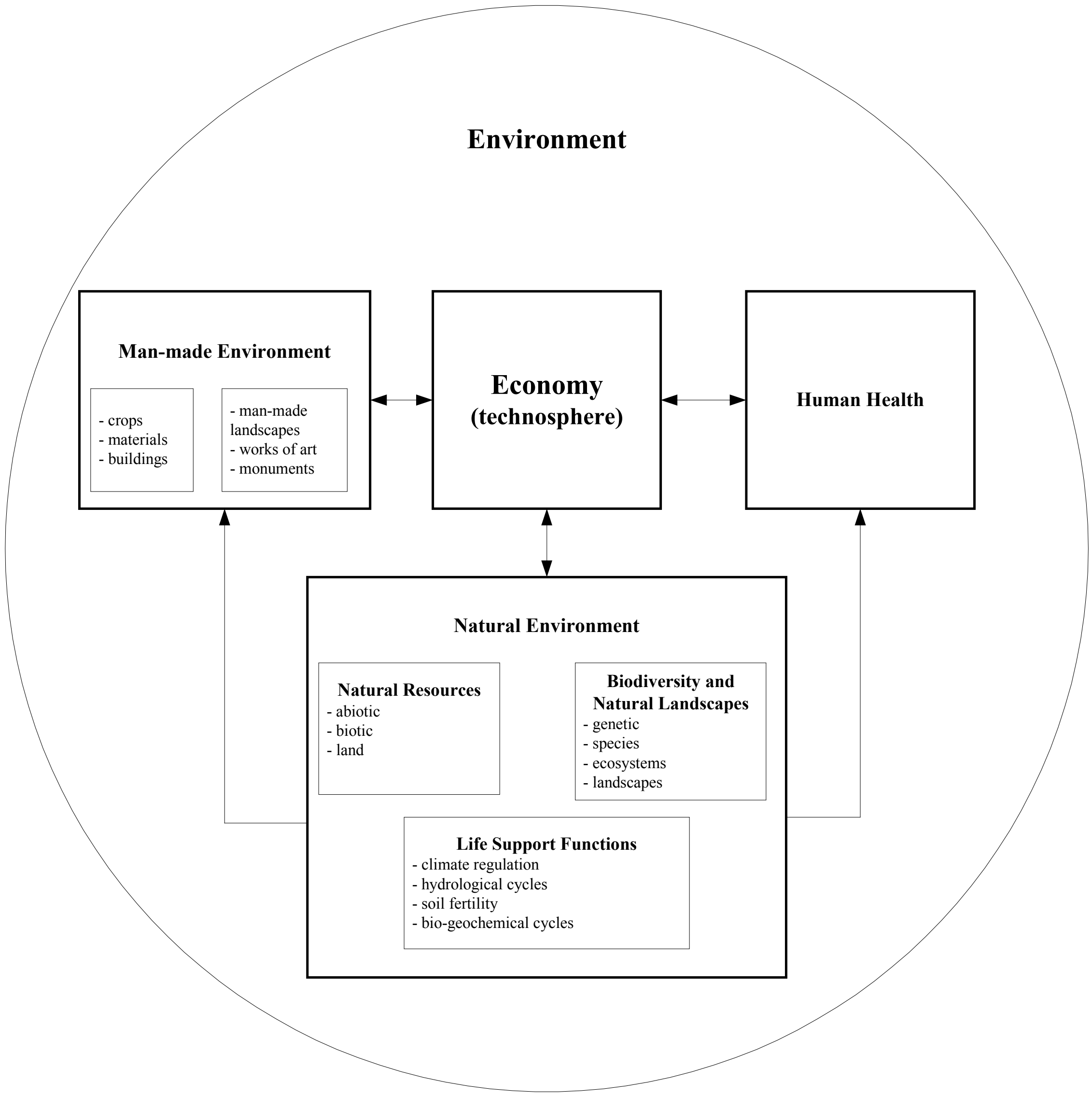


Figure 2.14: Classification of AoPs according to societal values. Arrows pointing both ways express interactions between economy and AoPs. Other arrows indicate main relationships between AoPs. (Udo de Haes & Lindeijer, 2001)

2.3.5 IMPACT POTENTIALS

In this section, a brief introduction to impact potentials is provided to facilitate the understanding of the conventional LCIA framework, by the means of the examples Global Warming Potential (GWP) and the Human Toxicity Potential (HTP), which are applied in the case study.

Global Warming Potential and Human Toxicity Potential

Most of the radian energy which the earth receive from the sun in the form of short-wave radiation is reflected directly or re-emitted from the atmosphere, or the surface of the earth, as longer wave infrared (IR) radiation. This natural greenhouse effect is increased by man-made emissions of substances or particles that can influence the earth's radiation balance, causing a raise in temperature of the planet.

Lot of the substances emitted to the atmosphere as a result of human activities contribute to this man-made greenhouse effect and have to be classified in this impact category. The most important are, in order, the following (Hauschild & Wenzel, 1998):

- CO_2 (carbon dioxide)
- CH_4 (methane)
- N_2O (nitrous oxide or "laughing gas")
- Halocarbons (hydrocarbons containing chlorine, fluorine or bromine)

The potential contribution to global warming is computed with the aid of a procedure that expresses the characteristics of the substance relative to those of the other gases. For use in political efforts to optimise initiatives to counter man-made global warming, the Intergovernmental Panel of Climate Change (IPCC) has developed a characterisation factor system that can weight the various substances according to their efficiencies as greenhouse gases (Houghton et al., 1995).

The system allocates the various substances to the Global Warming Potential (GWP), which is calculated as the anticipated calculation to warming over a chosen time period from a given emission of the substance divided by the contribution to warming from an emission of a corresponding quantity of CO_2. Multiplying a known emission of greenhouse gas by the relevant GWP yields the magnitude of the CO_2 emission that, under the chosen conditions, will result in the same contribution to global warming, i.e. the emission of the greenhouse gas expressed as CO_2-equivalents.

CO_2 was chosen by the IPCC as reference substance because it is the substance that makes by far the most significant contribution to the man-made greenhouse effect. The expected contribution to warming from a greenhouse gas is calculated on the basis of knowledge of its specific IR absorption capacity and expected lifetime in the atmosphere. The GWP is internationally accepted, well documented, and provides characterisation factors for all substances encountered in a life cycle assessment. See Table 2.8 below with an example of GWP values for direct contribution of three substances mentioned before (CO_2, CH_4 and N_2O).

Table 2.8: GWP for some substances depending on the time horizon (Houghton et al., 1995)

Substance	Formula	Lifetime years	GWP (kg CO_2/kg substance)		
			20 years	100 years	500 years
Carbon dioxide	CO_2	150	1	1	1
Methane	CH_4	14.5	62	24.5	7.5
Nitrous oxide	N_2O	120	290	320	180

A frequently used indicator for evaluating human health effects of a functional unit is the Human Toxicity Potential (HTP) (Hertwich et al., 2001; Guinée et al., 1996). Two other HTP methods that are applied in this study are: CML (Heijungs et al., 1992) and EDIP (Hauschild & Wenzel, 1998). HTP is a site-generic impact potential that is easy to apply, but has got a limited environmental relevance (Bare et al., 2000).

The HTP of the EDIP method has the unit [m^3] and expresses "the volume to which the substance emitted must be diluted in order to avoid toxic effects as a consequence of the emission in question in the relevant compartment". The HTP of the CML methods is dimensionless. The Human Toxicity Potential for every pollutant p ($HTP_{p)}$ is calculated using the human toxicity factor for every pollutant ($HTF_{p)}$ and the mass of every pollutant (M_p), see expression (2.14).

$$HTP_p = HTF_p \, M_p \tag{2.14}$$

The HTF_p is expressed in units of [m^3/kg] in the EDIP method (Hauschild & Wenzel, 1998) and in [-/kg] for the CML method (Heijungs et al., 1992). The overall HTP for the functional unit is then the sum of all HTP_p (expression (2.15)).

$$HTP = \Sigma HTP_p \tag{2.15}$$

Table 2.9 shows the HTF for the pollutants considered in this study. It has to be mentioned that ozone, nitrate and sulphate are not considered in these HTF. This is due to the fact that no mass of these substances is available in the Life Cycle Inventory because they are not directly emitted but formed during dispersion in the atmosphere. Particulate Matter with a diameter of less than 10 µm (PM10) is not included either because no HTF is available.

Table 2.9: HTF from the methods CML (Heijungs et al., 1992) and EDIP (Hauschild & Wenzel) for different substances

Pollutant	CML [-/kg]	EDIP[m^3/kg]
As	4700	$9.5\cdot10^{9}$
BaP	17	$5.0\cdot10^{10}$
Cd	580	$1.1\cdot10^{11}$
Ni	0.014	$6.7\cdot10^{7}$
NO_x	0.78	$2.0\cdot10^{6}$
SO_2	1.2	$1.3\cdot10^{6}$

Other impact categories and proposed midpoint indicators

Table 2.10 gives an overview of the other impact categories that are currently used in LCIA, for each impact category one proposal of a possible indicator is shown.

Table 2.10: Impact categories and possible indicators (Udo de Haes et al., 1999)

Impact categories	Possible indicator
Input related categories	
Extraction of abiotic resources	Resource depletion rate
Extraction of biotic resources	Replenishment rate
Output related categories	
Climate change	Kg CO_2 as equivalence unit for the Global Warming Potential (GWP)
Stratospheric ozone depletion	Kg CFC-11 as equivalence unit for the ozone depletion potential (ODP)
Human toxicity	Kg benzene as Human Toxicity Potential (HTP)
Eco-toxicity	Kg phenol as Aquatic Eco-Toxicity Potential (AETP).
Photo-oxidant formation	Kg ethene as equivalence unit for photochemical ozone creation potential (POCP)
Acidification	Release of H^+ as equivalence unit for the Acidification Potential (AP)
Nutrification	Stoichiometric sum of macro-nutrients as equivalence unit for the Nutrification Potential (NP)

Udo de Haes et al. (1999) propose as input related categories extraction of abiotic resources and extraction of biotic resources. Moreover, they suggest considering land use as an impact category consisting of three subcategories: increase of land competition, degradation of life support functions and bio-diversity degradation. As output related categories they propose climate change, stratospheric ozone depletion, photo-oxidant formation, acidification and nutrification. For each impact category it is possible to use different indicators. In Table 2.10 for every category one of the mostly used indicators is shown as an example.

The impacts of the different categories have consequences on the environment and the human welfare on different spatial scales. This has nothing to do with the importance of the categories, but with a need of spatial differentiation within the fate and exposure for some impact categories. Since economic processes are spread worldwide local impacts have as well a global extension.

The climate change and the stratospheric ozone depletion are phenomena that affect the whole planet. In principle, this holds true also for the extraction of abiotic and biotic resources. However, not all regions of the world have the same need of all resources. Acidification and nutrification are generally caused by pollutants whose residence time in the atmosphere permits a continental dispersion. The impact categories human and ecotoxicity can be considered to have a regional dimension. Depending on the characteristics of the pollutant and the medium where it is emitted to, fate can be considered to be continental or local. Finally the impacts caused by photo-oxidant formation and land use are totally depending on the local situation, meteorological conditions and land characteristics. The need for spatial differentiation in the fate and exposure analysis in different impact categories is illustrated in Figure 2.15.

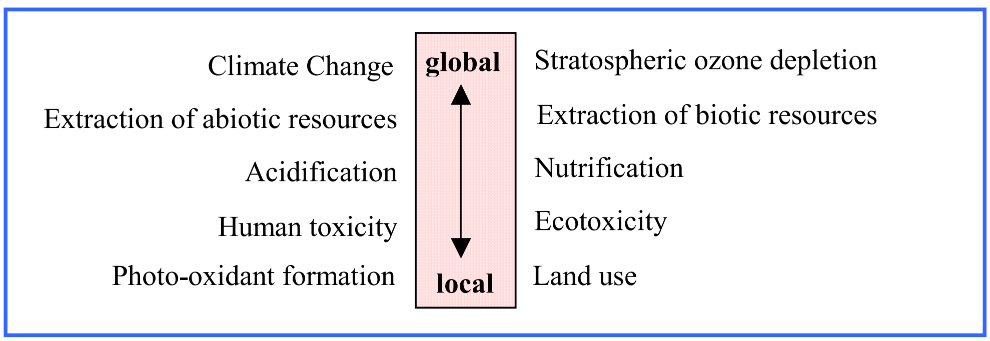


Figure 2.15: Need for spatial differentiation in different impact categories

2.3.6 SINGLE INDEX APPROACHES

In general, weighting is used for the prioritisation of one impact category as global warming in comparison to another impact category such as stratospheric ozone depletion. The prioritisation of impact categories depends normally on political targets or business strategies. Weighting is necessary to obtain a single index of environmental performance of a functional unit.

The weighting methods in LCIA can be distinguished and classified according to five types of concepts they apply (Udo de Haes, 1996). In Table 2.11 these concepts are described indicating at the same time their advantages and disadvantages. There is no simple truth to decide what works best.

Table 2.11: Comparison of concepts weighting in LCIA across impact categories

Type of concept	Description	Costs	Advantages	Disadvantages
I Proxy	Selection of one parameter for the representation of the total impact	No	Simple application.	The parameter is only a bad approximation of the total impact.
II Distance to target	Standard or environmental objectives established by the authorities as reference	No	The reference value is accepted if it exists.	There is no accepted reference value for the comparison of different impact categories.
III Panels	Consideration of the different opinions of experts and/ or the general society	No	Achievement of a value that is accepted by a group.	The result depends on the composition of the panel and/ or the selected individuals.
IV Abatement technology	Efforts to reduce the pollution by technological means as reference	Internal	The efforts can be expressed by costs that are known.	The internalised costs do not correspond to the external costs.
V Monetisation	Expression of the environmental damages in monetary values	External	It is tried to estimate the actual damage costs.	External costs can only be estimated.

Examples for the proxy approach are the Sustainable Process Index (SPI) (Sage, 1993) and the Material Intensity Per Service-unit (MIPS) (Schmidt-Bleek, 1994). Important cases for the distance-to-target methods are Eco-Scarity (Braunschweig, 1994), Eco-indicator 95 (Goedkoop, 1995) and Environmental Design of Industrial Products (EDIP) (Hauschild, 1998). Panel approaches have been used for instance by the German EPA (Schmitz et al., 1995) and in the Eco-indicator 99 weighting step (Goedkoop & Speedesma, 1999). A similar approach, the Multi-Criteria Evaluation (MCE), has been proposed for LCA by Powell & Pidgeon (1994). The abatement technology concept has been used in the method developed by the Tellus Institute (1992). It consists in an evaluation of internal environmental costs by the means of the most adequate technology to fulfil the legal requirements. Monetisation has been used as weighting scheme in some of the damage-oriented methods like Environmental Priority Strategies (EPS) (Steen, 2000) and in the Uniform World Model (UWM) (Rabl et al., 1998).

Besides the weighting scheme used, the single index approaches can be differentiated according to the fact whether impact potentials are the basis for the weighting or not. This is for instance the case for Eco-indicator 95 and EDIP, but not for Tellus and EcoScarity, where directly weighting factors are applied.

Example of a single index method: Eco-indicator 95

The single-index method used in the case study is Eco-indicator 95 (Goedkoop, 1995). It belongs to group II of Table 2.11, in the same way as the EDIP method (Hauschild, 1998) that in somehow is structured similarly. An overview of the principle of Eco-indicator 95 is given in Figure 2.16.

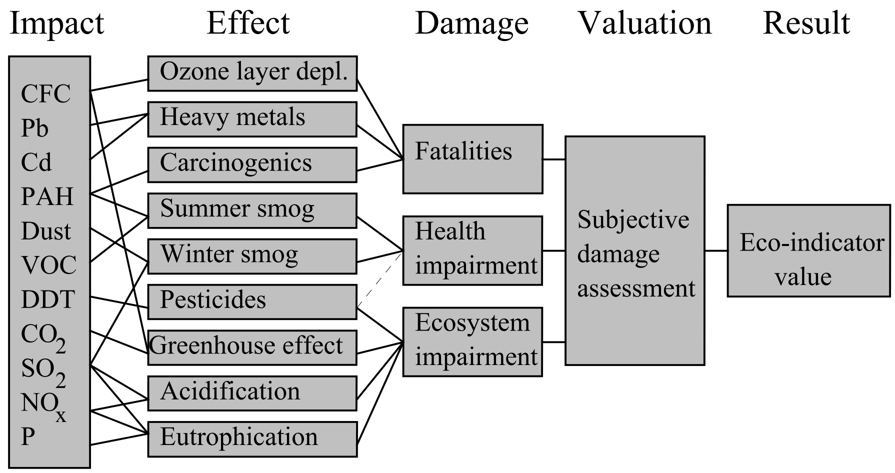


Figure 2.16: Overview of the structure of Eco-indicator 95 (Goedkoop, 1995)

The problem, of course, lies in determining the weighting factors, the subjective damage assessment phase. Much consideration has been given to this subject in the Eco-indicator 95 project. After detailed analysis of the options the so-called distance-to-target principle was chosen for determining the weighting factors. The underlying premise is that there is a correlation between the seriousness of an effect and the distance between the current level and the target level. Thus if acidification has to be reduced by a factor of 10 in order to achieve a sustainable society and smog by a factor of 5, then acidification is regarded as being twice as serious; the reduction factor is the weighting factor.

Setting equivalents for these damage levels is a subjective choice that cannot be scientifically based. It is therefore also possible to make different assumptions, which could cause the weighting factors to change. To establish a correlation between these damage levels and the effects a detailed study of the actual state of the environment in Europe was carried out within the Eco-indicator 95 project. The resulting data were used to determine the level of an environmental problem and by what factor the problem must be reduced to reach an acceptable level. Table 2.12 lists the weighting factors and the criteria applied.

Table 2.12: Weighting factors in Eco-indicator 95 (Goedkoop, 1995)

Environmental effect	Weighting factor	Criterion
Greenhouse effect	2.5	0.1°C rise every 10 years, 5% ecosystem degradation
Ozone layer depletion	100	Probability of 1 fatality per year per million inhabitants
Acidification	10	5% ecosystem degradation
Eutrophication	5	Rivers and lakes, degradation of an unknown number of aquatic ecosystems (5% degradation)
Summer smog	2.5	Occurrence of smog periods, health complaints, particularly amongst asthma patients and the elderly, prevention of agricultural damage
Winter smog	5	Occurrence of smog periods, health complaints, particularly amongst asthma patients and the elderly
Pesticides	25	5% ecosystem degradation
Airborne heavy metals	5	Lead content in children's blood, reduced life expectancy and learning performance in an unknown number of people
Waterborne heavy metals	5	Cadmium content in rivers, ultimately also impacts on people (see airborne)
Carcinogenic substances	10	Probability of 1 fatality per year per million people

2.3.7 DAMAGE-ORIENTATED METHODS

All damage-orientated methods have in common that they try to assess the environmental impacts not in the form of impact potentials, but at the damage level; that is "further down" in the cause-effect chain. In the case of human health effects that means not as HTP but as cancer cases for example.

In order to illustrate the theory behind these damage-orientated methods the reader is introduced to the Eco-indicator 99 methodology (Goedkoop & Spriensma, 1999) and the Uniform World Model (Rabl et al., 1998). Another method based on the same principles bas been developed by Steen (1999): The Version 2000 of the Systematic Approach to Environmental Priority Strategies (EPS) in Product Development. The modeling in the EPS Version 2000 indicates that it is possible - despite large uncertainties - to find an estimate of the magnitude of various impacts that are meaningful after all.

Eco-indicator 99 and Uniform World Model

As a well-documented example of a method orientated to damage level, that is those focusing at endpoints level, we have chosen the Eco-indicator 99 method developed by a large team of experts from 1997 to 1999 (Goedkoop & Spriensma, 1999).

In the development of the Eco-indicator 99 methodology, the weighting step is considered to be the most difficult, controversial and uncertain step. To simplify the weighting procedure damage categories had to be identified, and as a result new damage models were developed that link inventory results into *three damage categories*:

Damage to Human Health

Damage models were developed for respiratory and carcinogenic effects, as well as the effects of climatic change, ozone layer depletion and ionising radiation. In these models for Human Health four steps are used:

1. **Fate analysis**, linking an emission to a temporary change in concentration.
2. **Exposure analysis,** linking this temporary concentration changes to a dose.
3. **Effect analysis,** linking the dose to a number of health effects, such as occurrence and type of cancers.
4. **Damage analysis,** links health effects to DALYs (Disability Adjusted Life Years) using estimates of the number of Years Lived Disabled (YLD) and Years of Life Lost (YOLL).

Damage to Ecosystem Quality

Damages to Ecosystem Quality are expressed as percentage of species disappeared in a certain area (Potentially Affected Fraction or PAF) due to environmental stressors. The PAF is then multiplied by the area size and the time period to obtain damage. 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 2.4 an example of the calculation procedure is given for an emission to air and the resulting damage in natural soil. Finally the damages in PAFm^{2}yr of the different compartments can be added up, resulting in the total damage in Europe.

Table 2.13: PAF calculation for emissions to air and resulting damage in natural soil for 1 kg in Europe (Goedkoop & Spriensma, 1999)

Calculation step	**Calculation procedure**	**Result**
Emission to air in Europe	10,000 kg/d standard flow	1.0E-06 kg/m^2/yr
Concentration increase (C) in natural soil	EUSES	6.96E-07 mg/l
No Effect Concentration (NOEC terrestrial)	Geometric mean NOECs	1.04 mg/l
Hazard Units increase (HU)	ΔHU = ΔC/NOEC	6.69E-07
PAF/HU at Combi-PAF=24% (European average)	Slope factor = 0.593 PAF/ΔHU	
PAF increase in natural soil for 10.000 kg/d in Europe	ΔHU*0.593 = ΔPAF	4.13E-07
PAF increase in natural soil for 1 kg/yr in Europe	ΔPAF/(10,000*365)	1,130E-13
PAFm^{2}yr in natural soil (2.16E+6 km^2)	1.13E-13*surface area natural soil	0.244 PAFm^{2}yr

The whole damage category consists of:

1. **Ecotoxicity** is expressed as the percentage of all species present in the environment living under toxic stress.
2. **Acidification and Eutrophication** are treated as one single category. Damage to target species in natural areas is modelled.
3. **Land use and land transformation** is based on empirical data of occurrence particular target as a function of land use types and area size. Both local damage on occupied or transformed area and regional damage on ecosystems are taken into account.

Damage to resources

Unlike the damage categories Human Health and Ecosystem Quality, no more or less accepted unit to express damages to Resources has been found. 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 resources 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. Therefore, damage to resources, minerals and fossils fuels, is expressed as surplus energy for the future mining of resources. The surplus energy is defined as the difference between the energy needed to extract a resource now and at some point in the future:

1. **For minerals** geo-statistical models are used. Geo-statistical 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, the “decrease” of a concentration as a result of an extraction is modelled.
2. **For fossil fuels** surplus energy is based on future use of non-conventional resources, especially oil shale and tar sands. 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, the model for the surplus energy is constructed.

Eco-indicator 99 methodology used basically *three types of models*:

1. Modelling of technosphere in the inventory phase.
2. Modelling of ecosphere in the impact assessment phase.
3. Modelling of valuesphere in weighting and ranking, as well as in order to deal with unavoidable value choices.

In this way, a complete “top-down” re-engineered impact assessment method, based on the decisions made in the valuesphere, with four clearly detailed steps were developed: fate, exposure, land use and damage analysis. This is in contrast with the “bottom-up” approach that can be found in the more traditional midpoint methods. The Figure 2.17 gives an overview of the Eco-indicator 99 method.

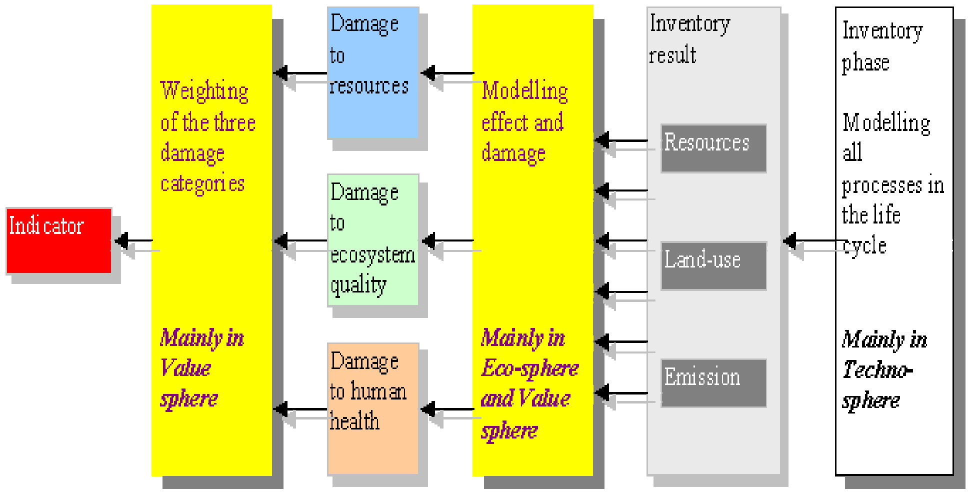


Figure 2.17: Overview of the Eco-indicator 99 method. The term sphere is used to indicate that the method integrates different fields of science and technology. (Goedkoop and Spriensma, 1999)

Endpoint approach: Uniform World Model

Based on the IPA studies on a European level in the ExternE Project, see section 2.4.4, Rabl et al. (1998) compared the results of detailed site-specific calculations for more than 50 electric power stations and MSWIs all over Europe and introduced a Uniform World Model (UWM) with the expressions (2.16) and (2.17) for:

a) primary pollutants

$$D = D_{uni} = \frac{f_{CR} \cdot \rho_{uni}}{k_{uni}} \cdot Q \qquad (2.16)$$

where: $D = D_{uni}$ = (uniform) damage [cases/a]

f_{CR} = slope of concentration-response function (dose-response or exposure-response) [cases/persons*a*µg/m³]

ρ_{uni} = uniform receptor density [1.05E-04persons/m²][3]

k_{uni} = uniform removal velocity [m/s]

Q = emission [µg/s]

[3] This density is 2/3 of the average EU15 countries' population density of 158 persons per km² (Rabl et al., 1998).

b) secondary pollutants

$$D_{2uni} = D_{uni} = \frac{f_{CR2} \cdot \rho_{uni}}{k_{2uni,eff}} \cdot Q_1 \quad (2.17)$$

where: D_{2uni} = uniform damage due to secondary pollutant [cases/a]
f_{CR2} = slope of concentration-response function for secondary pollutant (dose-response or exposure-response) [cases/persons*a*µg/m³]
$k_{2uni,eff}$ = effective uniform removal velocity for secondary pollutant[m/s]
Q_1 = emission of primary pollutant [µg/s]

The slope of C-R functions states the incremental number of cases (e.g. hospitalisations per concentration increment). Table 2.14 shows typical removal velocity values as they had been found by calculations with EcoSense (see section 2.4.6.).

Table 2.14: Typical removal velocity values for different pollutants (Rabl et al., 1998)

a) Primary pollutants

Pollutant	$\mathbf{k_{uni}}$ [m/s]
PM10	0.01
SO_2	0.01
CO	0.001
Heavy metals	0.01
PCDD/Fs	0.01

b) Secondary pollutants

Pollutant	$\mathbf{K_{2uni,eff}}$ [m/s]
$NO_2 \rightarrow$ nitrates	0.008
$SO_2 \rightarrow$ sulphates	0.019

Even though the assumption that the removal velocity k_{uni} is universal may not appear very realistic, especially for near point sources, Rabl et al. (1998) found out that the deviation is surprisingly small. The reason is that the total damage is dominated by regional damages, which occur sufficiently far away from the source, where the pollutant is well diluted and the difference of the model from real conditions is negligible. Thus it is plausible that these results are fairly representative and that the UWM can be a useful tool to have a first estimate within an order of magnitude of damage estimates expressed as external costs, monetised according to the guidelines of the European Commission (1995). Table 2.15 shows the results obtained of that study. The multipliers indicate how much the costs can change with site and stack conditions (height, temperature and exhaust velocity).

Table 2.15: European health damage costs per kg emitted pollutant calculated with the Uniform World Model (Rabl et al., 1998)

Pollutant	**Cost** [mEuro/kg Poll]	**Multiplier for site** (rural ↔ urban)	**Multiplier for stack emissions** (height 250 ↔ 0m,T, v*)
CO	2.07E+00	?	?
NO_x via Nit	1.69E+04	≈ 0.7 ↔ 1.5	≈ 1.0
SO_2 tot	1.22E+04	≈ 0.7 ↔ 1.5	≈ 1.0
PM 10	1.36E+04	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
As	1.50E+05	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Cd	1.83E+04	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Cr	1.23E+05	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Ni	2.53E+03	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
PCDD/Fs	1.63E+10	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0

* T – temperature, v – exhaust velocity

2.3.8 Dose-response and exposure-response functions in LCIA methods

As all analytical environmental impact assessment methods LCIA includes an effect analysis. However, due to the use of characterisation and weighting factors, which in general are not at all transparent, the importance of the dose-response and exposure-response functions is not so obvious. However, it exists and can be seen as one reason why different HTP approaches, for example CML by Heijungs et al. (1992) and EDIP by Hauschild & Wenzel (1998), do not converge. Here is certainly a need to communicate to the user of such LCIA factors the value choices made when selecting the source of the damage functions.

As another example using dose-response functions and the DALY concept, here the recent proposal for a new LCIA human toxicology approach by Crettaz et al. (2002) is briefly mentioned. Derived from the health risk assessment concept of benchmark doses, the approach is based on the maximum likelihood estimate of the effect dose inducing a 10% response over background, denoted ED_{10}, which provides a point of departure to calculate the linear slope factor β_{ED10} using bioassay data and the tumour dose TD_{50} for cancer effects or the NOEL for non-cancer effects. The resultant slope factors range from 10^{-6} to 10^{3} [risk per mg/kg-day dose]. It has to be seen if the ED_{10} can find the same acceptance for LCIA as the RfD value for ERA and if the extrapolation done through the slope factor β_{ED10} is not implying a similar level of uncertainty.

Differing from many default non-cancer risk assessment approaches adopted for regulatory compliance, current LCIA approaches often assume, sometimes implicitly, a linear low dose-response function and the absence of thresholds (Crettaz et al., 2002). These assumptions remain controversial hypotheses (Potting, 2000) and are discussed here shortly. Slob (1999) provided a discussion of the threshold/ no-threshold debate and argued, however, that dose-thresholds in a strict sense cannot exist in dose-response relationships that are studied by toxicologists. Barton et al. (1998) stated that even though frequently the concept of a threshold may be biologically plausible, it is very difficult, if not impossible, in practice to differentiate between the observance of no response because a dose is below a threshold and the failure to observe a response, due to being below the limit of detection in a typical laboratory study with a limited number of animals. Pilkington et al. (1997) recommended that the quantification of all respiratory effects should be made on a non-threshold basis, since recent epidemiological studies suggest that there is no safe level of air pollution. These sources indicate that sharp thresholds cannot be demonstrated, at least for many air pollutants.

An alternative to linear low-dose methods in LCA are the "only above threshold" approaches (White et al., 1995; Hogan et al., 1996) adapted from regulatory health risk assessments. The method proposed by White et al. (1995), for example, identifies processes resulting in the largest share of emissions in a product's life cycle. Information about the actual location of these processes and their emissions is then gathered. This site-specific information is then analysed using typical health risk assessment techniques. An emission is only considered further if the guidance measure, such as an RfD, is exceeded by an exposure at a given location and point in time. These methods therefore differ from the objectives of most LCAs. For example, there is no estimation of the residual risks below regulatory measures associated with the full time and space integrated exposures. This difference is particularly important when considering chemicals that are widely dispersed and subject to long-range transport.

2.3.9 SOPHISTICATION IN LCIA

Sophistication in LCIA has been an important topic for scientific discussion (Bare et al., 1999). Sophistication is considered to be the ability to provide very accurate and comprehensive reports that reflect the potential impact of the stressors to help decision-making in each particular case. In language more consistent with recent ISO publications, the practitioners of LCA are faced with the task of trying to determine the appropriate level of sophistication in order to provide a sufficiently comprehensive and detailed approach to assist in environmental decision-making. Sophistication has many dimensions and, dependent upon the impact category in LCIA, may simulate the fate and exposure, effect and temporal and spatial dimensions of the impact. It has the ability to reflect the environmental mechanism with scientific validity (Udo De Haes et al., 1999; Owens, 1997; Udo de Haes, 1996; Fava et al., 1993).

Levels of sophistication

Traditionally LCIA uses linear modelling, takes the effects of the substances into account, but not their fate and background concentrations, and aggregates the environmental consequences over:

- time,
- locations, ⇒ "potential impact"
- chemicals.

All this only allows calculating potential impact scores, but not actual damages.

Therefore, the appropriate level of sophistication of LCIA involves quite a number of issues. A major point concerns the extension of the characterisation modelling to include also the fate of the substances and not only their effects. Another issue concerns a possible differentiation in space and time. Studies can include impact models that use data just at world level and do not specify time periods; in contrast, more recent options involve spatial differentiation of impacts and distinguish between different time periods. A further point concerns the type of modelling. More sophisticated possibilities arise which take background levels of substances into account and make use of non-linear dose-response functions. An important question here is whether there are real science-based thresholds, or whether these thresholds are always of a political origin. A further question relates to the role and practicality of including uncertainty analysis. Sensitivity analysis is increasingly included in LCA studies; but this is not yet the case for uncertainty analysis. Finally, there is the question of how to apply these different options for sophistication of LCIA, which applications can afford to keep it simple, and for which applications a more detailed analysis is needed. An overview of these different levels of detail in the characterisation step of LCIA is given in Figure 2.18.

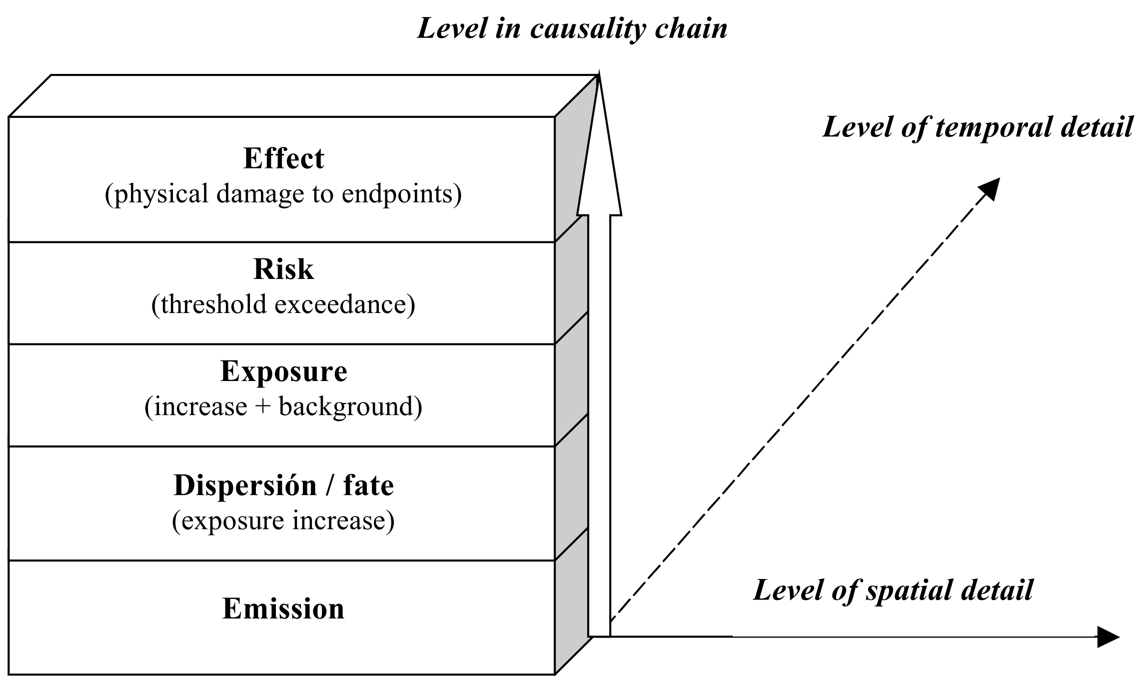


Figure 2.18: Levels of detail in impact characterisation (Potting, 2001)

The important issue of deciding the appropriate level of sophistication is typically not addressed in LCA. Often, the determination of the level of sophistication is based on considerations that may be appropriate for a scientific point of view, but which include practical reasons for limiting sophistication (e.g., the level of funding). A discussion of the most appropriate ways of determining sophistication will include:

- Study objective
- An uncertainty and/ or sensitivity analysis
- The inventory data and availability of accompanying parameters
- Depth of knowledge and comprehension in each impact category
- The quality and availability of modelling data
- Available supporting software
- The level of financial resources

Midpoint and endpoint approaches

Although the terms have yet to be clearly defined, midpoints are considered to be a point in the cause-effect chain (environmental mechanism) of a particular impact category, prior to the endpoint, at which characterisation factors can be calculated to reflect the relative importance of an emission or extraction in a Life Cycle Inventory (e.g. global warming potentials defined in terms of radioactive forcing and atmospheric half-life differences). That is, midpoints are located a step before endpoints and allow to calculate in a relative way the environmental impact of any emission defined in the Life Cycle Inventory (Bare et al., 2000). Historically, the midpoint approaches have set the scene in LCIA, taken as prominent examples the thematic approach (Heijungs et al., 1992), the Sandestin workshop on LCIA (Fava et al., 1993), the Nordic LCA guide (Lindfors et al., 1995), the Eco-indicator 95 method (Goedkoop, 1995) and the EDIP model (Wenzel et al., 1997). They also have mostly structured the way of thinking and examples chosen in ISO 14042 (2000).

Since the middle of the nineties the endpoint approach was set on the agenda. Particularly in LCA studies that require the analysis of tradeoffs between and/ or aggregation across impact categories, endpoint-based approaches are gaining popularity. Such methodologies include assessing human health and ecosystem impacts at the endpoint that may occur as a result of climate change, ozone depletion and other categories traditionally addressed using midpoint category indicators. The endpoint approach had already a longer history, particularly in the EPS approach from Steen and Ryding (Steen &Ryding, 1992; Steen, 1999), but got strong impetus from the Netherlands in the Eco-indicator 99 approach (Goedkoop & Spriensma, 1999). At the moment also in Japan impact assessment models are developed according this approach (Itsubo & Inaba, 2000). This approach starts from the main values in society, connected with Areas of Protection. From these values and the connected endpoints the modelling goes back to the emissions and resources consumptions (Udo de Haes & Lindeijer, 2001).

Figure 2.19 shows the steps that can be involved if a practitioner wishes to take an LCA study from the inventory stage, via midpoints and endpoints in the impact assessment to valued scores. Not all possible environmental loads can be considered in the inventory since not for all of them data are available. Based on the inventory table, two different routes are presented to come to valued scores, representing the routes taken when using midpoint and endpoint approaches (Bare et al., 2000). On the one hand, the impact categories that can be expressed in the form of midpoints are directly presented as valued scores; on the other hand, as far as possible according to current knowledge, the impact are expressed in the form of endpoints by relating the midpoints to endpoints or by modelling effects directly from the inventory to the endpoints. Then several endpoints can be aggregated to a valued score if the selected weighting scheme it allows.

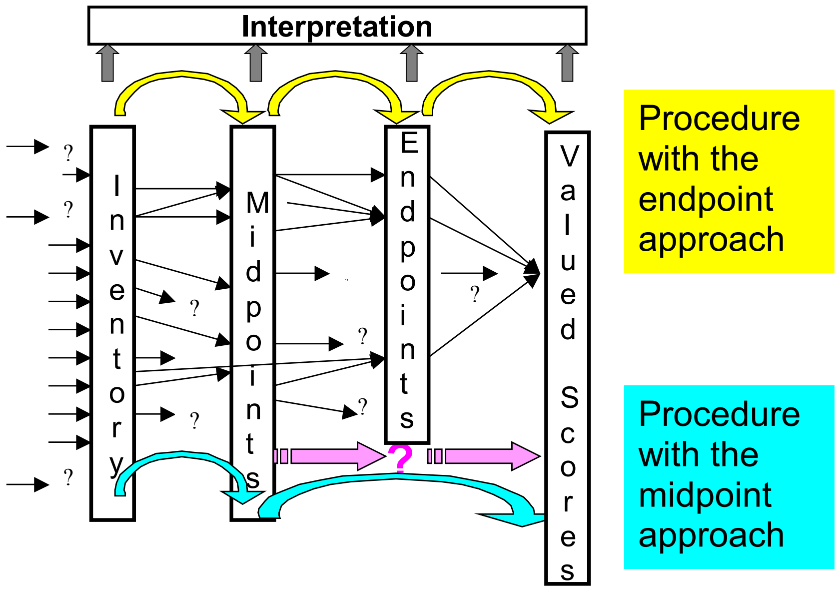


Figure 2.19: Some basic differences between the midpoint (lower row of swinging arrows) and the endpoint approach (upper row of swinging arrows) (Bare et al., 2000)

One of the key differences between midpoint and endpoint approaches is the way in which the environmental relevance of category indicators is taken into account. In midpoint approaches, the environmental relevance is generally presented in the form of qualitative relationships. Indicators chosen at an endpoint level are generally considered more understandable to the decision makers. As a consequence different types of results are presented to the decision-maker (Bare et al., 2000).

Endpoint modelling may facilitate more structured and informed weighting, in particular science-based aggregation across categories in terms of common parameters (for example, human health impacts associated with climate change can be compared with those of ozone depletion using a common basis such as DALYs – Disability Adjusted Life Years).

Proponents of midpoint modelling believe, however, that the availability of reliable data and sufficiently robust models remains too limited to support endpoint modelling. Many believe that extending the models to endpoints reduces their level of comprehensiveness (the number of pathways and endpoints in the cause-effect chains that are represented as a step beyond the well characterised midpoints) and that such extensions will be based on a significant number of additional, unsubstantiated assumptions and/ or value choices, which may not reflect the viewpoint of other experts and/ or the users to fill in missing gaps. One major concern is that uncertainties may be extremely high beyond well-characterised midpoints, resulting in a misleading sense of accuracy and improvement over the midpoint indicators. Many modellers believe that the additional complexity and detail is only warranted if it can be demonstrated to provide an improvement in the decision-making basis (Bare et al., 2000).

2.4 SITE-ORIENTATED ENVIRONMENTAL IMPACT ASSESSMENT

In opposite to these process chain-orientated tools, site-orientated environmental impact assessment tools are presented in this section. As principal methods for this, the site-dependent approaches Environmental Risk Assessment and Impact Pathway Analysis are described here together with their respective models.

2.4.1 ENVIRONMENTAL RISK ASSESSMENT (ERA)

Risk assessment has become a commonly used approach in examining environmental problems. Risk assessment is the procedure in which the risks posed by inherent hazards involved in processes or situations are estimated quantitatively. Due to the unclear definition of the word "risk" in the everyday use, two basic definitions are necessary:

- *Definition of hazard*: Hazard is commonly defined as the potential to cause damage. A hazard can be defined as a property or situation that in particular circumstances could lead to harm (Fairman et al., 1998).
- *Definition of Risk*: The combination of the probability or frequency of occurrence of a defined hazard and the magnitude of the consequences of the occurrence (Fairman et al., 1998), see expression 2.18.

$$\text{consequence/ event} \ * \ \text{event/ year} = \text{consequence/ year} \qquad (2.18)$$

There has been a gradual move in environmental policy and regulation from hazard-based to risk-based approaches. A risk based approach attempts to examine the actual risks imposed by an environmental issue rather than the potential hazards that may, or may not arise.

In order to estimate the *individual risk* of the affected inhabitants in an analysed region the location of the towns and settlements close to the plant whose emissions are assessed is relevant. For the computation of the *population risk* the number of inhabitants have to be multiplied with the calculated individual risk.

In the life cycle of a chemical, risk can arise during manufacture, distribution, use, or the disposal process. Environmental risk assessment of the chemical involves, in principle, the identification of the inherent hazards at every life cycle stage and an estimation of the risk posed by these hazards.

The Environmental Risk Assessment (ERA) consists of the procedure to evaluate the probability that adverse effects on the environment or the human health occur or may occur, as consequence of the exposure to one or more physical, chemical or biological agents. The evaluation of the environmental risk requires the knowledge of the adverse effects that might be caused by the exposure to chemical substances or materials as well as of the necessary intensity and duration to enable them producing adverse effects on the environment including the population. Also, the intrinsic knowledge of the physical-chemical properties of the pollutants, the bio-degradability, the potential of bioaccumulation or potential effects of the chemical substances is necessary for the evaluation of the environmental risk. Moreover, it is necessary to carry out a detailed evaluation of the emission sources, of the fate and its distribution in the different media. For all this, the analysis of environmental samples in the laboratory and the application of mathematical models are vital (EC, 1996).

The National Academy of Sciences (1983) suggested to divide the environmental risk assessment in four steps. So the risk assessment procedure, in relation to both human health and the environment, entails a sequence of action, which is shown in Figure 2.20 and outlined below:

1. Assessment of effects, compromising
   - hazard identification: identification of the adverse effect which a substance has an inherent capacity to cause, and
   - effect assessment: estimation of the relationship between dose, or level of exposure to a substance, and the incidence and severity of an effect, i.e. the above-mentioned dose-response and exposure-response functions
2. *Exposure assessment:* estimation of the concentrations/ doses to which human populations (i.e. workers, consumers and individuals exposed indirectly via the environment) or environmental compartments (aquatic environment, terrestrial environment and air) are or may be exposed.
3. *Risk Characterisation:* estimation of the incidence and severity of the adverse effects likely to occur in a human population or environmental compartment due to actual or predicted exposure to a substance, i.e. the quantification of that likelihood.

In the EU a Technical Guidance Document on ERA exists in support of the Commission Directive 93/67/EEC on Risk Assessment for New Notified Substances and the Commission Regulation (EC) 1488/94 on Risk Assessment for Existing Substances (EC, 1996).

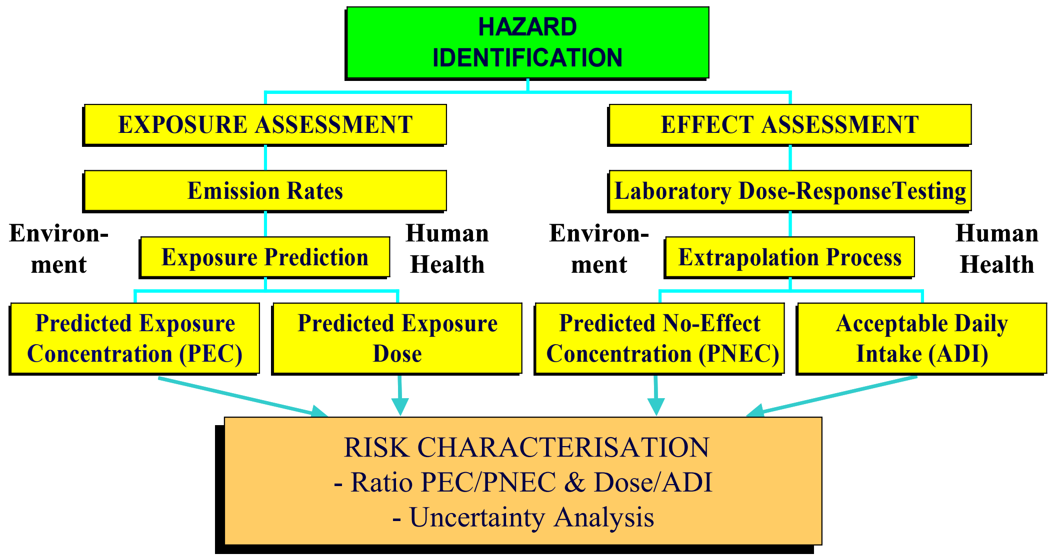


Figure 2.20: Framework of Environmental Risk Assessment (adapted from Fairman et al., 1998).

The examination of the total exposure is a comprehensive assessment that needs multimedia modelling. A chemical's final distribution in the media and the respective concentrations are the result of numerous highly complex and interacting processes that are not easy to assess. Fate models have been developed to simulate pollutant transport among and transportation within multiple environmental media. These models are referred as multimedia fate models. The development of the ERA methodology made an important step forward with the inclusion of such multimedia models for the pollutant fate calculations. A general overview of the exposure assessment by the multimedia modelling for human health gives Figure 2.21.

This figure shows the influence of human activities on the environment. It illustrates the connections of the different important impact pathways within the total exposure. So this figure describes the main aim of a multimedia modelling: to identify the different pathways of the pollutant fate and to calculate the total exposure due to the different sources. Starting with the emission of the plant, the distribution of the pollutants goes over the air, the soil, the ground water and the surface water in the other bordering compartments. Through these paths the pollutants can enter in the food chain of human beings and animals. With the direct consumption of agricultural product, i.e. plants or animals, humans stand in the direct and indirect influence of the pollutants. So the exposure is divided into the consumer exposure itself and the part of the exposure via the environment. At the end there is the man with different health effects through the hazardous activity.

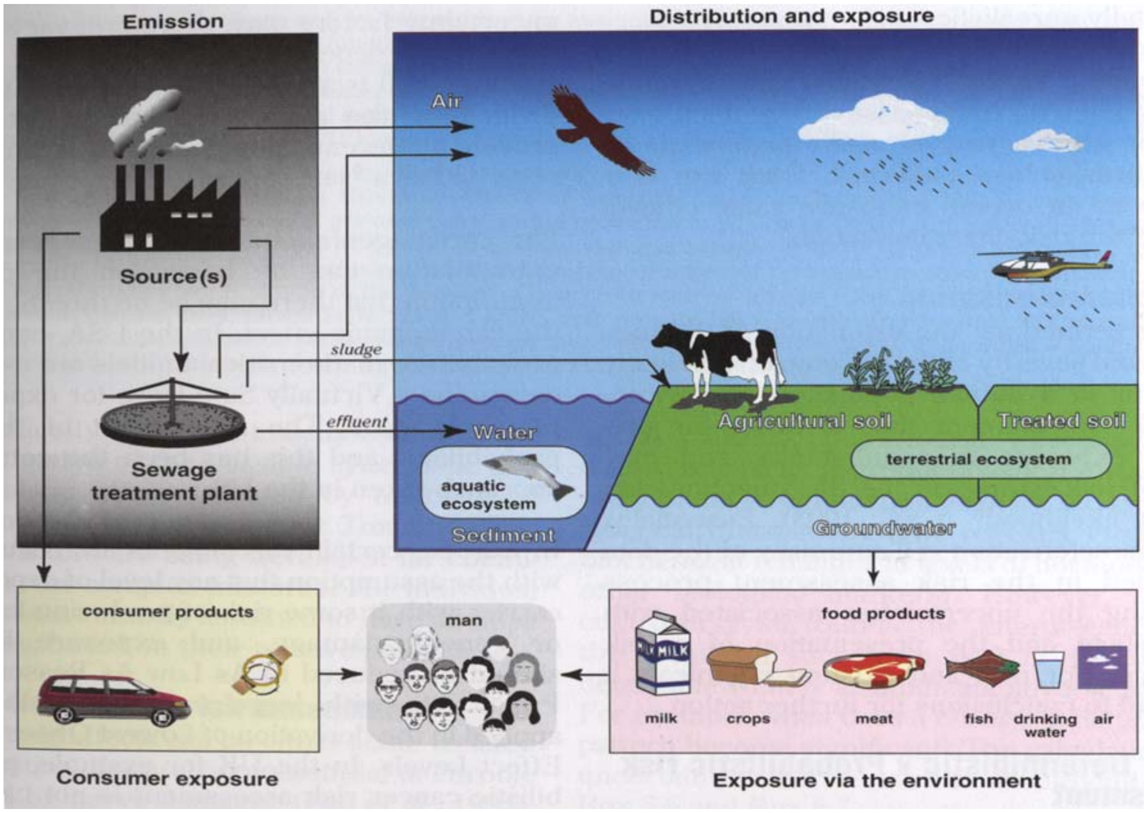


Figure 2.21: Cause-effect chain for ecosystem and human health as basis for exposure assessment by multi-media modelling (Fairman et al., 1998)

The present work will only assess the risk of a hazardous activity to human health. The used steps and methods in the present paper are principally valid for all kind of Environmental Risk Assessments, i.e. also for other affected environmental receptors, as plants, materials etc.

The steps for the environmental health risk assessment with multimedia modelling are the following (STQ, 1999):

1. Identification of the different emitted pollutants
2. Characterisation of the fate and transport in the environment
3. Characterisation of the exposure
4. Quantification of the health effects

2.4.2 Comparison of Environmental Impact Assessment and Life Cycle Assessment

Table 2.16 shows a comparison of Environmental Risk Assessment and Life Cycle Assessment.

Table 2.16: Comparison of Environmental Risk Assessment and Life Cycle Assessment

Criteria	Environmental Risk Assessment	Life Cycle Assessment
Object	Industrial process or activity	Functional unit, i.e. product or service, with its life cycle
Spatial scale	Site specific	Global/ site-generic
Temporal scale	Dependent on activity	Product life
Objective	Environmental optimisation by risk minimisation	Environmental optimisation by reduction of potential emissions and resource use
Principle	Comparison of intensity of disturbance with sensitivity of environment	Environmental impact potential of substances
Input data	Specific emission data and environmental properties	General input and output of industrial processes
Dimension	Concentration and dose	Quantity of emissions
Reference	Exposure potential to threshold	Characterisation factor
Result	Probability of hazard	Environmental effect score

This comparison shall be illustrated by the example of electricity generated from coal and produced in the same way but in two different regions, in which obviously the combustion of coal is an important part of the life cycle:

- Case 1: In very populated and acidification-sensitive area next to the mining site
- Case 2: In purely populated and no acidification-sensitive area far away from mining site

According to Sonnemann et al. (1999b), probably the LCA will state the minimal total emissions and energy demand for the Case 1 due to the importance of the additional transport and the negligence of the specific region. In contrast, the ERA will state the minimal risk for the environment for the Case 2, because the focus in only on the main process in the life cycle, but the extra transport is not considered.

This example shows in a simple way the significance of the difference highlighted in Table 2.16. It demonstrates also clearly the need for a more integrated approach that does not allow so easily that two environmental impact assessment tools provide such contradictory and inconsistent results.

Olsen et al. (2001) emphasise the feature of LCA as a relative assessment due to the use of a functional unit, while ERA is an absolute assessment, which requires very detailed information on e.g. the exposure conditions. It is concluded that the conceptual background and the purpose of the tools are different, but that there are overlaps where they may benefit from each other.

2.4.3 FATE AND EXPOSURE MODELS

Exposure risk assessment needs fate and exposure models to simulate the behaviour of the emitted pollutants in the environment. Basically, two types of models exist: those highly developed for the transport in one media, e.g. Gaussian air dispersion models or aquifer pollutant transport models, and the so-called multimedia fate models that simulate the inter-media transport.

Corresponding to the case study chosen, the most relevant models in the application of ERA are Gaussian air dispersion models and multimedia fate models. Therefore, they are presented in this section.

Gaussian air dispersion models

Close to the plant, i.e. at distances of some 10-100 km from the plant, chemical reactions in the atmosphere have little influence on the concentrations of primary pollutants. Due to the elevated emission height of a tall stack, the near surface ambient concentrations of the pollutants at short distances from the stack are heavily dependent on the vertical mixing of the lower atmosphere. Vertical mixing depends on the atmospheric stability and the existence and height of inversion layers (whether below or above the plume). For these reasons, the most economic way of assessing ambient air concentrations of primary pollutants on a local scale is a model which neglects chemical reactions, but is detailed enough in the description of turbulent diffusion and vertical mixing.

An often-used model that meets these requirements is the so-called Gaussian plume model. The concentration distribution from a continuous release into the atmosphere is assumed to have a Gaussian shape, see expression 2.19:

$$c(x,y,z)=\frac{Q}{u2\pi\sigma_y\sigma_z}\cdot\exp\left[-\frac{y^2}{2\sigma_y^2}\right]\cdot\left(\exp\left[-\frac{(z-h)^2}{2\sigma_z^2}\right]+\exp\left[-\frac{(z+h)^2}{2\sigma_z^2}\right]\right) \tag{2.19}$$

where: *c(x,y,z)* = concentration of pollutant at receptor location (x,y,z)
Q = pollutant emission rate (mass per unit time)
u = mean wind speed at release height
σ_y = standard deviation of lateral concentration distribution at downwind distance
σ_z = standard deviation of vertical concentration distribution at downwind distance x
h = plume height above terrain

The assumptions embodied into this type of model include those of idealised terrain and meteorological conditions so that the plume travels with the wind in a straight line, mixing with the surrounding air, both horizontally and vertically, to produce pollutant concentrations with a normal (Gaussian) spatial distribution (Figure 2.22). Dynamic features that affect the dispersion, for example vertical wind shear, are ignored. These assumptions generally restrict the range of validity of the application of these models to the region within some 100 km of the source. Pollution transport however extends over much greater distances. The assumption of a straight line is rather justified for a statistical evaluation of a long period, where mutual changes in wind direction cancel out each other, than for an evaluation of short episodes.

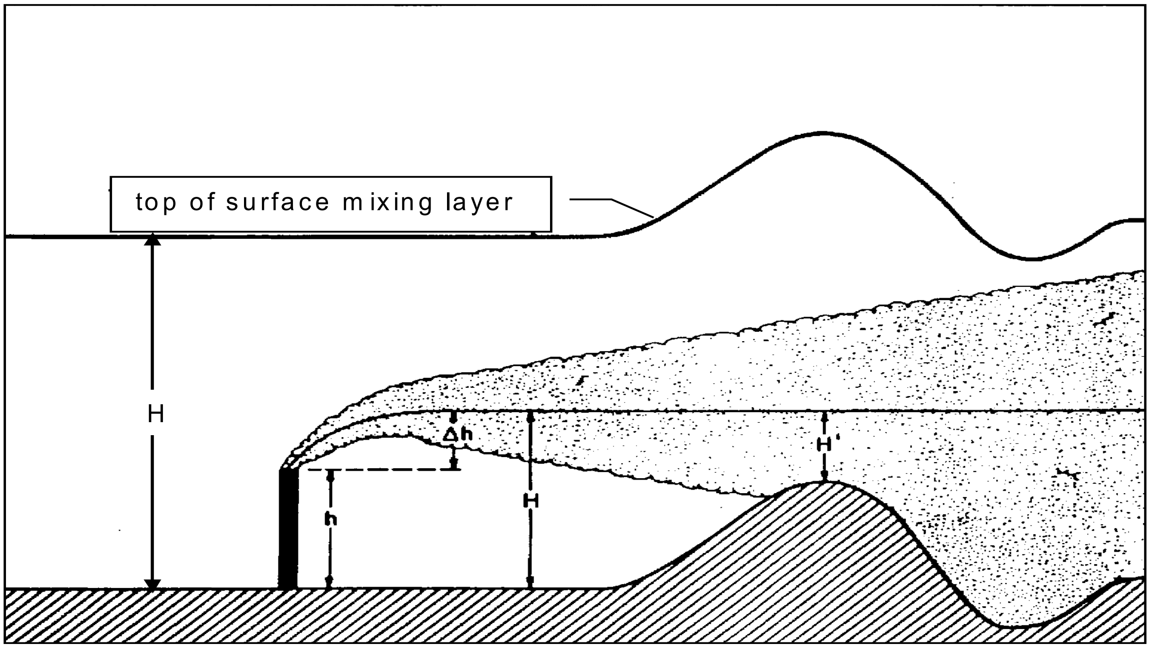


Figure 2.22 Gaussian plume shape (*H* – mixing layer height, h – stack height, H – plume height assuming flat terrain, H* - plume height above terrain)

In this study the Industrial Source Complex Short Term model, version 2 (ISCST-2) of the US-EPA (1992) and version 3 (ISCST-3) of US-EPA (1995) in the form of BEEST (Beeline, 1995) have been applied. The model calculates hourly concentration values of gases and particulate matter for one year at the centre of each specified grid. Effects of chemical transformation are neglected. Annual mean values are obtained by temporal averaging of the hourly model results. Currently US-EPA has proposed the establishment of a new regulatory dispersion model AERMOD (US-EPA, 2002).

The σ_y and σ_z diffusion parameters are based on the results of tracer experiments at emission heights of up to 195 m (Nester & Thomas, 1979). More recent mesoscale dispersion experiments confirm the extrapolation of these parameters to distances of more than 10 km (Thomas & Vogt, 1990).

The ISCST model assumes reflection of the plume at the mixing height, i.e. the top of the atmospheric boundary layer. It also provides a simple procedure to account for terrain elevations above the elevation of the stack base:

- The plume axis is assumed to remain at effective plume stabilisation height above mean sea level as it passes over elevated of depressed terrain.
- The mixing height is terrain following.

- The effective plume stabilisation height h_{stab} at receptor location *(x,y)* is given by expression (2.20).

$$h_{stab} = h + z_s - \min\left(z\big|_{(x,y)}, z_s + h_s \right) \tag{2.20}$$

where: h = plume height, assuming flat terrain
h_s = height of the stack
z_s = height above mean sea level of the base of the stack
$z\big|_{(x,y)}$ = height above mean sea level of terrain at the receptor location

Mean terrain heights for each grid cell are necessary and it is also the responsibility of the user to provide the meteorological input data. These include wind direction, wind speed, stability class as well as mixing height, wind profile exponent, ambient air temperature and vertical temperature gradient.

The Beest software (Beeline, 1998) works with a further developed version, the ISCST 3. The major enhancements in the new version ISCST-3 consist of a new dry-deposition algorithm and the addition of a wet-deposition algorithm. The physical processes that affect deposition include pollutant properties such as size, shape and density of particles, surface characteristics and atmospheric variables. Deposition velocity is characterised by gravitational settling and aerodynamic resistance. Therefore, the physical attributes of the particle, such as diameter, shape, and density, are important and are now direct inputs into the ISCST 3 model (Atkinson et al., 1997). Calculations of the aerodynamic resistance require user input values for surface friction velocity and Monin-Obukhov length. These input parameters have been incorporated into the RAMMET meteorological pre-processor program for the ISCST 3 model (US-EPA, 1995).

It would be very helpful for the comparability of dispersion modelling if the respectively most recent version of ISCST (or another model) would be accepted as an international reference. There are more sophisticated models, and nearly each research group has developed its own model, but nevertheless all models have their inherent uncertainties; and the ISCST model can be considered as the one mostly used worldwide, especially for regulatory purposes in EIA and ERA. The existence of such a reference model would allow to compare the results of different studies with less problems what is often more important as the overall accuracy.

Multimedia models

Table 2.17 includes a list of multimedia environmental fate and exposure models. The models are grouped according to certain similarities. The second column indicates special properties of the model. The two models that have been used in LCA, USES and CalTOX, were until recently the only models that had an exposure assessment component. TRIM, which was recently developed by US-EPA and still undergoes evaluation and testing, also has an exposure assessment component (Hertwich et al., 2000). The majority of multimedia models can only be used for analysing the fate of organic pollutants.

In the case study the EXCEL based program UNIRISK was used. UNIRISK has been developed within the Environmental Management and Analysis (AGA) Group in cooperation with the Toxicology and Environmental Health Group at the Universitat Rovira i Virgili. The program is based on the unit-risk concept for the estimation of the dose-response relationship (WHO, 1987), as explained in section 2.2.1.

The objective of UNIRISK is to estimate the health risk that signifies for the population the emission from an industrial process. UNIRISK estimates the pollutant transport from air to soil and plant compartments. In order to validate the models, the predicted results have been compared with data obtained in a monitoring program performed in the study region. When we talk about risk calculation models, food and general life habits of a particular region have to be taken into account. A general model should include all these variables as an input.

UNIRISK calculates only the increased human health risk of emitted pollutants. So the estimation of other kinds of risk is not included. Furthermore, it is restricted to pollutant emissions into the air. The third restriction is the adaptation of the risk calculation to the living conditions in the Mediterranean region Catalonia, i.e. the consideration of the habits of life, e.g. eating, staying outside etc. For a calculation in another region than Catalonia the program needs to be adapted to the new region.

Table 2.17: Prominent multimedia fate models (Hertwich et al., 2000)

Model	Remarks
CalTOX	Integrated fate and exposure model for humans, used by Californian EPA for hazardous waste site assessment and by EDF for risk calculations. Level III model with time dependent soil compartment.
EQC	Level III fugacity model that describes the regional partitioning of chemicals. Currently used by DuPont and US-EPA for scoring methods.
SimpleBox 1.0, 2.0	Level III fugacity based fate model; recent release is an evaluative model with nested boxes (local, regional, continental, global).
USES 1.0, 2.0 and EUSES1.0	Integrated fate and exposure model both for humans and a limited number of animals, based on SimpleBox.
TRIM	Combination of transport model and multimedia fate model, not based on the fugacity approach. Exposure assessment both for humans and ecosystems. Highly complicated, only for expert use.
MendTox	Level IV fugacity model implemented as simulation model for local, site-specific environments.

UNIRISK is formed by two simple-models to predict the concentration in vegetation and in soils. Both models consider three compartments: air, vegetation and soil. The soil and vegetation concentration is calculated based on the concentration in the air, proceeding from given (calculated or measured) pollutant concentration in this medium. The model uses several constants that are depending on the examined region. The exposure through the compartments water, sediment and biota is not considered.

The program calculates the cancer and the non-cancer risk of the different emitted pollutants. The total risk is the result of the different types of risk through ingestion, inhalation and dermal adsorption.

Inhalation of air
To estimate the exposure through the inhalation of air, UNIRISK uses a ventilation volume of 5 m^3 per day for little children and 20 m^3 for adults. The retention of the pollutant in the body depends on the chemical and physical properties of the pollutant and is calculated from 25 % up to 80 % of the inhaled air.

Inhalation of resuspended soil particles
The inhalation of resuspended soil particles is one way to come in contact with soil. For this estimation UNIRISK makes a difference between indoor and outdoor contact. The exposure is calculated by multiplying the concentration of the adsorbed pollutants (at the atmospheric particles) with the breath volume in a defined time and the retention rate of the lungs. The atmospheric particle concentration is a calculated or measured value. The model of Hawley for the presented pollutants in the soil supposes that 50 % of the particles in the atmosphere are coming from the resuspension of the soil (Hawley, 1985).

Ingestion of soil
Humans ingest small amounts of soil indirectly (hand-to-mouth transfer) which depend on the potential extent of exposure to soil. The level of the diary ingestion of soil is considered in UNIRISK for little children with a value of 150 mg/day. This value is above the estimated value of 30 mg up to 90 mg per day for the indirect intake through dirty hands and toys and below the value of the direct intake of soil, which could be up to 10 g/day. The ingestion of soil varies depending upon the age of the individual, the relation of outdoor to indoor activity, frequency of hand-to-mouth contact and seasonal climate. The dairy intake of soil by the adults is calculated in UNIRISK with a value of 60.45 mg/day.

Ingestion of vegetables from the area
The population may consume vegetables grown in the studied area; usually most vegetables in a diet do not come from the area of potential influence of the plant. For the consumption of vegetables UNIRISK makes the assumption that a child until two years eats in average 36 g/day within only 10 % of this vegetables coming direct from the assessed zone. It means that the ingestion of contaminated plants amounts to 3.6 g/day. Adults eat an average of 122 g vegetables per day. With the assumption that only 10 % are coming directly from the region the total ingestion of plants from the surroundings is about 12.2 g/day.

Dermal adsorption
For the estimation of the dermal adsorption UNIRISK makes also a difference between indoor and outdoor exposure. The outdoor exposure of children is assumed with a corporal surface of 0.21 m^2 and a cargo to the skin of 0.51 mg/cm^2 of the soil. It was calculated with 5 days/ week and 6 months/ year. For the indoor computation UNIRISK assumes a corporal surface of 0.05 m^2 and a cargo to the skin of 0.056 mg/cm^2 of the soil. Altogether it means for

the children an average exposure of 7.5 mg/day. The corporal surface of an adult for outdoor activities was assumed with 0.17 m^2, the cargo to the skin with 0.51 mg/cm^2 and the exposure time with 8 hours/ day and 5 months/ year. For the indoor situation a corporal surface of 0.091 m^2 and a cargo factor of 0.056 mg/cm^2 was considered. The total sum for the adsorption path for adults is 2.53 mg/day. For adults a penetration factor of 6 % for the indoor as well for the outdoor exposure was additionally considered.

2.4.4 IMPACT PATHWAY ANALYSIS (IPA)

The ExternE-Project (Externalities of Energy) started in 1991 as the European part of a collaborative study between the European Commission and the US Department of Energy. It was the first systematic approach to external cost evaluation of different energy generating process chains (fuel cycles), and consisted of two major phases, which are the followings (EC, 1995; ORNL/ REF (1995):

- Collaboration with the US Department of Energy to develop the conceptual approach and the methodology
- Application of the methodology to a wide range of different fuel cycles at reference sites in Europe and extension to energy use in transport and domestic sector

The major result of that work was a developed methodology, which provides a transparent basis on which different impacts, technologies and locations may be compared. It is also suitable for the evaluation of health and environmental damages with or without monetary valuation due to electricity production. It uses a “bottom-up” approach that is known as damage function or Impact Pathway Analysis. The Impact Pathway Analysis traces the passage of pollutants from the place where they are emitted to the endpoint, hence the receptor that is affected by them. The approach thus provides a logical and transparent way of quantifying externalities. The difference to the earlier used “top-down” methodologies (Hohmeyer, 1992; Friedrich & Voss, 1993) is that specific emission data are used for individual locations. These data are computed with pollution dispersion models, detailed information about receptors and dose-response and exposure-response functions in order to calculate the physical impact of increasing emissions. Finally the impacts are valuated economically. The principle steps are illustrated in Figure 2.23 and described as follows:

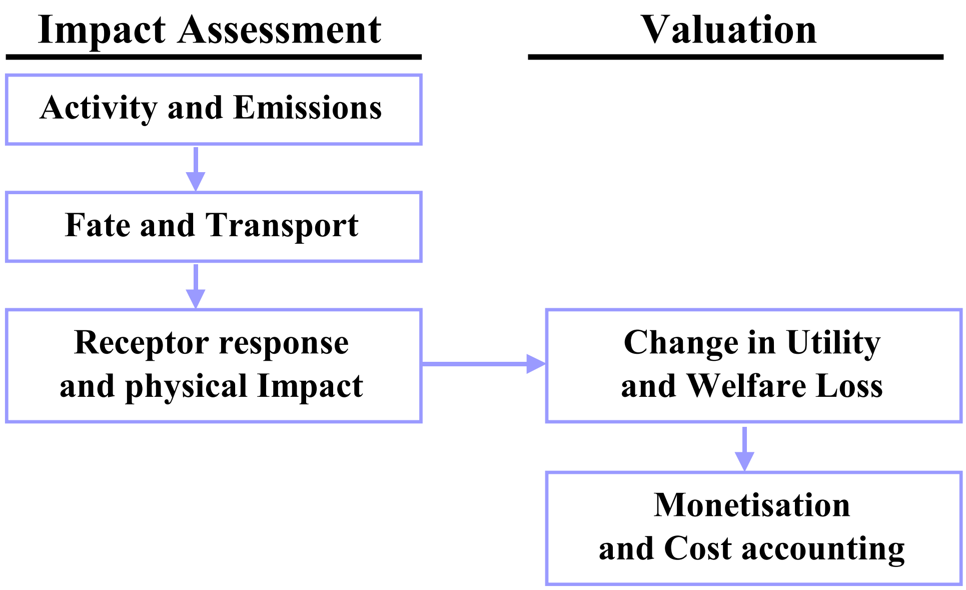


Figure 2.23: Illustration of the main steps of the IPA (EC, 1995)

- **Activity and Emission**: characterisation of the relevant technologies and the environmental burdens they cause (e.g. mg of SO_2 per Nm^3 emitted by the considered process)
- **Fate and Transport**: calculation of the increasing concentration in the effected regions via atmospheric dispersion models and chemical reactions (e.g. SO_2 transport and transformation into sulphates)
- **Receptor response and physical Impact**: characterisation of the receptors exposed to the incremental pollution, identification of suitable dose- response and exposure-response functions and their linkage to given estimated physical impacts (e.g. number of asthma cases due to increase of sulphates).
- **Monetisation and Cost accounting**: economic valuation of the mentioned impacts, determination of external costs, that have not been internalised by governmental regulations (e.g. multiplication of the monetary value with the asthma incidents gives the damage costs).

2.4.5 Comparison of Impact Pathway Analysis and Environmental Risk Assessment

In Figure 2.24 the steps of the Impact Pathway Analysis are compared to those in Environmental Risk Assessment. The difference of the IPA in comparison to the exposure Risk Assessment consists of the following elements:

- Simplified way to assess the environmental fate and exposure, in principle of emissions to air, by not including the multimedia modelling
- Inclusion of damages on a regional level due to pollutants with a long residence time
- Use of exposure-response functions based on epidemiological studies additionally to dose-response functions based on toxicological tests
- Expression of the effects in physical impact parameters such as cancer cases and restrictive activity days
- Valuation step, in principle by monetisation

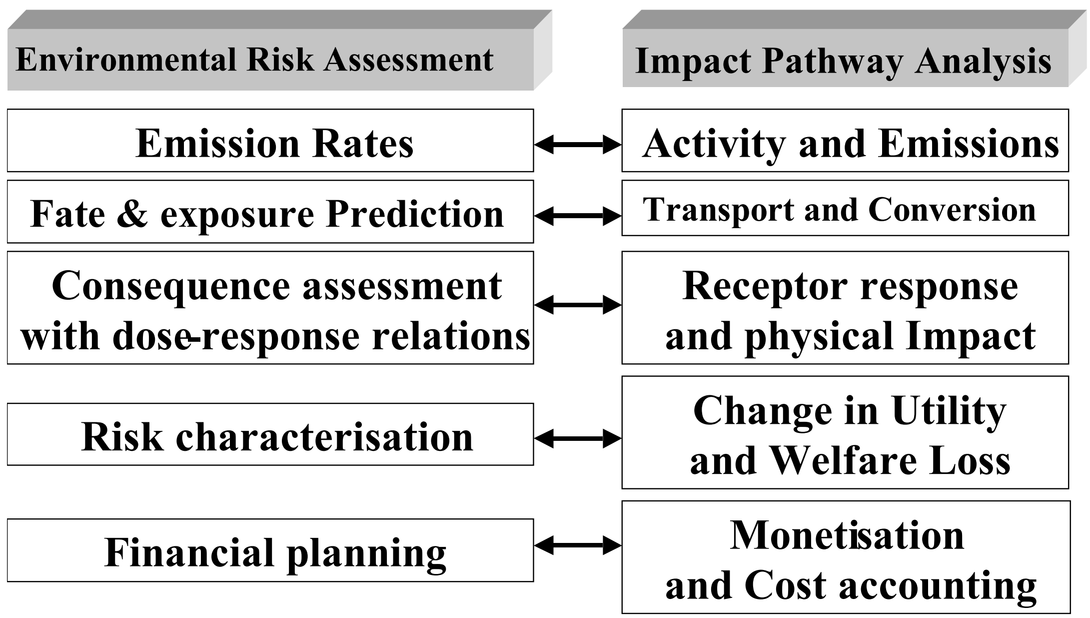


Figure 2.24: Steps of the Impact Pathway Analysis compared to those in Environmental Risk Assessment

2.4.6 INTEGRATED IMPACT ASSESSMENT MODELS

Although the IPA is a complex approach its application is quite easy thanks to the support by integrated impact assessment models like EcoSense developed by Krewitt et al. (1995) or PathWays (Rabl et al., 1998). In this study the EcoSense model was applied. Therefore, here further details are given about this model.

EcoSense stems from the experiences learned in the ExternE Project (EC, 1995) to support the assessment of priority impacts resulting from the exposure to airborne pollutants, namely impacts on health, crops, building materials, forests, and ecosystems. Although global warming is certainly among the priority impacts related to air pollution, EcoSense does not cover this impact category because of the very different mechanism and the global nature of impact. Priority impacts like occupational or public accidents are not included either because the quantification of impacts is based on the evaluation of statistics rather than on modelling.

Version 2.0 of EcoSense covers 13 pollutants, including the 'classical' pollutants SO_2, NOx, particulate matter and CO, as well as some of the most important heavy metals and hydrocarbons, but does not include impacts from radioactive nuclides.

In view of the increasing understanding of the major importance of long-range transboundary transport of airborne pollutants, also in the context of external costs from electricity generation, there was an obvious need for a harmonised European-wide database supporting the assessment of environmental impacts from air pollution. In the very beginning of the ExternE Project, work was focused on the assessment of local scale impacts, and teams from different countries made use of the data sources available in each country. Although many teams spent a considerable amount of time compiling data on e.g. population distribution, land use etc., it was realised that country specific data sources and grid systems were hardly compatible when the analysis had to be on an European scale. So it was logical to set up a common European-wide database by using official sources like EUROSTAT and make it available to all ExternE teams. Once there was a common database, the consequent next step was to establish a link between the database and all the models required for the assessment of external costs to guarantee a harmonised and standardised implementation of the theoretical methodological framework. (EC, 1995)

Taking into account this background, the further objectives for the development of the EcoSense model were:

- To provide a tool supporting a standardised calculation of fuel cycle externalities,
- To integrate relevant models into a single system,
- To provide a comprehensive set of relevant input data for the whole of Europe,
- To enable the transparent presentation of intermediate and final results, and
- To support easy modification of assumptions for sensitivity analysis.

As health and environmental impact assessment is a field of large uncertainties and incomplete, but rapidly growing understanding of the physical, chemical and biological mechanisms of action, it was a crucial requirement for the development of the EcoSense system to allow an easy integration of new scientific findings into the system. As a consequence, all the calculation modules (except for the integrated ISCST-model) are designed in a way that they are a model-interpreter rather than a model. Model specifications like e.g. chemical equations, exposure-response functions or monetary values are stored in the database (Paradox format) and can be modified by the user. This concept should allow an easy modification of model parameters, and at the same time the model should not necessarily appear as a black box, as the user can trace back what the system is actually doing.

Figure 2.25 shows the modular structure of the EcoSense model. All data - input data, intermediate and final results - are stored in a relational database system. The two air quality models integrated in EcoSense (ISCST-2 and the Windrose Trajectory Model – WTM) are stand-alone models, which are linked to the system by pre- and postprocessors. There are individual executable programs for each of the impact pathways, which make use of common libraries. The following sections give a more detailed description of the different EcoSense modules.

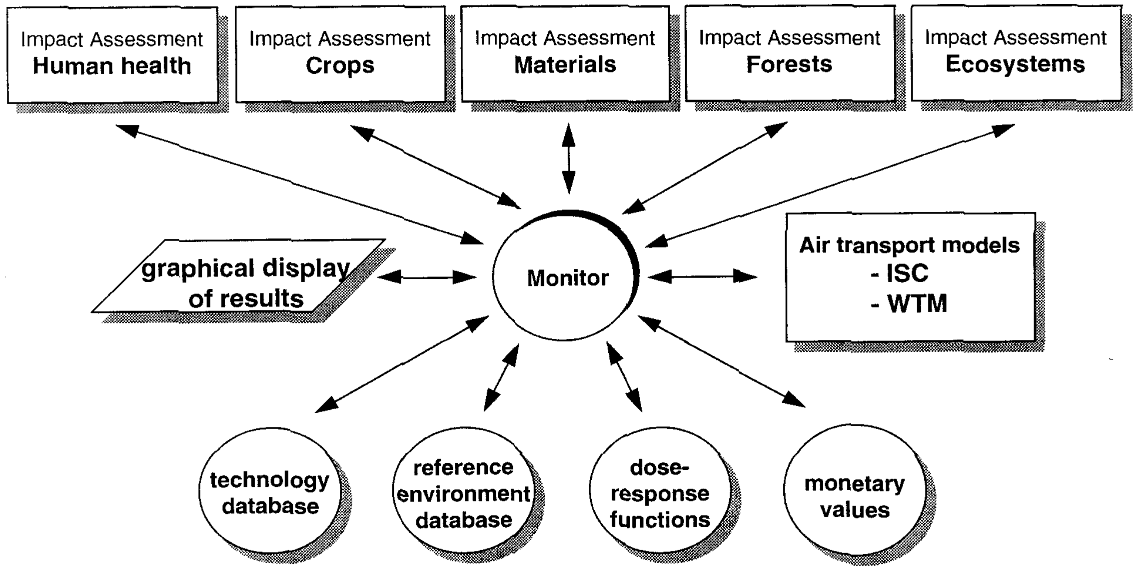


Figure 2.25: Structure of the EcoSense model (EC, 1995)

The EcoSense modules are described as follows:

Reference Technology Database
The reference technology database holds a small set of technical data describing the emission source (power plant) that are mainly related to air quality modelling, including e.g. emission factors, flue gas characteristics, stack geometry and the geographic co-ordinates of the site.

Reference Environment Database
The reference environment database is the core element of the EcoSense database, providing data on the distribution of receptors, meteorology as well as a European wide emission inventory. All geographical information is organised using the EUROGRID co-ordinate system, which defines equal-area projection grid cells of 10 000 km² and 100 km² (Bonnefous & Despres, 1989), covering all EU and European non-EU countries. Data on population distribution and crop production are taken from the EUROSTAT REGIO database, which in some few cases have been updated using information from national statistics. The material inventories are quantified in terms of the exposed material area from estimates of representative buildings. Critical load maps for nitrogen deposition are available for nine classes of different ecosystems, ranging from Mediterranean scrub over alpine meadows to tundra areas. To simplify access to the receptor data, an interface presents all data according to administrative units (e.g. country, state) following the EUROSTAT NUTS classification scheme. The system automatically transfers data between the grid system and the respective administrative units. In addition to the receptor data, the reference environment database

provides elevation data for the whole of Europe on the 10 km * 10 km grid, which are required to run the Gaussian plume model. Meteorological data (precipitation, wind speed and wind direction) and a European-wide emission inventory for SO_2, NOx and NH_3 from EMEP (Sandness & Styve, 1990), transferred to the EUROGRID-format, are also included for the long-range pollutant transport model. Hourly meteorological data for year have to be added for each specific site to run the incorporated ISCST-model for short-range pollutant transport.

Dose-Response and Exposure-Response Functions

Using an interactive interface, the user can define any exposure effect model as a mathematical equation. The user-defined function is stored as a string in the database, which is interpreted by the respective impact assessment module at runtime. All dose-response and exposure-response functions compiled by the various experts of the ExternE Project are stored in the database.

Monetary Values

The database provides monetary values for most of the impact categories following the recommendations of the ExternE economic valuation task group according of guidelines from the European Commission.

Impact Assessment Modules

The impact assessment modules calculate the physical impacts and - as far as possible - the resulting damage costs by applying the dose-response and exposure-response functions selected by the user to each individual grid-cell, taking into account the information on receptor distribution and concentration levels of air pollutants from the reference environment database. The assessment modules support the detailed step-by-step analysis for a single endpoint as well as a more automated analysis including a range of pre-specified impact categories.

Presentation of Results

Input data as well as intermediate results can be presented on several steps of the Impact Pathway Analysis in either numerical or graphical format. Geographical information like population distribution or concentration of pollutants can be presented as maps. EcoSense generates a formatted report with a detailed documentation of the final results that can be imported into a spreadsheet programme.

Air Quality Models

A special feature of EcoSense is the fact that air quality models are included. Apart from the local-scale ISCST-2 model described in section 2.4.3, for which a set of site-specific meteorological data has to be added by the user, a long-range pollutant transport model is included, which has also been applied separately in this study for the calculation of site-dependent impact factors.

With increasing distance from the stack the plume spreads vertically and horizontally due to atmospheric turbulence. Outside the area of the local analysis (i.e. at distances beyond 100 km from the stack), it can be assumed for most purposes on the one hand that the pollutants have vertically been mixed throughout the height of the mixing layer of the atmosphere; and on the other hand, chemical transformations can no longer be neglected on a regional scale. The most economic way to assess annual, regional scale pollution is a model with a simple representation of transport and a detailed enough representation of chemical reactions.

The Windrose Trajectory Model (W'TM) used in EcoSense to estimate the concentration and deposition of acid species on a regional scale was originally developed at Harwell Laboratory by Derwent & Nodop (1986) for atmospheric nitrogen species, and extended to include sulphur-species by Derwent et al. (1988). The model is a receptor-orientated Lagrangian plume model employing an air parcel with a constant mixing height of 800 m moving with a representative wind speed. The results are obtained at each receptor point by considering the arrival of 24 trajectories weighted by the frequency of the wind in each 15° sector. The trajectory paths are assumed to be along straight lines and are started at 96 hours from the receptor point. In addition to dealing with primary pollutants the WTM is also able to calculate concentrations of secondary sulphate and nitrate aerosols formed from emissions of SO_2 and NOx, respectively. The chemical reaction schemes implemented in the model are shown in Figure 2.26.

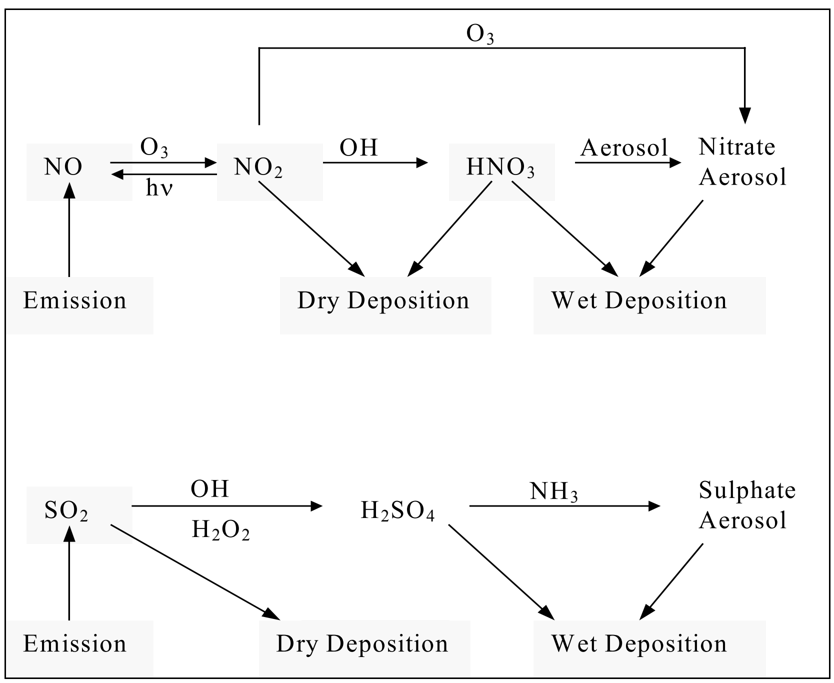


Figure 2.26: Chemical scheme in WTM (EC, 1995)

The new version of EcoSense (3.0) also includes a photochemical ozone creation model and corresponding impact assessment information (Hertwich et al., 2000).

2.5 Industrial Process Chains: Differentiation of the life cycle type

Looking at the presented picture of process chain- and site-orientated environmental impact assessment methods it seems that it is necessary to come to a spatial differentiation of life cycles in order to facilitate a more accurate way of environmental damage estimations in a chain-perspective. In this way the poor accordance between impact potentials and actual impacts can be overcome in LCIA as shown broadly by Potting (2000). This is site-dependent impact assessment and will be further studied in chapter 6.

However, another approach might help, too, to overcome the above-mentioned inconsistency. This is to differentiate the life cycles also according to their number of processes considered. That means to use different levels of sophistication for different applications that here are defined by their chain length.

It seems to be unfeasible to estimate environmental damages for each process of a full LCA, i.e. of a complex product system with a huge number of industrial processes, e.g. computers, because all the local or regional information is not accessible and each process is contributing only marginally to the total environmental impact. Such a life cycle is illustrated in Figure 2.27.

However, if the LCA methodology is applied to industrial process chains, i.e. chains with a small number of industrial processes (let us say < 100 processes), e.g. waste treatment process chains, the localisation of the processes is often known and, moreover, in general only a small number of processes is responsible for the main part of the environmental impact, as can be seen in the example of Figure 2.28. Hence, for such applications of LCA the main individual processes can be assessed in its corresponding surroundings.

This differentiation of the life cycle type according to the chain-length is crucial for the way environmental damages can be estimated in the most accurate way possible. This work will focus on the methodology development for damage estimations in industrial process chains, here defined as life cycles with a relative small number of processes involved, in contrast to product systems, i.e. process chains with a high number of different sites to consider. In chapter 5 a comprehensive methodology for such life cycle types is presented. It is evident that in these cases different levels of detail in the impact assessment can be used.

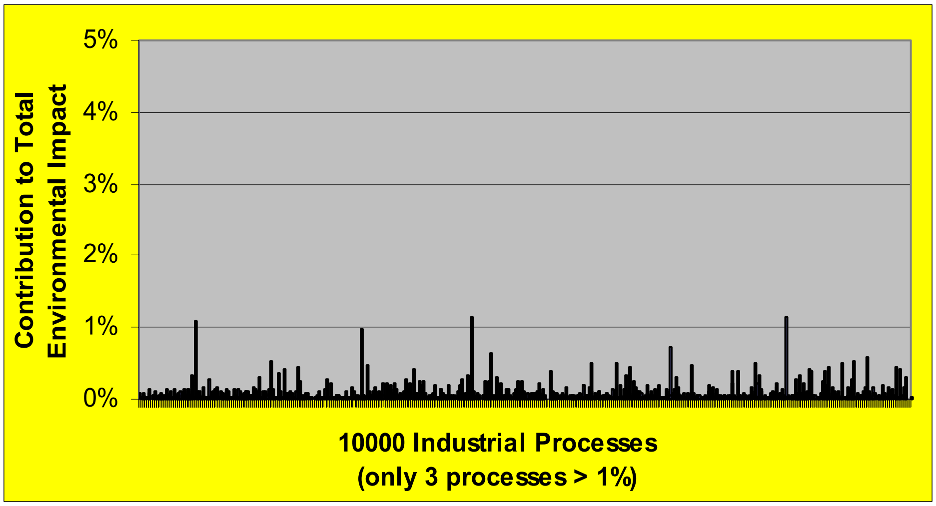


Figure 2.27: Full LCA studies of complex products - with huge number of industrial processes, e.g. computer. In the example less than 3 processes contribute with more than 1 % to the total environmental impact.

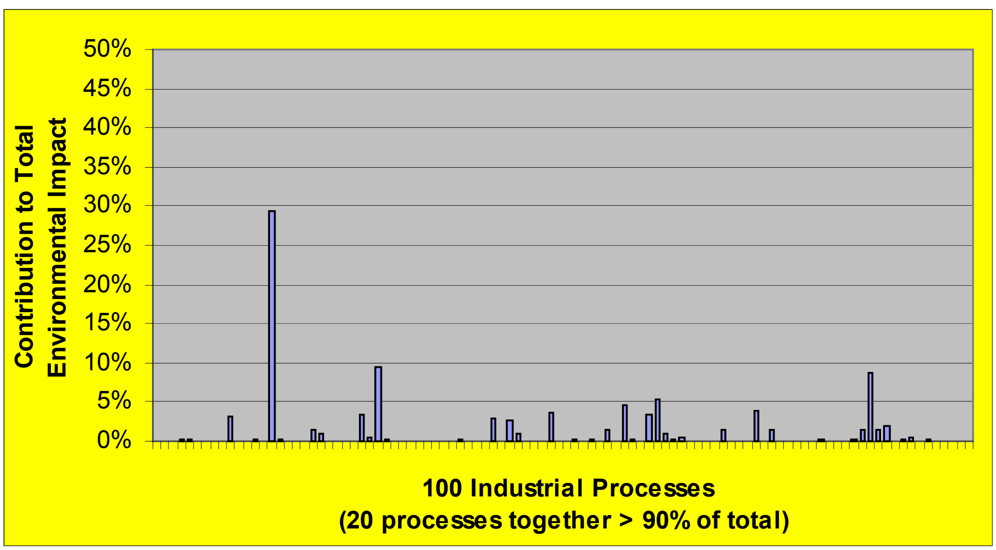


Figure 2.28: LCA methodology applied to industrial process chains - with small number of industrial processes, e.g. waste treatment. In the example 20 processes together contribute to more than 90 % of the total environmental impact.

3 PRESENTATION OF THE CASE STUDY ON WASTE INCINERATION WITH THE RESULTS OF COMMONLY USED METHODS

The Municipal Solid Waste Incinerator (MSWI) plant of Tarragona (SIRUSA) operates since 1991. It is located in the industrial polygon area, approximately three kilometres outside of Tarragona. The main roads, such as the motorway A-7 provide excellent access to the plant. SIRUSA is a public incineration plant and it is operated and maintained by the public funds of the surrounding cities: Tarragona, Reus, Valls, Salou, Cambrils, Vila-Seca and Constantí. In 1997 an advanced acid gas removal system was installed. Hence, two situations were studied: the operation of the plant during 1996 (situation 1) and the current operation with the advanced acid gas removal system working (situation 2).

3.1 TECHNICAL DATA OF THE PLANT AND THE PROCESS CHAIN

The incinerator has parallel grate-fired furnaces with primary and secondary chambers. The combustion process is based on Deutsche Babcock Anlagen technology. Each of the furnaces has a capacity of 9.6 tons per hour, which makes approximately 460 tons daily incineration capacity of municipal waste. The temperature in the first combustion chamber varies between 950ºC and 1000ºC. In the secondary, post-combustion chamber the temperature is 650-720ºC and the output temperature of the flue gas is 230-250ºC. The minimum incineration conditions are two seconds incineration time at 850ºC with 6% minimum oxygen excess. The combustion process is controlled by on-line measurements (CO_2, O_2) and visually with the help of TV-monitors (Nadal, 1999). The process generates electricity of the steam at a rate of 44.8 tons per hour. About 80 % of the total electricity produced is sold and 20 % is used for the operation. The scrap metal is collected separately and iron is recycled (STQ, 1998). The incinerated residues are only solids. The average composition of the municipal waste is shown in Table 3.1. A schematic overview of the plant is given in Figure 3.1.

Table 3.1: Waste composition of the MSWI plant in Tarragona (average 1999)

Component	Percentage
Organics	46%
Paper and cardboard	21%
Plastics	13%
Glass	9%
Metals	3%
Ceramics	2%
Soil	1%
Others	5%
Total	*100%*

The flue gas cleaning process is a semi-dry process consisting of an absorber of Danish technology (GSA). The acid compounds of the flue gas, such as HCl, HF, SO_2 are neutralized with lime, $Ca(OH)_2$. The reaction products are separated in a cyclone and after that the gases are treated with injected active carbon to reduce dioxin and furan concentrations. The last cleaning step, a bag filter house, secures that the total emissions meet the legislative emission limits; Spanish RD 1088/92 Directive and also a regional Catalan Directive 323/1994 which is an improved version of the European 89/369/CEE Directive.

The total emissions and other process data are presented in Table 3.2. The emissions are also controlled by the local authorities (Delegació Territorial del Departament de Medi Ambient de la Generalitat de Catalunya). Hence, the plant is under continuous, real-time control, which guarantees independent information on the emissions.

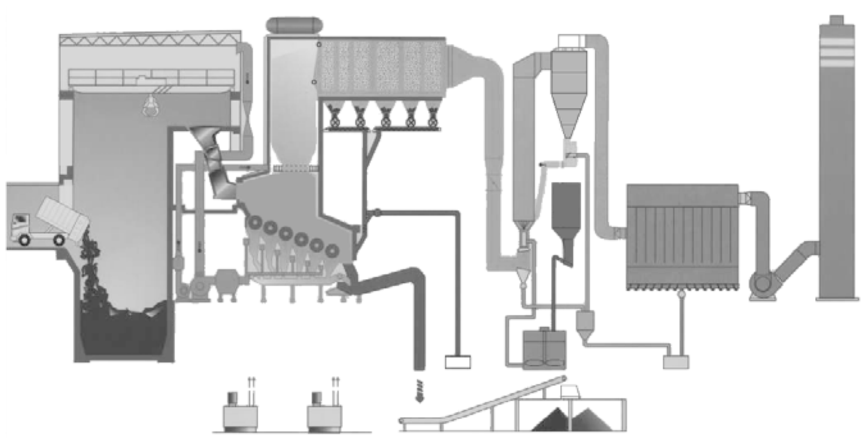


Figure 3.1: Scheme of the MSWI plant in Tarragona (Nadal, 1999)

Table 3.2: Overview of data from the MSWI plant in Tarragona

Situation	"Without new filters"	"With new filters"
Alternative no.	1	2
Production data		
Produced electricity (MW)	6	6
Electricity sent out (MW)	5.2	4.9
Working hours per year (h)	8,280	8,280
Emission data		
CO_2* (g/Nm3)	186	186
CO (mg/Nm^3)	40	40
HCl (mg/Nm^3)	516	32.8
HF (mg/Nm^3)	1.75	0.45
NOx (mg/Nm^3)	191	191
Particles (mg/Nm^3)	27.4	4.8
SO_2 (mg/Nm^3)	80.9	30.2
As (µg/Nm^3)	20	5.6
Cd (µg/Nm^3)	20	6.6
Heavy metals** (µg/Nm^3)	450	91
Ni (µg/Nm^3)	30	8.4
PCDD/Fs (ng/Nm^3) as toxicity equivalent (TEQ)	2	0.002
Materials		
IN		
CaO (t/a)	0	921
Cement (t/a)	88.5	518
Diesel	148.8	148.8
OUT		
Slag (t/a)	42,208	42,208
Scrap for treatment (t/a)	2,740	2,740
Ashes for treatment (t/a)	590	3,450
Ashes for disposal (t/a)	767	4,485
Plant data		
Gas volume (Nm^3/h)	90,000	
Gas temperature (K)	503	
Stack height (m)	50	
Stack diameter*** (m)	1.98	
Latitude (º)****	41.19	
Longitude (º)****	1.211	
Terrain elevation (m)	90	

* Corresponds to the measured value, not to the adjusted one used in the LCA study.

** Heavy metals is a sum parameter in form of Pb equivalents of the following heavy metals (As, B, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Sb). Cd is considered apart for its toxic, As and Ni for its carcinogenic relevance.

*** In reality there are two stacks with 1.4 m, but due the limitations of the dispersion models used one stack with a diameter of 1,98 was considered.

**** Initially the data was in UTM, the Mercator transversal projection. The conversion was made using the algorithm in http://www.dwap.co.uk/welcome.

Sub-processes that are important for the process chain and which were taken into account with their site-specific data are mentioned below. An overview of the process chain is given in Figure 3.2.

TRI S.A. is a treatment company for industrial wastes situated in the industrial zone of Constantí, somehow in the suburbs of Tarragona. In this plant, the ashes from the MSWI SIRUSA are treated. At TRI S.A. incineration ashes are mixed with cement to inert the material so that it can be deposed on a sanitary landfill as ‘normal’ waste afterwards. The inert product fulfils the lixiviation criteria for sanitary landfills. In general, landfill takes place in the region of Tarragona.

The company LYRSA, situated next to Madrid, treats the scraps, which are extracted from the incineration slag at SIRUSA by magnetic separation. These scraps are transformed into iron, which can be used in the metal and steel industry.

The Lime (CaO) used for the neutralization of the acid compounds in the flue gas comes from UNILAND in S[ta] Margarida i els Monjos/ Penedes near Barcelona. Also the cement that is used for the ash treatment at TRI S.A. is produced there.

For the transports it is supposed that a 16-ton truck is used. Moreover, it is assumed that the electricity consumed by SIRUSA is the Spanish average electricity mix. Finally, diesel and lubricant oil production in Europe is also part of the process chain.

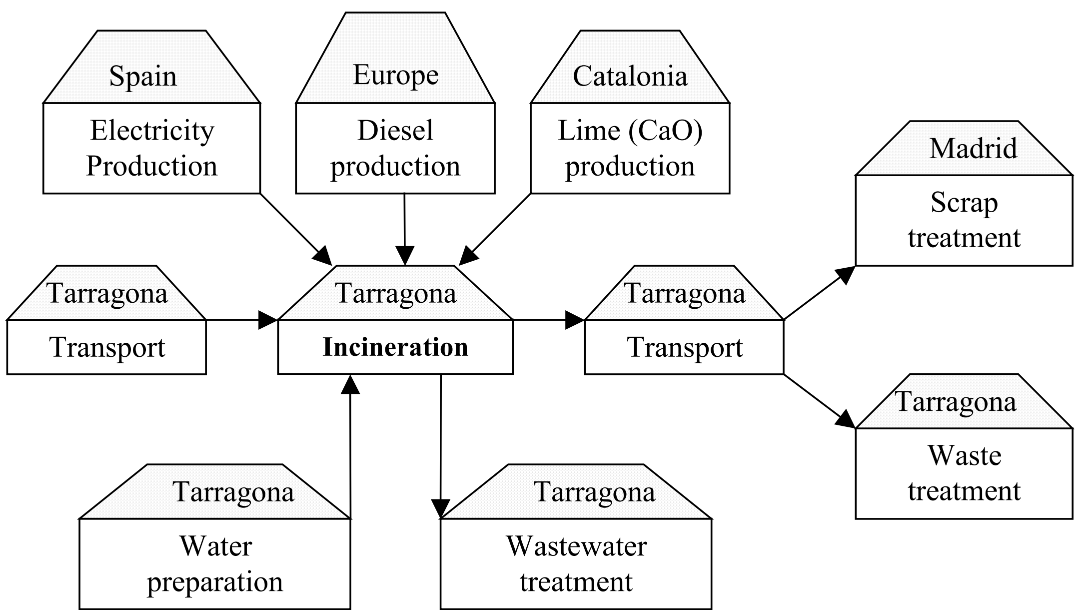


Figure 3.2: Overview of the MSWI process chain

3.2 Modular Municipal Solid Waste Incinerator (MSWI) model

On the basis of the model described by Kremer et al. (1998) and a spreadsheet version by Ciroth (1998) a modular steady-state process model with several enhancements has been created by Hagelüken (2001), in cooperation with the Environmental Management and Analysis (AGA) Group of the Universitat Rovira i Virgili. The MS-Excel based model takes into account the elementary waste input composition and the important plant data, such as plant layout and process specific constants.

3.2.1 Structure of the model

In the model, the steam generator consists of grate firing and heat recovery system, and a regenerative air pre-heater. Energy production is calculated using the heating value of the waste input and the state points of the steam utilisation process. For the macroelements (C, H, N, O, S and Cl, F), the flue gas composition is determined by simple thermodynamic calculation of the combustion, taking excess air into account. The heavy metals, however, are calculated on the basis of transfer coefficients (Kremer et al., 1998). Emissions of CO and TOC depend on the amount of flue gas. For the emissions of NO_x and PCDD/Fs, empirical formulas are used. Due to the fact that acid forming substances like S, Cl, and F are partly absorbed by basic ash components, the total amount of SO_2, HCl, and HF in the flue gas is reduced respectively. The flue gas purification consists of an electrostatic precipitator, a two-stage gas scrubber for the removal of acid gases (using NaOH and $CaCO_3$ for neutralization), a denitrogenation unit (DeNOx with selective catalytic reduction using NH_3) and an entrained flow absorber with active carbon injection for the removal of dioxins and heavy metals. The plant is of the semi-dry type; all wastewater is evaporated in a spray dryer after the heat exchanger.

The processes and calculations are distributed to several MS-Excel workbooks. The processes represented by the workbook files are linked together by their input/ output sheets. The division into workbooks and their major dependencies are shown in Figure 3.3.

3.2.2 Scenario

Based on the modular model a scenario was created of a MSWI similar to the current plant in Tarragona, but with DeNOx as an additional gas cleaning system. An overview of the calculated inputs and outputs for the SIRUSA waste incineration plant is given in Table 3.3. All the calculated emissions are lower as the current situation 2 of SIRUSA. Also the corresponding transport distances are presented, because they are necessary for the estimation of environmental damages for industrial process chains in chapter 6.

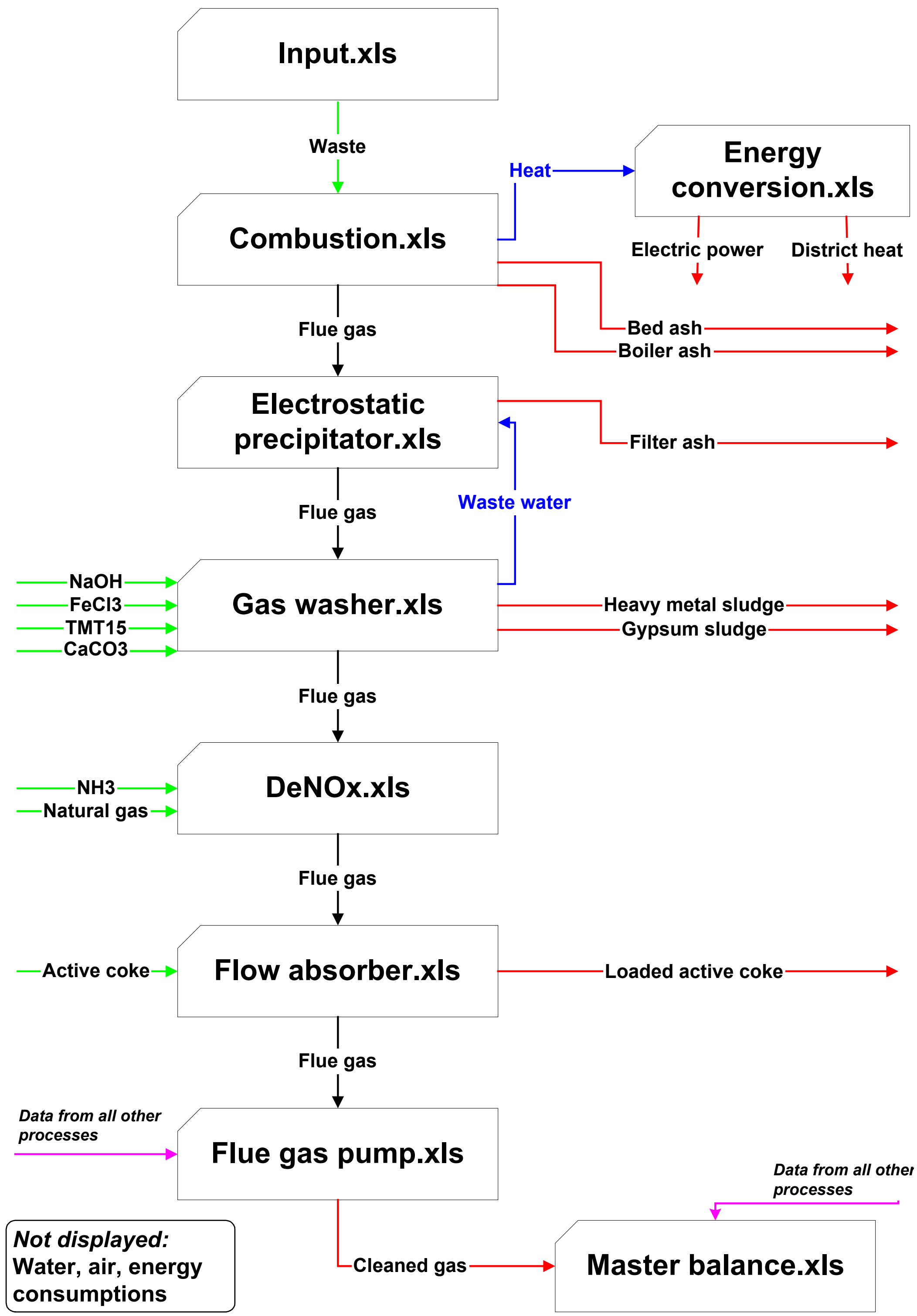


Figure 3.3: Workbook structure of the modular spreadsheet model (Hagelüken, 2001). Major dependencies, some energy and material flows are also shown.

Table 3.3: Overview of inputs and outputs of the scenario

	Scenario
Alternative no.	3
Overall transport [tkm/a]	
Municipal waste	2,762,406
Slag	1,032,278
CaO	224,080
Ammonia	6,859
Scrap treatment	2,236,524
Ash treatment	71,497
Ash disposal	2,439,849
Cement	134,058
Electricity [TJ/a]	
Consumption	18.41
Production	290.11
Materials [t/a]	
IN	
CaO	2,241
Ammonia	686
Cement	1,341
OUT	
Slag	32,259
Scrap for treatment	2,094
Ashes for treatment	8,937
Ashes for disposal	11,618
Emissions	
Flue gas [Nm^3/h]	96,000
As [(µg / Nm^3]	0.035
Cd [(µg / Nm^3]	1.50
Ni [(µg / Nm^3]	0.51
NOx [mg/ Nm^3]	58
PM10 [mg/ Nm^3]	0.45
SO_2 [mg/ Nm^3]	6.9

3.3 LCA OF THE MSWI PROCESS CHAIN

In 1997 the company SIRUSA who is in charge of the municipal urban waste incineration in the district of Tarragona/ Spain collaborated with the Universitat Rovira i Virgili (URV) in order to carry out an environmental impact assessment of the electricity produced by its municipal waste incinerator (STQ, 1998). The final goal was to evaluate the improvement in the environmental performance obtained by the installation of an advanced acid gas removal system. The study was to be based on the LCA methodology.

In order to carry out the computation for the LCI analysis and the impact assessment, a special Excel based programme has been used that was developed according to the eco-vector approach developed by Castells et al. (1994) (see section 2.3.2). The results obtained by this model have been compared with the results obtained when using TEAM as done by Schäfer (2000).

The LCIA is performed according to the Eco-indicator 95. The results obtained with the Eco-indicator 95 have been compared with other LCIA methods by Sonnemann (1998).

3.3.1 Excel-based LCA study with Eco-indicator 95 as LCIA method

The LCA study (STQ, 1998) was developed on the basis of the "Code of Practice" (SETAC, 1993) and according to the steps in the ISO 14040 (1997). The Inventory was based on provider's information, literature data on raw materials and a detailed analysis of the incineration process.

The objective of the study was to identify, evaluate and compare the environmental loads derived from the electricity production by the municipal waste incinerator of Tarragona in the two situations described above in order to analyse the environmental efficiency of the new investment.

In the following the project is described according to the points indicated in ISO 14040 (1997).

The function of the incineration process is to reduce the volume and the toxicity of the municipal waste treated. The production of electric energy has to be seen as an added value to the incineration process. As the objective of the study is the analysis of the electric energy generated in the incineration process, the functional unity selected is "TJ of produced electricity".

The study comprises all the processes from the municipal waste disposal in containers to the landfill of the final waste, as shown in Figure 3.4. Consequently, the following processes are considered: transport of the municipal waste to the incinerator, combustion, gas treatment and ashes removal as well as slag disposal (including their transport to the final localisation). The final step with its emissions associated to the landfill is not analysed.

The incineration process has been divided into subsystems that are the followings:

- Waste incineration plant with combustion process itself and including gas treatment
- Water treatment (treatment process applied in the ash bath, demineralisation process applied in the kettles by osmosis and refrigeration process by the means of a tower)
- Ash treatment (ionic ashes and waste from the gas treatment filters)
- Scrap treatment (Iron waste recycling process)

The latter were analysed as processes that generate a utile product for the incineration process. The environmental loads are associated to the consumed product by the principal process (e.g. water) or the treated product (e.g. municipal solid waste). For the raw material production and the transport process (truck of 16 t) literature values have been used.

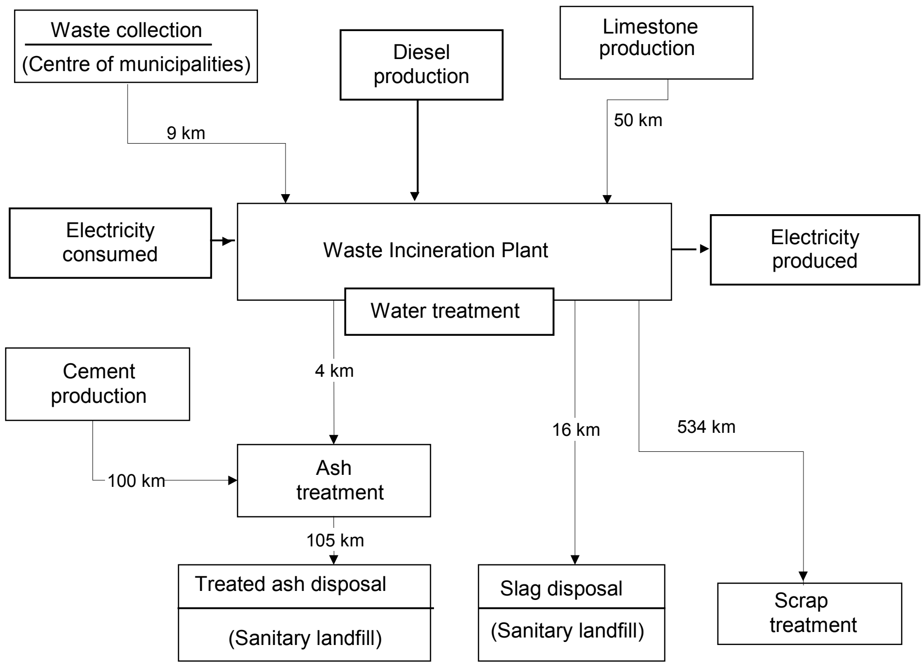


Figure 3.4: Processes considered in the LCA study within the boundaries of the system (including transport distances)

Allocation Procedures and Impact Assessment

Due to the study's objective, the electricity produced is considered to be the only useful measurable product to which all environmental loads are assigned, For the impact assessment the Eco-indicator 95 method (Goedkoop, 1995) is applied. The method is based on the impact categories described by CML (Heijungs et al., 1992).

Data requirements and quality

In order to carry out the study, three types of data were used:

- Literature data of environmental loads for the raw material used in the analysed processes. The considered source has been the report referred as ETH 1996 (Frischknecht et al., 1996). The exactness of the report and the agreement of this data with the particular situation in Spain determine the data quality.
- Real data of consumption and emissions associated to the incineration process, average values from 1996 (situation 1) and average values for two months with the advanced acid

gas removal system in operation (situation 2). The data quality can be considered as reliable because they have been obtained directly form the process.

- Real data of consumption and emissions associated to the waste treatment processes, obtained by visits and questionnaires answered by the treatment companies. The reliability of the data delivered by the companies depends on the available information and the degree of collaboration.

Assumptions and Limitations

The principal assumptions made are the followings:

- The incineration plant operates during 345 days a year and 24 hours each day.
- For the analysis of the environmental load associated to the transport the journey to the destination and the return of the truck is considered, assuming the same environmental load for the empty truck as for the full one.
- The internal consumption of electricity is covered by the properly produced electric energy. Only in cases of an operation stop the importation of electricity is necessary. By this, the environmental load associated to the electric energy consumed during the process is the result of the total inventory analysis (need of iterative computation).
- In the case of the scrap-metal treatment, the generated utile product has been classified as iron within the raw materials.

The main limitations of the study are the followings:

- There is neither an analysis nor a total characterisation of the municipal waste entering the system.
- The environmental load associated to the emissions of the final waste disposal in a landfill has not been considered.
- It was not possible to simulate the following products used in the incineration process because of an information lack in the databases consulted: ferric chloride, active carbon and additives used in the osmosis process.
- Due to the limitation of the impact assessment method, priority impacts have been evaluated but not all types of possible impacts.

Results of the Inventory analysis and the Impact assessment

The results for the Life Cycle Inventory (Table 3.4) are:

Raw material consumption
From the 14 analysed parameters, that are considered according to (Frischnkecht et al., 1996), the current situation is unfavourable for all of them due to the higher consumption of raw materials, especially cement (for the higher waste quantity per produced TJ) and CaO (for the advanced gas treatment), and more transport activity because of the higher raw material consumption and waste quantity.

Energy consumption
The current situation is unfavourable due to the higher energetic consumption per produced TJ because of the additional energy consumption of the advanced gas treatment system.

Air emissions
Form the 37 analysed parameters the current situation is unfavourable for all of them except of 9, which are As, Cd, PCDD/Fs dust, HCl, HF, Ni, SO_2 and other heavy metals. Those are basically the parameters reduced by the operation of the advanced gas treatment system.

Water emissions
Form the 23 analysed parameters the current situation is unfavourable for all of them except of 4, which are BOD, COD, Cd and Hg. Those are basically the parameters reduced by the current operation that works without water emissions into the sewage.

The results of the impact assessment are shown in Table 3.5:

Situation 2, the current operation of the incineration plant after the installation of an advanced gas treatment system, has associated a higher Global Warming and Nutrification Potential than Situation 1, i.e. the former operation, because it has a higher CO_2 and NOx emission per produced TJ due to the additional energy consumption for the advanced gas treatment system. The Situation 2 has also a higher Ozone Depletion and Photochemical Ozone Creation Potential caused by the higher contribution of the transport.

Situation 2 is more favourable than Situation 1 in the case of the Acidification Potential and the Winter Smog (SO_2 equiv.) due to the reduction of HCl, SO_2 and dust. The Situation 2 is also favourable for the Heavy metals (Pb equiv.) and Carcinogen substances (PAH eq.) because they are both removed by the advanced gas treatment system.

The global environmental evaluation according the Eco-indicator 95 is positive for the installation of the advanced gas treatment system. The method assigns especially high weightings to those impacts that are reduced by the advanced gas treatment system (mainly acidification and heavy metals).

Hence, it can be concluded that the installation of an advanced gas treatment decreases the stack emissions and the related inventory and impact assessment data, but it increases the majority of the other environmental loads considered, due to higher raw material and energy consumption per produced TJ as well as more transport activity. Nevertheless, the overall environmental efficiency measured according Eco-indicator 95 clearly improves.

Table 3.4: Results of the LCI analysis

EMISSIONS TO AIR	**Unit**	**Electricity Sit. 1* TJ**	**Electricity Sit. 2** TJ**	**Diff.#**	**EMISSIONS TO WATER**	**Unit**	**Electricity Sit. 1* TJ**	**Electricity Sit. 2** TJ**	**Diff.#**
Aldehydes	kg	4,54E-05	5,92E-05	-30%	AOX	kg	1,15E-03	1,36E-03	-18%
Ammonia	kg	2,82E-02	3,70E-02	-31%	Aromatics	kg	1,73E-01	2,06E-01	-19%
As	kg	1,17E-01	3,96E-02	66%	As	kg	5,13E-03	1,04E-02	-103%
Benzene	kg	1,32E-01	1,71E-01	-30%	B	kg	1,17E-02	1,51E-02	-29%
Benzo(a)pyrene	kg	4,94E-05	5,97E-05	-21%	Ba	kg	9,27E-01	1,27E+00	-37%
Cd	kg	1,13E-01	3,98E-02	65%	BOD	kg	5,07E+00	8,93E-02	98%
CO	kg	2,75E+02	3,01E+02	-9%	Cd	kg	9,30E-03	8,17E-03	12%
CO_2	kg	2,33E+05	2,57E+05	-10%	COD	kg	1,51E+01	1,37E+00	91%
CxHy aromatic	kg	9,22E-04	1,28E-03	-39%	Cr	kg	3,03E-02	5,48E-02	-81%
Dicloromethane	kg	5,00E-05	5,40E-05	-8%	Cu	kg	1,39E-02	2,73E-02	-96%
Dust	kg	1,71E+02	1,61E+02	6%	Dissolved subst.	kg	1,01E+00	2,14E+00	-112%
Ethanol	kg	2,60E-03	3,56E-03	-37%	Hg	kg	7,14E-05	2,73E-05	62%
Ethene	kg	2,10E+00	2,55E+00	-21%	Mn	kg	7,00E-02	1,28E-01	-83%
Ethylbenzene	kg	1,47E-02	1,78E-02	-21%	Mo	kg	8,40E-03	1,64E-02	-95%
Formaldehyde	kg	9,10E-03	1,61E-02	-77%	NH_3	kg	5,23E-01	6,13E-01	-17%
H_2S	kg	1,85E-02	2,91E-02	-57%	Ni	kg	1,34E-02	2,69E-02	-101%
Halon-1301	kg	2,25E-03	2,66E-03	-18%	Nitrates	kg	6,75E-01	8,06E-01	-19%
HCl	kg	2,89E+03	1,95E+02	93%	Pb	kg	2,56E-02	3,55E-02	-39%
Heavy Metals	kg	2,65E+00	7,17E-01	73%	Phosphate	kg	1,51E-01	3,10E-01	-105%
HF	kg	9,93E+00	2,80E+00	72%	Sb	kg	4,43E-05	7,10E-05	-60%
Methane	kg	3,32E+01	4,92E+01	-48%	SO^{4-}	kg	2,31E+01	3,74E+01	-62%
N_2O	kg	1,71E+00	2,06E+00	-20%	Unsolved subst.	kg	2,68E+01	2,94E+01	-10%
Ni	kg	1,80E-01	6,51E-02	64%	TOC	kg	3,22E+01	3,82E+01	-19%
Non methane VOC	kg	6,98E+01	8,31E+01	-19%	**RAW MATERIAL**	**Unit**	**Electricity Sit. 1* TJ**	**Electricity Sit. 2** TJ**	**Diff.#**
NOx	kg	1,23E+03	1,33E+03	-8%	Bauxite (ore)	kg	3,59E+01	4,27E+01	-19%
PAH	kg	7,37E-04	1,02E-03	-38%	Clay	kg	2,12E+02	1,01E+03	-376%
Phenol	kg	7,39E-06	9,93E-06	-34%	Coal	kg	1,50E+03	3,15E+03	-110%
Phosphate	kg	1,94E-03	3,88E-03	-100%	Copper (ore)	kg	2,71E+00	3,36E+00	-24%
Pentane	kg	5,51E-01	6,54E-01	-19%	Iron (ore)	kg	-1,32E+04	-1,38E+04	-5%
Propane	kg	4,35E-01	5,57E-01	-28%	Lignite	kg	6,56E+02	8,86E+02	-35%
Propene	kg	3,07E-02	4,49E-02	-46%	Limestone (ore)	kg	1,03E+03	1,62E+04	-1473%
SO_2	kg	5,08E+02	2,56E+02	50%	Natural gas	Nm^3	1,06E+02	5,48E+02	-417%
TCDD-Equiv.	ng	1,12E+07	1,43E+04	100%	Nickel (ore)	kg	1,00E+00	1,19E+00	-19%
Tetracloromethane	kg	2,55E-05	3,63E-05	-42%	Oil	kg	5,79E+03	6,86E+03	-18%
Toluene	kg	6,68E-02	8,39E-02	-26%	Silica (ore)	kg	8,74E+03	1,04E+04	-19%
Tricloromethane	kg	1,93E-06	3,07E-06	-59%	Uranium (ore)	kg	5,69E-02	7,43E-02	-31%
Vinylchloride	kg	1,19E-05	1,89E-05	-59%	Water	kg	3,57E+05	6,41E+05	-80%
					Wood	kg	3,35E+01	4,99E+01	-49%

Table 3.5: Differences in the impact categories and according to the Eco-indicator 95

	GWP	**ODP**	**POCP**	**NP**	**AP**	**Pb equiv.**	**PAH equiv.**	**SO_2 equiv.**	**Eco-ind. 95**
Difference*	**-10,3 %**	**-18,5 %**	**-19,5 %**	**-7,5 %**	**65,4 %**	**64,9 %**	**63,0 %**	**38,6 %**	**59,9 %**

***(Situation 1 – Situation 2) / Situation 1**

The results of this LCA were to compare with data on the Spanish electricity mix. This comparison depends strongly on CO_2 emissions. If the neutrality of CO_2 from renewable resources is considered, the life cycle inventory results of the Spanish electricity mix will not change remarkably, but in a life cycle study of municipal waste incineration the high content of renewable materials that are burned will provoke different results. Hence, adapting the LCA-methodology (Sonnemann et al., 1999c), the total amount of CO_2 in the incineration process had to be distributed in two parts between the carbon containing wastes: one for waste whose origin is in renewable resources and the other for waste with its origin in fossil fuels. Therefore, it is considered that the waste contains 13 % plastics, as the only component with its origin in fossil fuels, and that the plastic's content of carbon is 56.43 %, according to US-EPA (1996b). The total amount of CO_2 is calculated by the real stack emissions. The results indicate that the global warming potential would have been overestimated by a factor of more than four if the adaptation had not been done.

3.3.2 COMPARISON WITH TEAM ECOBALANCE AND ETH STUDY

The results of the TEAM based ecobalance of Schäfer (2000) for each of the two situations were imported into an Excel spreadsheet LCA to calculate the differences between the two situations. They were compared with the data from the Excel based LCA study with regard to the identified priority pollutants (see chapter 4) and additionally with the ETH study on waste treatment processes by Zimmermann et al. (1996). The results, except for particles, are presented in Figure 3.5.

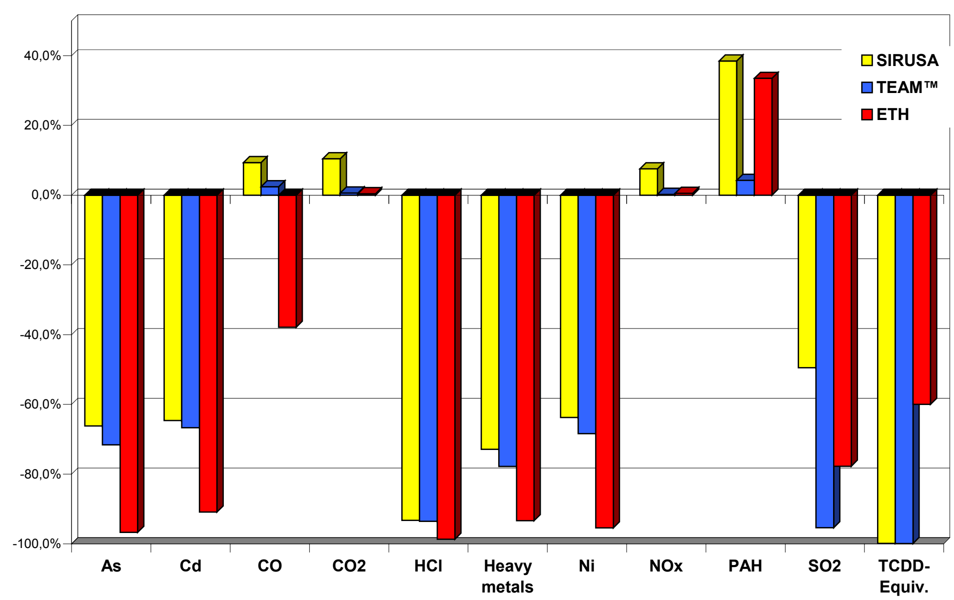


Figure 3.5 Comparison of Differences in LCA studies on MSWIs

For most of the pollutants all of the three sources lead to the same magnitude or at least to the same direction of change. The only exceptions are the differences of carbon monoxide. The literature data of ETH (Zimmermann et al., 1996) shows a reduction, but the SIRUSA results (STQ, 1998) show an increase. That may be caused by the assumption in the literature data that the organic compounds were completely oxidised or perhaps by another method of treatment. The particles are not presented since they behave very differently in the ETH study, probably due to the fact that another reference of what to consider as dust was chosen. A further analysis of the geographic and technical differences in life cycle inventories for MWSI process chains has been done by Ciroth el al. (2001).

The electricity produced by SIRUSA is fed into the Spanish electricity net. One ton of waste produces about one MJ electrical energy in the current situation with advanced flue gas treatment. This can be seen as an energy benefit because SIRUSA replaces the electricity production of a conventional electrical power plant. On one hand this saves resources; on the other hand some of the emissions are reduced in comparison to the Spanish electricity mix.

This quantitative comparison was also made with the software TEAM. For the emissions of Spanish electricity production the database integrated in TEAM was used. The calculation was normalised to 1.016 MJ electricity, which corresponds to the incineration of one ton municipal solid waste as functional unit. This correlation is the result of the ecobalance of TEAM for the current situation. The absolute values in Table 3.6 show the results for the priority air pollutants selected (see Figure 4.4).

Table 3.6: Comparison of the differences between the LCI data of 1 TJ electricity produced by the MSWI and the Spanish mix, indicating the relevance of the consideration of electricity benefit in such an LCA study

Pollutant	**Unit**	**MSWI Situation 2 “With filters”**	**Electricity Spain**	**Difference**	
		TJ	TJ	TJ	%
(a) Arsenic (As)	g	3.388E-02	2.985E-02	4.030E-03	-11.9%
(a) Cadmium (Cd)	g	3.972E-02	1.299E-02	2.673E-02	-67.3%
(a) Carbon Dioxide (CO_2, fossil)	g	2.702E+05	1.627E+05	1.075E+05	-39.8%
(a) Carbon Monoxide (CO)	g	3.084E+02	1.328E+03	-1.019E+03	330.6%
(a) PCDD/Fs (TEQ)	g	1.277E-08	1.997E-08	-7.197E-09	56.4%
(a) Heavy metals (sum)	g	6.073E-01	3.683E+00	-3.076E+00	506.4%
(a) Hydrogen Chloride (HCl)	g	1.972E+02	5.060E+01	1.466E+02	-74.3%
(a) Nickel (Ni)	g	5.902E-02	2.541E-01	-1.951E-01	330.6%
(a) Nitrogen Oxides (NOx as NO_2)	g	1.324E+03	3.321E+02	9.917E+02	-74.9%
(a) Particles (unspecified)	g	2.734E+03	7.724E+02	1.962E+03	-71.7%
(a) Sulphur Oxides (SOx as SO_2)	g	2.231E+02	9.289E+02	-7.058E+02	316.4%

In Table 3.6 positive values represent higher emissions of the average electricity production in Spain than in the SIRUSA plant. This means that emissions of CO, heavy metals, Ni and SO_2 are much higher in the conventional electrical power plants. The negative values present pollutants that are lower for the average Spanish electricity mix than in the MSWI incineration process chain of Tarragona.

3.3.3 Comparison with other single index LCIA methods

In Sonnemann (1998) several LCIA methods were applied to the LCI results of the Excel-based LCA study of the MSWI process chain. The overall results for the difference between the situation 1 without and situation 2 with an avanced gas system is presented in Figure 3.6 apart from the Eco-indicator 95, for MIPS (Schmidt-Bleek, 1994), EPS (Steen & Ryding, 1992) and the method of the Tellus Institute (1992).

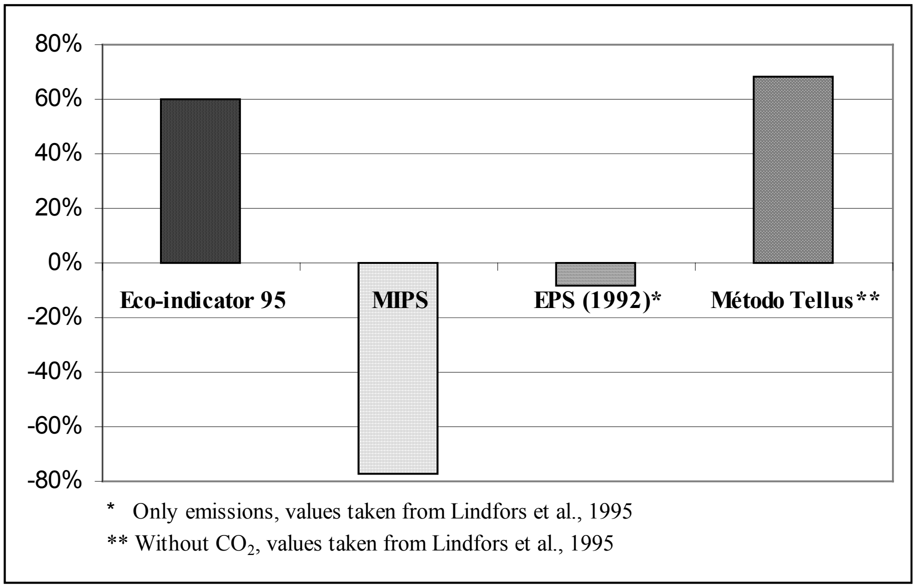


Figure 3.6: Comparison of different single index LCIA methods

It can be observed that the selected methods, corresponding to different weighting approaches (Table 2.11), do not deliver results with the same tendency. Two methods show an improvement and two others show a worsening of the environmental performance of the process chain under study. This observation questions the validity of an approach based on only a value to measure the environmental performance. However, above all it has to be highlighted that such kind of comparison is difficult anyway because the number and the type of environmental loads considered in each method vary significantly.

According to the EU environmental policy, the installation of advance gas treatment systems is obligatory to reduce the emission of gases such as SO_2 and HCl as well as particulate matter, PCDD/Fs and heavy metals. The results of the Eco-indicator 95 and the Tellus methods are in agreement with this policy, whereas the MIPS and EPS indicate the contrary. Using MIPS, such a result is found because more raw materials are necessary for the emission reduction technologies. In the EPS result obtained, more than 96 % of the total is caused by CO_2. In the same way as explained for the Eco-indicator 95 result the contribution of the heavy metals decreases, while the values for NOx and CO_2 increase, since they are not eliminated and the overall energy efficiency declines.

3.4 SITE-SPECIFIC IMPACT ASSESSMENT OF THE MWSI EMISSIONS

In parallel to the chain-orientated environmental impact assessment of the MSWI carried out by the LCA study, site-specific impact assessment methods have been applied. The site-specific impact assessment methods need geographic data, which are presented first of all in this section.

3.4.1 GEOGRAPHIC INFORMATION

Geographic information is crucial for site-specific environmental impact assessment. First, this data includes the information about the chosen grid. Next, the terrain evaluation in the grid has to be available as well as the distribution of the impact receptor (in the case of human health effects the population densities). Last but not least, the meteorological conditions have to be known, especially the wind speed and its direction.

To visualise geographic data like, for instance population densities and terrain evaluations, the Geographic Information System (GIS) MiraMon developed by the Universitat Autonoma de Barcelona was applied (Pons & Maso, 2000). Data in the format of this GIS were available through the internet from http://www.gencat.es, the server of the Generalitat de Catalunya (GenCat).

Grid

A screenshot of MiraMon in Figure 3.7 shows the grid used in the site-specific impact assessment studies. The figure illustrates clearly the usefulness of this powerful technical element.

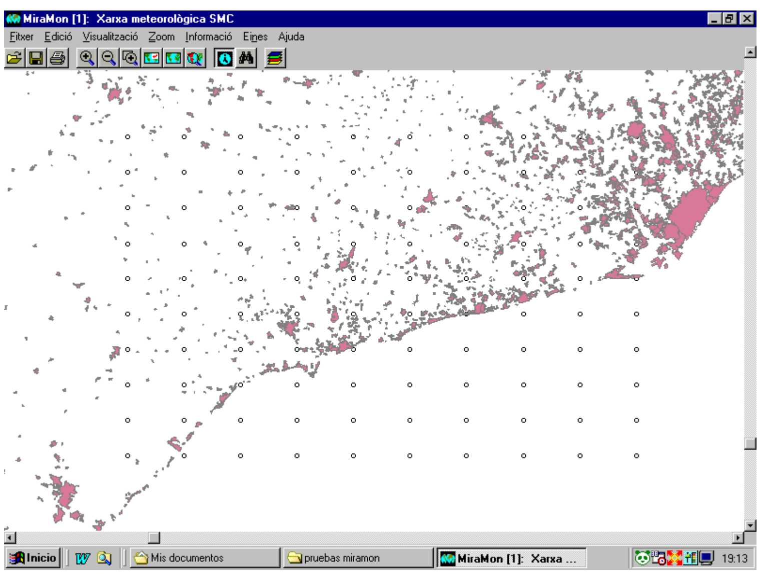


Figure 3.7: Screenshot of MiraMon with the grid used in the case study

The chosen grid corresponds to the local grid used by EcoSense for the Tarragona region (see section 3.4.3), with 100 separated squares. The incinerator is located approximately in the middle of the grid. Hence, also the grid used for ERA consists of a square with a side length of 126 km * 79 km, so that the whole size is 10,000 km. However, the grid in the ERA study is only subdivided into 25 separated squares.

Terrain elevation and population density

Figure 3.8 shows the geographical elevation in the grid area. In spite of the relative small extension of the grid the geographical properties of the area are varying quite a lot. There are a few mountains inside the grid up to an elevation of 900 m. The portion of the land area is with the size of 7,050 km^2 about 70 %. The rest of 2,950 km^2 is sea area. In the grid, there is a population of approximately 1.47 million inhabitants. Only 6.5 % of them (94,000 inhabitants) live in the town Tarragona. The biggest portion of the 56 % of the population stems from the western part of Barcelona (830,000 inhabitants). The rest is spread in the towns Reus, Montblanc, Tortosa, Hospitalet, Sitges, Vilafranca del Penedès, Vilanova i la Geltrú and other smaller towns. The graphical distribution of the population density, based on data from MiraMon/ GenCat, is shown in Figure 3.9.

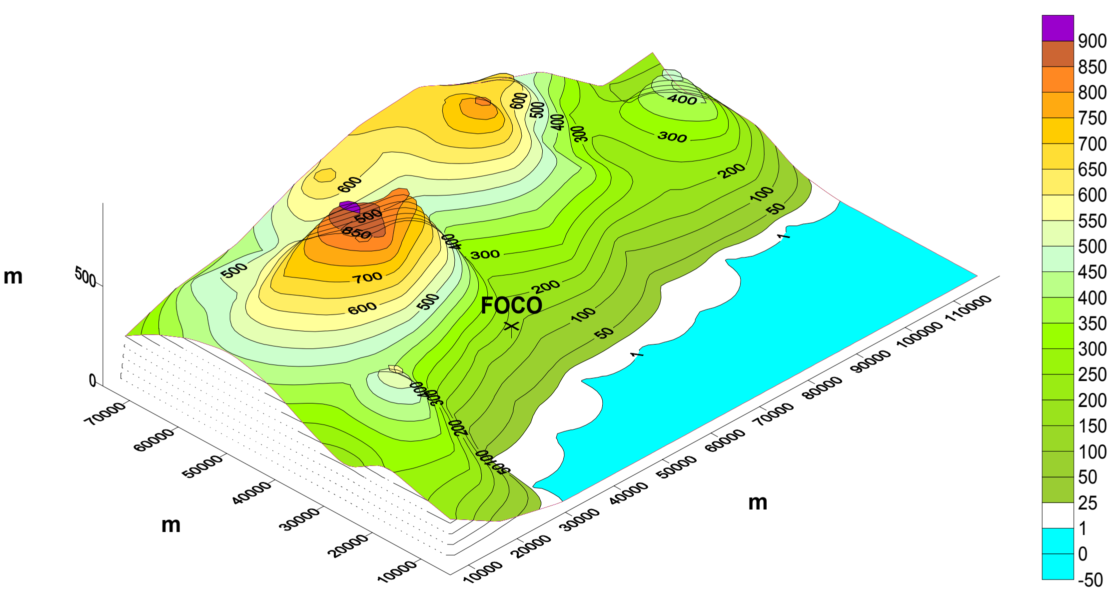


Figure 3.8: Elevation of the study area (data from MiraMon/ Generalitat de Catalunya)

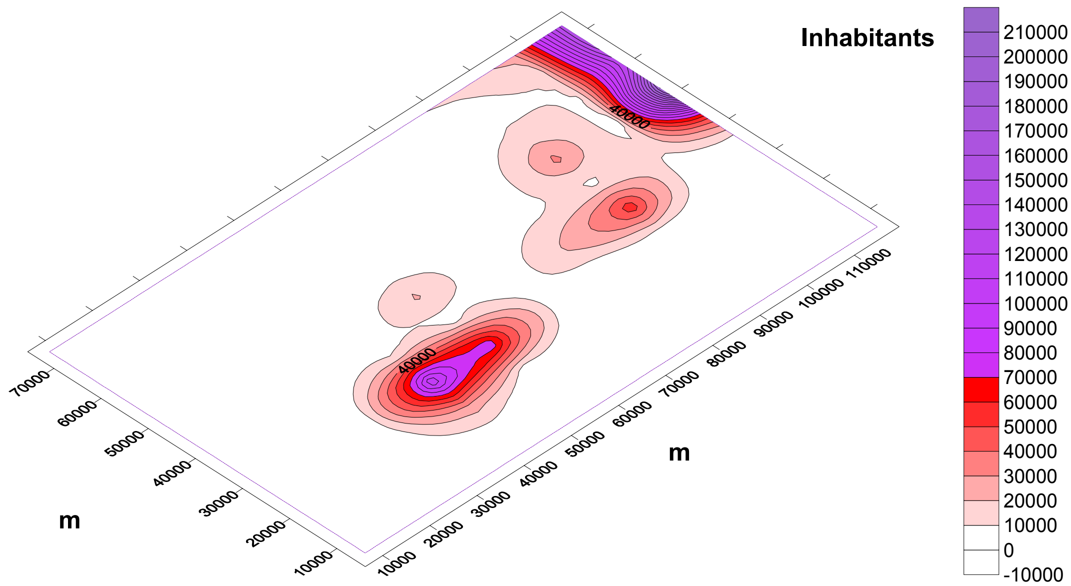


Figure 3.9: Population distribution in the study area (data from MiraMon/ Generalitat de Catalunya)

Meteorological situation

The calculation with the ISC-models need prepared meteorological data. The raw data were obtained from the Servei Meteorologic de Catalunya (SMC, 1999). They had to be prepared according to the algorithm provided by Cunillera (2000). The following parameters had to be inserted in the ISCST-3 model for every hour of the whole examined year: year, month, day, hour, flow-vector, the wind speed [m/s], temperature [K], wind stability class, rural mixing height [m], urban mixing height [m], the friction velocity [m/s], Monin-Obukhov-Length [m], the surface roughness [m], the precipitation code and the precipitation rate [mm/hr]. For the calculation of the pollutant distribution in the ERA and IPA case study meteorological data for the year 1997 were used.

3.4.2 ERA: CANCER RISK

An Environmental Risk Assessment with multimedia modelling of the MSWI incinerator emissions was carried out comparing the situation in 1996 (situation 1) and the current situation with an advanced gas treatment (situation 2) to evaluate the individual cancer risk reduction in the surroundings of the MWSI through the installation of the advanced acid gas removal system. In the case study only a few pollutants were assessed: the heavy metals Arsenic, Cadmium and Nickel as well as the organic pollutant PCDD/Fs.

The principal assumptions made are the following:

- In the present work only the risk due to the stack emission of the incinerator is estimated. It is not looked at other influences of the incinerator to the population, as for instance the emissions from the transport of the ash treatment.
- In each grid the pollutant is well mixed, so the calculated concentration at each receptor points is valid for a whole grid square.
- The population living in one grid is uniformly distributed over the square.

The distribution of the pollutants for the grid was calculated with the program BEEST, which uses the ISCST-3 model. The heavy metals were assumed as a part of the emitted particles because this is the only possible medium for the dispersion of these pollutants. The PCDD/Fs emission is divided into two paths according to Lee & Jones (1999): one part is in the gas phase (76 %), and the other part (24 %) is assumed to be adsorbed on the emitted particles. This is the distribution for one isomer; the mixture might be different from case to case.

The Figure 3.10 shows the simulation of the particles distribution over the grid calculated with a model emission stream of 1 g/s. The showed values in the figures are calculated from BEEST as an average value over one year.

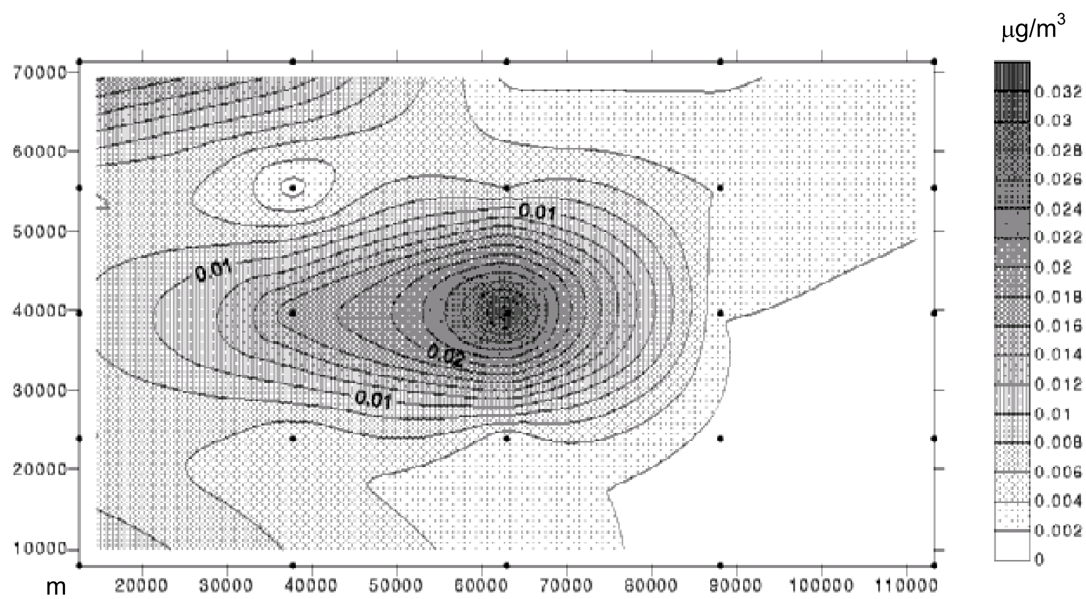


Figure 3.10: Simulation of the particles distribution over the grid as concentration increment due to the model MSWI emission stream of 1 g/s

Proceeding from the calculated immission concentration of the pollutants at the receptor points, the next step was the multimedia modelling of the pollutant distribution into the different environmental compartments with UNIRISK (see section 2.4.3). These concentrations are the basis for the following risk calculation.

In Table 3.7 the final results of the calculated risk are presented. The risk was subdivided into the risk of adults and the risk for children until the age of 2.5 years. The given values in the table are average values of the risk in the separate receptor points. Normally, the risk is expressed in the form of 1:1,000,000; that means that 8.55E-10 stands for $8.55*10^{-4}:10^{6}$ or 0.000855 cases per one million persons. US-EPA intends to control the exposition of toxic compounds to levels that cause a risk along the lifetime of 10^{-7} to 10^{-4} (Masters, 1991).

Table 3.7: Overview and comparison of the ERA results on human health

		Emission real	ADULT - RISK OF CANCER THROUGH					
			Ingestion soil	Ingestion vegetables	Inhalation particles	Inhalation air	Adsorption	**Total**
Arsenic	Without filter	0,02 mg/Nm³	2,80E-11	2,20E-10	1,10E-11	2,27E-08	2,64E-12	**2,30E-08**
	With filter	0,0056 mg/Nm³	7,84E-12	6,15E-11	3,09E-12	6,37E-09	7,39E-13	**6,44E-09**
	% Difference	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*
Cadmium	Without filter	0,02 mg/Nm³	5,60E-12	2,20E-10	1,39E-12	2,87E-09	9,51E-12	**3,10E-09**
	With filter	0,0066 mg/Nm³	1,85E-12	7,25E-11	4,59E-13	9,46E-10	3,14E-12	**1,02E-09**
	% Difference	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*
Nickel	Without filter	0,03 mg/Nm³	8,67E-12	8,36E-11	3,94E-13	8,12E-10	2,69E-12	**9,07E-10**
	With filter	0,0084 mg/Nm³	2,43E-12	2,34E-11	1,10E-13	2,27E-10	7,54E-13	**2,54E-10**
	% Difference	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*
Dioxin	Without filter	2,0E-06 ng/Nm³	6,15E-11	6,48E-10	3,81E-12	3,52E-09	1,74E-11	**4,25E-09**
	With filter	2,0E-09 ng/Nm³	6,15E-14	6,48E-13	3,81E-15	3,52E-12	1,74E-14	**4,25E-12**
	% Difference	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*
TOTAL	**Without filter**		**1,04E-10**	**1,17E-09**	**1,66E-11**	**2,99E-08**	**3,22E-11**	**3,13E-08**
	With filter		**1,22E-11**	**1,58E-10**	**3,66E-12**	**7,54E-09**	**4,65E-12**	**7,72E-09**
	% Difference		***-88,3%***	***-86,5%***	***-78,0%***	***-74,8%***	***-85,6%***	***-75,3%***

		Emission real	CHILD - RISK OF CANCER THROUGH					
			Ingestion soil	Ingestion vegetables	Inhalation particles	Inhalation air	Adsorption	**Total**
Arsenic	Without filter	0,02 mg/Nm³	3,89E-10	3,63E-10	3,58E-11	3,18E-08	1,76E-11	**3,26E-08**
	With filter	0,0056 mg/Nm³	1,09E-10	1,02E-10	1,00E-11	8,91E-09	4,93E-12	**9,14E-09**
	% Difference	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*
Cadmium	Without filter	0,02 mg/Nm³	7,79E-11	3,63E-10	6,77E-12	4,01E-09	6,34E-11	**4,52E-09**
	With filter	0,0066 mg/Nm³	2,57E-11	1,20E-10	2,24E-12	1,32E-09	2,09E-11	**1,49E-09**
	% Difference	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*	*-67,0%*
Nickel	Without filter	0,03 mg/Nm³	4,41E-11	2,06E-10	1,92E-12	1,14E-09	1,80E-11	**1,41E-09**
	With filter	0,0084 mg/Nm³	1,24E-11	5,76E-11	5,37E-13	3,18E-10	5,03E-12	**3,94E-10**
	% Difference	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*	*-72,0%*
Dioxin	Without filter	2,0E-06 ng/Nm³	8,55E-10	1,07E-09	1,24E-11	4,93E-09	1,16E-10	**6,98E-09**
	With filter	2,0E-09 ng/Nm³	8,55E-13	1,07E-12	1,24E-14	4,93E-12	1,16E-13	**6,98E-12**
	% Difference	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*	*-99,9%*
TOTAL	**Without filter**		**1,37E-09**	**2,00E-09**	**5,69E-11**	**4,19E-08**	**2,15E-10**	**4,56E-08**
	With filter		**1,48E-10**	**2,80E-10**	**1,28E-11**	**1,06E-08**	**3,10E-11**	**1,10E-08**
	% Difference		***-89,2%***	***-86,0%***	***-77,5%***	***-74,8%***	***-85,6%***	***-75,8%***

The Figure 3.11 gives an impression of the risk distribution for the different ways of contact with the pollutants as result of the application of UNIRISK. For each pollutant the sum of the risk for all pathways is 100 %. The distribution scheme shows with an average of approximately 75 % clearly the strong influence of the air inhalation to the total risk. In opposition to this the skin adsorption and the inhaled particles are without importance. The figure shows also a different influence of each pollutant to the children and the adults because of the different kind of ingestion. It can be said that the direct ingestion of soil is only important for the children. In opposition to this the risk through air inhalation is smaller for the children for every pollutant because of the smaller breath volume.

The percentage of the cancer risk due to inhalation of air is very important for the case study. For example, Nessel et al. (1991) describe values of 45.6 % due to inhalation of air for the common case. The higher value in the case study can be explained by the fact that the ingestion of food like fish, cow milk and beef were not considered in the version used of the multimedia model.

In another study of the Group AGA, Schuhmacher et al. (2001) compare the exposure to PCDD/Fs form MWSI emissions in Catalonia to the PCDD/F ingestion through the diet. This study, which included Monte Carlo simulations with sensitivity analysis, demonstrated that the diet contributed with more than 99 % to the total risk, whereas the risk due to PCDD/Fs by MSWI emissions was lower than 1 %.

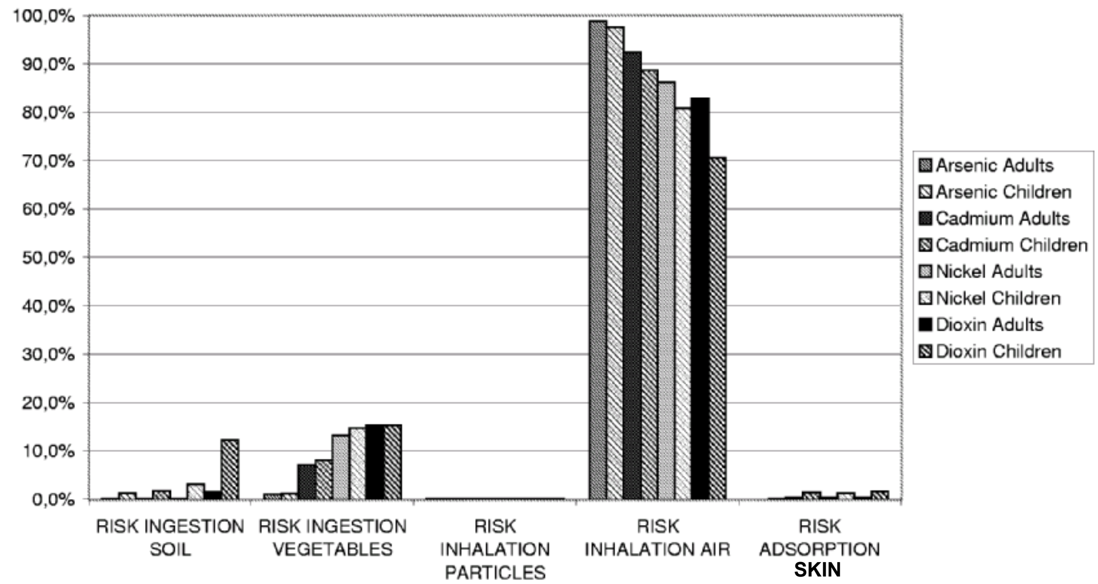


Figure 3.11: Risk distribution to the ingestions paths by calculations with UNIRISK

3.4.3 GENERIC IPA: AVOIDED EXTERNAL ENVIRONMENTAL COSTS

The Impact Pathway Analysis was applied to the MSWI emissions in a generic way through the integrated impact assessment model EcoSense. The aim was to estimate the avoided external environmental costs by the installation of an advanced acid gas treatment in the MSWI. In contrast to ERA, also the damages due to the long-range emissions are taken into account and not only cancer cases are considered.

The results can be presented on a local and regional scale as well as a total result, however the total is not the sum of regional and local values (Figure 3.12):

$$\text{Total} = \text{Regional} + \text{Local} - \cap\ (\text{local} + \text{regional})$$

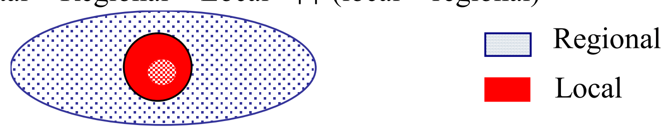


Figure 3.12: Consideration of local and regional damages in EcoSense

Some parameters cannot be evaluated on a local scale due to restricted data availability, for example the deposition of nitrogen. Furthermore, there are some impacts that cannot be converted into monetary values due to problems in the socio-economic evaluation, such as those to ecosystems.

In Table 3.8 the results of the generic IPA are presented. It includes the external environmental costs and the impacts that have not been converted into costs. In general, a clear reduction of the environmental costs can be identified. The environmental cost before is 0.023 €/ kWh and after the installation of the filters 0.018 €/ kWh. The type of damage that most contributes to the total are the damages to human health. The Relative Exceedance Weighted area (REW) is the only value that increase after the installation of the gas removal system. This is due to the fact that no NOx is reduced, but less electricity is produced.

Table 3.8: Environmental damage estimations for the MSWI emissions

	Situation 1 in 1996 “Without new filter”			**Current Situation 2 “With new filters”**		
	Per kWh	Per year	Percentage	Per kWh	Per year	Percentage
External environmental costs (€)	0.023	1,220,344	100%	0.018	953,414	100%
Humans (€)	0.023	1,214,215	99%	0.018	948,122	99%
Materials (€)	0.000115	6,021	0.49%	0.000093	4,871	0.51%
Crops (€)	0.000021	108	0.01%	0.000008	421	0.04%
Other impact indicators						
REW Ecosystems (km^2)	15.8	0.83		16.1	0.83	

Considering an average cost of the electricity of 0.07 €/ kWh, it can be concluded that approximately 25 % to 30 % of the benefit obtained by the selling the electricity are external costs. The installation of the advanced acid gas treatment system means that 0.005 €/ kWh of environmental costs are avoided; that is around 270,000 €/ year. Only less than 1 % of the external environmental costs does not stem from the damages to human health.

In Figure 3.13 the average environmental costs per kg of pollutant are presented. The environmental cost per pollutant according to this assessment method is PCDD/Fs, followed by Cd and PM10. The particles of the MSWI are considered to be mostly PM10.

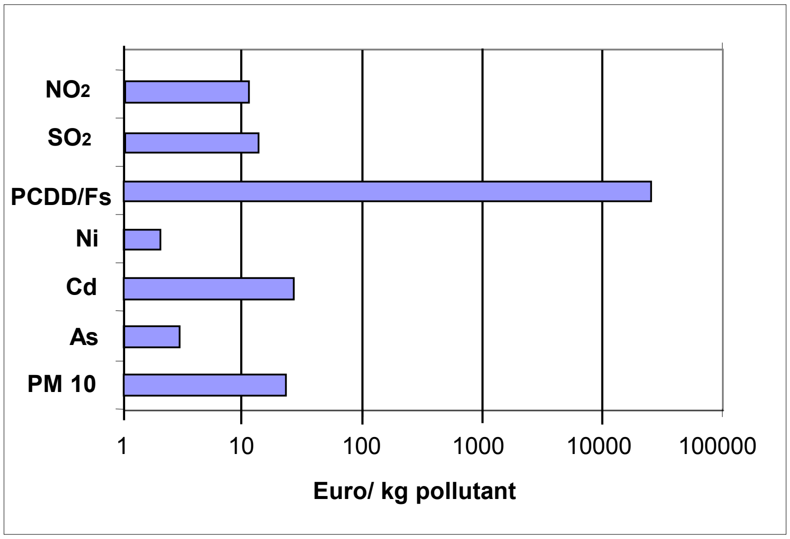


Figure 3.13: Environmental costs per kg of pollutant in the Tarragona region

During the realisation of the IPA study with EcoSense it was observed that the databases used in EcoSense from EuroStat is not as detailed as the one provided by the GIS MiraMon with the data from the Generalitat de Catalunya. This can be seen when comparing the population density data provided in Figure 3.9 (data from MiraMon/ GenCat) with the data shown in Figure 3.14 (data from EcoSense/ EuroStat).

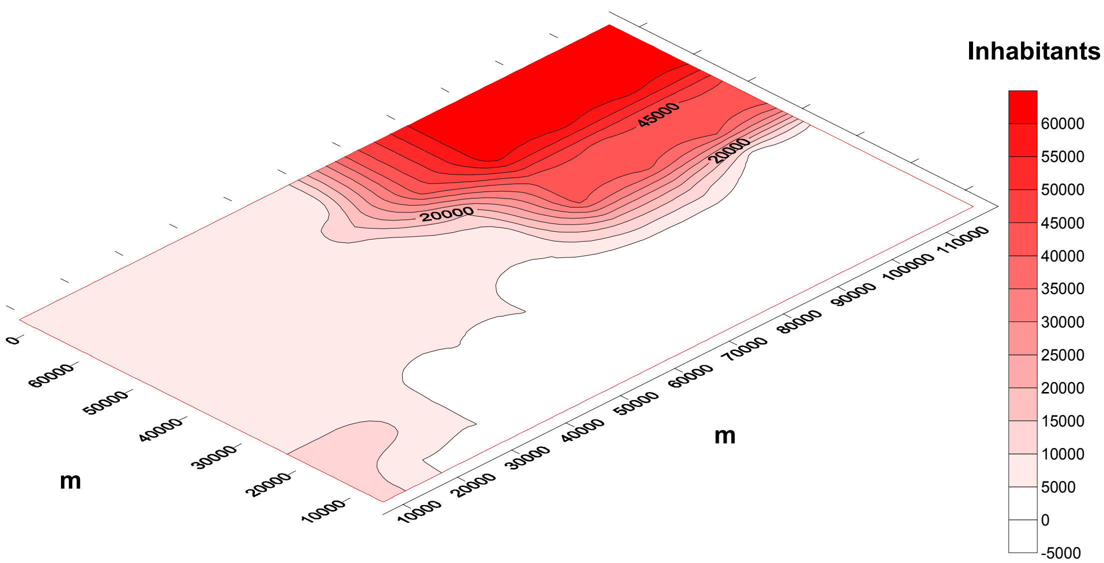


Figure 3.14: Population distribution in the study area (data from EcoSense/ EuroStat)

3.4.4 LOCAL IPA: USING MORE DETAILED DATA

The Impact Pathway Analysis has also been carried out for the estimation of damages to human health on a local scale. The focus has been on damages to human health because the former results in section 3.4.3 and other studies (Krewitt et al., 1999) have shown that these damages account for the main contribution to the total environmental damage. The spatial dimension has been local due to the particular interests for the district population that financed the advanced gas treatment system by additional costs for the treatment of their waste.

To calculate the incremental concentrations the dispersion model ISCST-3, integrated in the software BEEST by Beeline (1998), has been used like in the Environmental Risk Assessment study presented in chapter 3. Beside these immission concentrations, the population exposed is the crucial factor in the dose-response and exposure-response functions that are used in the effect analysis. Instead of using the information taken from the EuroStat Regio database incorporated in the EcoSense software (IER, 1998) more detailed population data from the Catalan government's are used. They were provided from the Departament de Medi Ambient by the Geographic Information System MiraMon (Pons & Masó, 2000) and from the Departament de Sanitat in their yearly public health report.

The numeric results of this application (as external environmental costs) are presented together with an uncertainty analysis in section 4.7.

3.5 IDENTIFICATION OF THE LIFE CYCLE TYPE

Looking at the presented life cycle of the waste incineration (Figure 3.4) it is clear that the case study is not a complex product system with a huge number of important processes involved, but an industrial process chain where also the sites or regions are generally known (Figure 3.2). The essential foreground and background processes amount to less than 20.

The number of processes to consider further in more detailed environmental damage estimations is even smaller, since a dominance analysis based on the Eco-indicator 95 results represented in chapter 5 (Figure 5.17) shows that only very few processes contributes significantly to the total environmental impact. The same holds true for the pollutants. Only a small number of pollutants are very important for the total environmental impact score (see Figure 4.4).

It can be assumed that, in general, also the predominant media where the pollutants are emitted to can be identified. In our case study it is air. In this way the effort of more detailed environmental damages estimations can be focused on a reduced number of processes and pollutants in one medium.

Part C:

Methodology Development

4 UNCERTAINTY ASSESSMENT BY MONTE CARLO SIMULATION

Uncertainty sources are numerous in environmental impact assessment (Weidema, 2000). As this is part of what is called system analysis we are using and creating models that are not easy to verify, and therefore they have their inherent uncertainty. This holds for life cycle inventories as well as for (multimedia) pollutant fate models. Furthermore, in the effect analysis, the dose-response and exposure-response functions are highly uncertain. Finally, the economic or any other type of weighting adds additional uncertainty. Other sources of uncertainties are simply due to the incorrectness or the lack of data.

4.1 TYPES OF UNCERTAINTIES IN ENVIRONMENTAL IMPACT ASSESSMENT

In order to systematise these sources the following types of uncertainties have been distinguished (Huijbregts, 1998):

Parameter Uncertainty
A large amount of data is usually needed in the LCI analysis and in the models that calculate the fate, exposure and effect in the impact assessment. Uncertainty of these parameters causes uncertainty in the outcome of any environmental impact method. Empirical inaccuracy (imprecise measurements), unrepresentativity (incomplete or outdated measurements) and lack of data (no measurements) are common sources of parameter uncertainty. Weidema & Wesnæs (1996) describe a comprehensive procedure for estimating the combined inaccuracy and unrepresentativity of life cycle inventory data both qualitatively and quantitatively. Although this procedure may substantially improve the credibility of the outcomes of LCAs, uncertainty analysis is generally complicated by a lack of knowledge of uncertainty distributions and correlations between parameters.

Model Uncertainty
According to our experience with the models described in the previous chapter, the predicted values of environmental impacts and risk respond generally in a linear manner to the amount of emitted pollutant. Moreover, in LCIA and IPA thresholds for the environmental interventions are disregarded. Additionally in LCIA the derivation of characterisation factors

causes model uncertainty because they are computed with the help of simplified environmental models, not considering spatial and temporal characteristics.

Uncertainty due to Choices

When performing environmental impact assessments choices are unavoidable. Examples of choices leading to uncertainty in the LCI analysis are the choice of the functional unit and the choice of the allocation procedure for multi-output processes, multi-waste processes and open-loop recycling. Moreover, the socio-economic evaluation in LCA and IPA is an area in which choices play a crucial role. Although experts from social science have suggested many weighting schemes, only a few are operational and no general agreement exists as to which one should be preferred.

Spatial and Temporal Variability

In most LCAs all environmental interventions are summed up regardless of the spatial context of the intervention, introducing model uncertainty in LCAs. Temporal variations are present in both the LCI and other environmental impact assessment method. In general, variations of environmental interventions over a relatively short time period, such as differences in industrial emissions on weekdays versus weekends or even short disastrous emissions, are not taken into account.

Variability Between Sources and Objects

In LCA, but also in other environmental impact assessment methods, variability between sources and objects may influence the outcome of a study. That means for example that differences in inputs and emissions of comparable processes in a product system due to the use of different technologies in factories, which produce the same material, cause variability in life cycle inventories. Furthermore, variability between objects exists in the weighting of environmental problems in the impact assessment due to the variability between human preferences. When, for instance, the WTP is used to determine the external environmental cost due to a determined damage differences between individual preferences cause inherent variation in the final result.

4.2 WAYS TO DEAL WITH THE DIFFERENT TYPES OF UNCERTAINTY

Huijbregts et al. (2000a) have offered some solutions on how to deal with the above-discussed issues of uncertainty. The tools available to address different types of uncertainty and variability in LCAs include probabilistic simulation, correlation and regression analysis, additional measurements, scenario modelling, standardization, expert judgement or peer review and non-linear modelling. Scenario modelling (Pesonen et al., 2000) should be useful especially in cases where there is uncertainty about choices and temporal variability.

When a model suffers from large model uncertainties, the results of a parameter uncertainty analysis may be misleading. The result of decreasing model uncertainty will in most cases be the implementation of more parameters in the computation, thereby increasing the importance of operationalising parameter uncertainty in the model.

In the following two sections an overview of previous efforts is given to assess the uncertainties in LCA (within a process chain-perspective) and in IPA (for site-specific damage estimations).

4.2.1 EXPERIENCES IN LIFE CYCLE ASSESSMENT

So far the influence of data quality on final results has rarely been analysed (Maurice et al., 2000). Only by making more experiences with data quality assessment approaches the existing methods can be improved and the uncertainties in life cycle studies could become better known.

In spite of the lack of published case studies, several approaches to evaluate the quality of results have been proposed during the last years. The existing methods can be differentiated in qualitative and quantitative assessment.

Qualitative assessment signifies to describe the used data by means of a characterisation of their quality. Weidema &Wesnaes (1996) and Weidema (1998) proposed to use data quality indicators depending on categories like reliability, completeness, temporal correlation etc. Finnveden & Lindfors (1998) suggested ranges for various inventory parameters as rules of thumb.

On the other hand, quantitative assessment means to quantify all the inherent uncertainties and variations in a LCA. Among others, Hansen & Asbjornsen (1996) used analytical statistics, Ros (1998) fuzzy logic, and Maurice et al. (2000) as well as Meier (1997) stochastic methods. Quantitative assessment is confronted in particular with the problem that it is hardly possible to analyse all types of uncertainty.

The presented methods are especially applicable for uncertainty analysis of Life Cycle Inventories. On the other hand, Meier (1997), Hofstetter (1998) and Huijbregts et al. (2000b) have reported uncertainties for the life cycle impact assessment.

On the basis of this review the uncertainty and variability in a LCI of the electricity produced by a waste incinerator has been assessed. In accordance with the conclusions of Coulon et al. (1997), in the present case study a mixed approach combining qualitative and stochastic quantitative methods has been used to evaluate the uncertainty in the LCI. As stochastic simulation model the Monte Carlo method has been chosen. The realisation of Monte Carlo simulations provides decision-makers far more information than a single estimate of damage.

4.2.2 FORMER UNCERTAINTY ASSESSMENT IN IMPACT PATHWAY ANALYSIS

The Impact Pathway Analysis is a quite complex approach and hence there is a risk for a lack of reliability in the final results. In the same way as in many other environmental methods and especially in commercial software tools, the uncertainty is the key problem that makes it difficult to convince decision-makers by the outcomes of a study.

The uncertainty and variability has been analysed with analytical statistical methods in an uncertainty analysis of damages and costs of air pollution undertaken by Rabl & Spadaro (1999). They show that the equation for the total damage is largely multiplicative, even though it involves a sum over receptors at different sites. This follows from the principle of conservation of matter which implies that overprediction of the dispersion model at one site is compensated by underprediction at another; the net error of the total damage arises mostly from uncertainties in the rate at which the pollutant disappears from the environment. They discuss typical error distributions for the factors in the equation for the total damage, in particular those of two key parameters: the deposition velocity of atmospheric dispersion models, and the value of statistical life; according to Rabl & Spadaro (1999) they are close to lognormal. They conclude that a lognormal distribution for the total damage appears very plausible whenever the dose-response or exposure-response function is positive everywhere. As an illustration they show results for several types of air pollution damage (health damage due to particles and carcinogens, damage to buildings due to SO_2, and crop losses due to O_3): the geometric standard deviation is in the range of 3 to 5.

4.3 INTRODUCTION TO MONTE CARLO SIMULATION

According to the experiences in LCA and the uncertainty study in the ExternE project for the IPA, it seems that the use of a stochastic model, like Monte Carlo Simulation (LaGrega et al., 1994), helps better to characterise the uncertainties, rather than a pure analytical mathematical approach. This is due to the fact that the relevant parameters follow different frequency distribution. For example, not all of them are lognormal nor all of them are normal distributed.

To perform Monte Carlo (MC) simulation, parameters have to be specified as uncertainty distributions. The method varies all the parameters at random, the variation is restricted by the given uncertainty distribution for each parameter. The randomly selected values from all the parameter uncertainty distributions are inserted in the output-equation. Repeated calculations produce a distribution of the predicted output values, reflecting the combined parameter uncertainties. This can be considered as the most effective quantification method for uncertainties and variability in environmental impact assessment tools (LaGrega et al., 1994).

The word simulation is meant to refer to any analytical method meant to imitate a real-life system, especially when other analyses are too mathematically complex or too difficult to reproduce. Without the aid of simulation, a spreadsheet model will only reveal a single outcome, generally the most likely or average scenario. Spreadsheet uncertainty analysis uses both a spreadsheet model and simulation to automatically analyse the effect of varying inputs on outputs of the modelled system.

The random behaviour in games of chance is similar to how Monte Carlo simulation selects variable values at random to simulate a model. When rolling a die, one knows that a 1, 2, 3, 4, 5, or 6 will come up, but does not know which one happens to be in any particular roll. It's the same with the variables that have a known range of values but an uncertain value for any particular time or event (Decisioneering, 1996).

The software Crystal Ball Version 4.0 from Decisioneering (1996) is a simulation programme that helps to analyse the uncertainties associated with Microsoft Excel spreadsheet models by Monte Carlo Simulation. To the best, worst, and most likely case versions of the same model - which only predict the range of outcomes - Crystal Ball adds probability and automates the 'what-if' process. Crystal Ball is an add-in to Excel. The user does not have to leave Excel to do such forecasting. The programme works with already existing models, so the calculations do not have to be recreated. As a fully integrated Excel add-in program with its own toolbar and menus, Crystal Ball picks up where spreadsheets end by letting perform Monte Carlo analysis. The user has to define a range for each uncertain value in the spreadsheet, and Crystal Ball uses this information to perform thousands of simulations.

Another function of Crystal Ball is the sensitivity analysis. Sensitivity charts show how much influence each assumption has on the results, allowing to focus further analytical effort on the most important factors.

The results are dynamically summarised in forecast charts that show all the outcomes and the likelihood of each. An example of the activated cells of an Excel sheet, the result of a sensitivity analysis and the final provision are shown in the Crystal Ball screenshot (Figure 4.1).

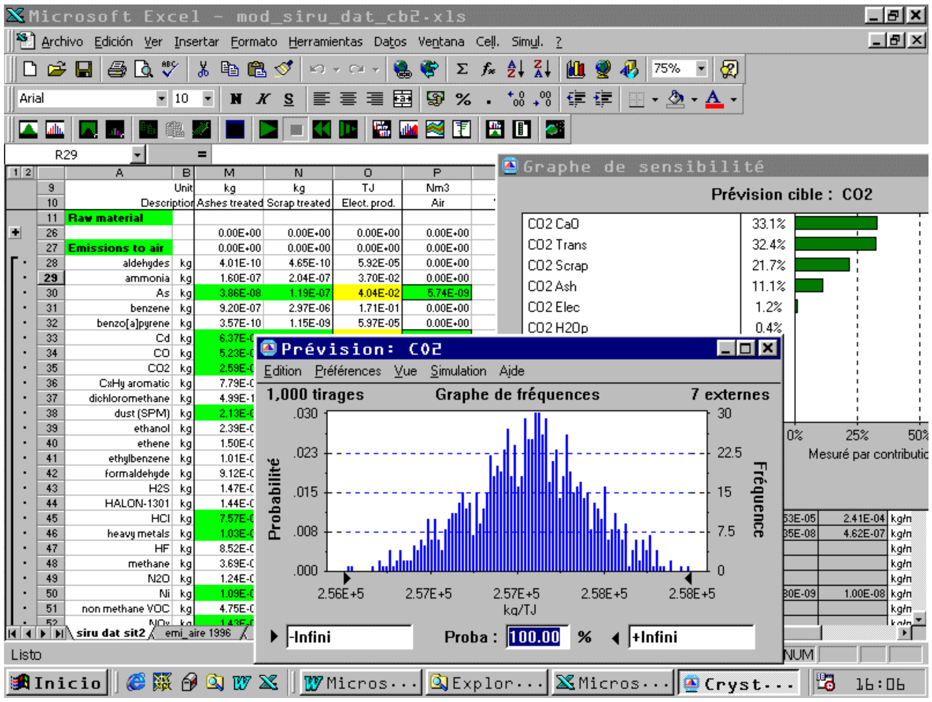


Figure 4.1: Crystal Ball screenshot (activated cells of an Excel sheet, the result of a sensitivity analysis and the final provision)

4.4 Framework for the Uncertainty Assessment by Monte Carlo Simulation in Environmental Impact Assessment Methods

Based on the information about previous studies on uncertainty evaluation in environmental impact assessment methods and the knowledge of the Monte Carlo simulation technique the following objectives have been established:

- Adapt the existing methods within a strategy for the assessment of uncertainties that is based on MC Simulation.
- Create a framework for the assessment of uncertainties in Life Cycle Inventories that is based on MC Simulation and apply it to the case study.
- Create a framework for the assessment of uncertainties in the Impact Pathway Analysis that is based on MC Simulation and apply it to the case study.

That means that in a first step the general elements have to be identified that are necessary for the uncertainty analysis by Monte Carlo simulation. In a second step, two similar frameworks or procedures have to be established for LCI and IPA; in this way first the uncertainties are analysed with regard to the collection of the environmental loads along the life cycle for the process chain and then the same is done for the damage estimations at one site.

4.4.1 General strategy

The general strategy can be characterised by the following elements:

- Compilation of necessary data
- Classification of this data (extensively available, based on little information and ignorable data)
- Identification of probability distribution for considered data
- Monte Carlo Simulation
- Sensitivity analysis
- Analysis and discussion of results

As shortly mentioned above, by means of a sensitivity analysis it is possible to show which parameters have most importance for the final result. If small modifications of one parameter characterised by a probability distribution influence strongly the final result it can be concluded that the sensitivity of the considered variable is elevated for the relation between parameter and final result. This information is crucial for decision-makers in order to understand which are the variables to act on and moreover it could be very important to know the parameters that might be neglected, especially if it is hard to get detailed information about them. The sensitivity can be analysed by an approach that displays the sensitivity as a percentage of contribution from each parameter to the variance of the final result. The software Crystal Ball Version 4.0 approximates this approach by lifting to square the correlation coefficients of ranks and normalising them to 100 %.

In the next two sections the elements presented above are adapted to fit in the context of LCI and IPA.

4.4.2 Uncertainty assessment in the Life Cycle Inventory

In Figure 4.2 the procedure for the uncertainty analysis in the LCI is presented. It has been adapted from the existing literature (Meier, 1997; Maurice et al., 2000). The first step is the compilation of the life cycle inventory data. If all the parameters that might have a repercussion on the final result were considered, an exhaustive study would have to be carried out. However, not all these data are relevant. Hence, only the most relevant factors have to be selected and for some parameters it can be assumed that they have a fixed value. Once the essential factors have been selected, a characterisation of the probability distributions is carried out. Therefore, the data are classified into two groups: extensively available data for which average and standard deviation can be calculated and data based on little information, for which literature and experts estimations have to be used. All these parameters feed the Monte Carlo simulation that gives the results in form of a probability distribution around a mean value and allows carrying out a detailed sensitivity analysis.

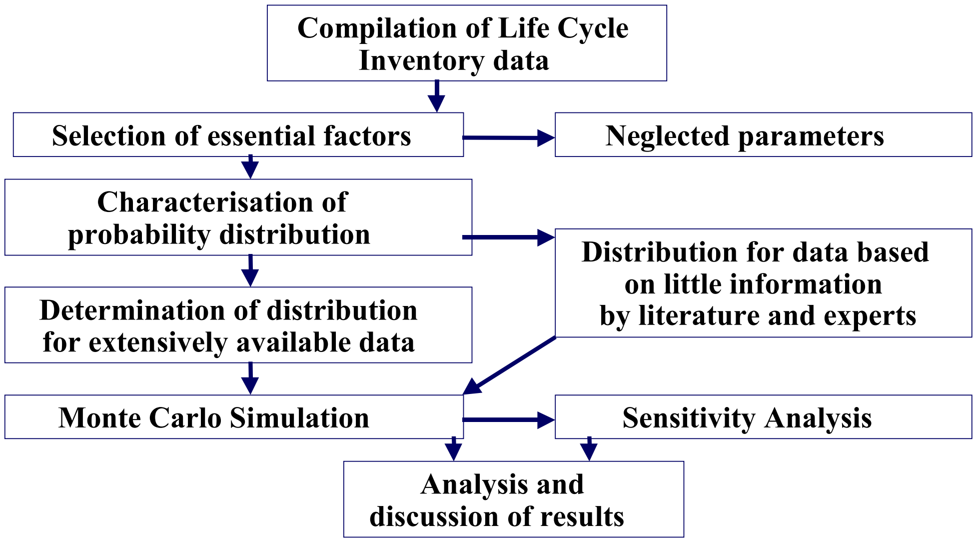


Figure 4.2: Procedure for the uncertainty and variability assessment in the Life Cycle Inventory

4.4.3 UNCERTAINTY ASSESSMENT IN THE IMPACT PATHWAY ANALYSIS

In Figure 4.3 the framework for uncertainty assessment in the impact pathway analysis is presented. The first step is the compilation of damage function data, wherefore an exhaustive study has to be carried out about all the parameters that have a repercussion on the final result. However, not all these data are relevant. Although the model is processing a huge quantity of data, only fundamental facts have really to be considered. Hence, a classification has to be made between the most significant parameters, for which probability distributions ought to be defined, and the parameters supposed to be invariant, which are called point estimates. The significant data are further classified into two groups for advanced evaluation: extensively available data, for which average and standard deviation can be calculated, and data based on little information, for which literature and experts estimations are used. In the same way as in the uncertainty assessment for the Life Cycle Inventory, these parameters feed the Monte Carlo simulation that gives the results in form of a probability distribution around a mean value and allows carrying out a detailed sensitivity analysis. The last step consists of the discussion of these results.

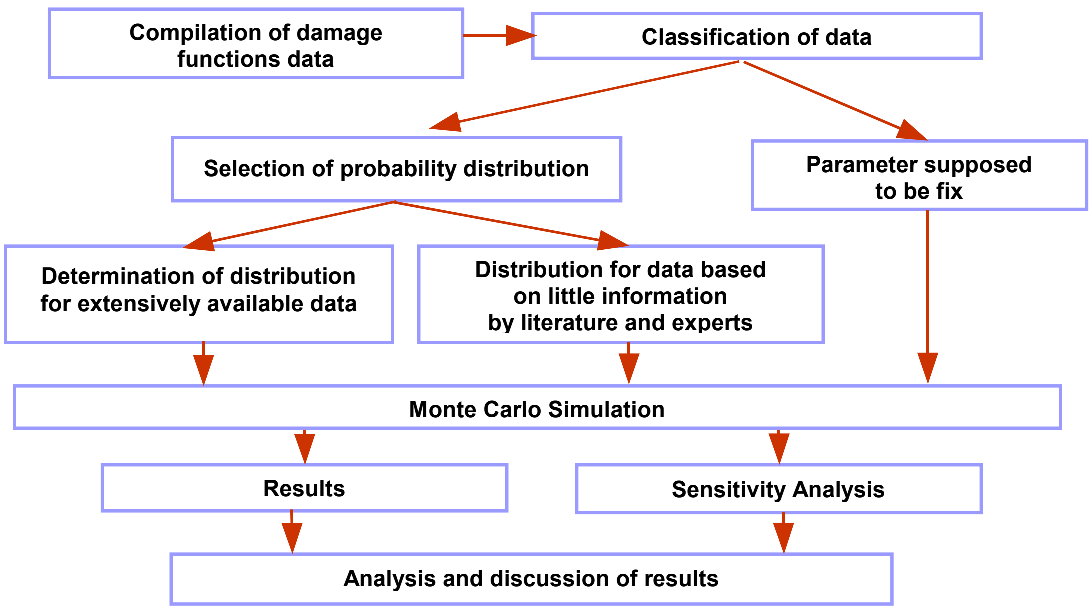
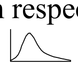


Figure 4.3: Framework for the assessment of uncertainty and variability in the Impact Pathway Analysis

4.5 TYPES OF PROBABILITY DISTRIBUTIONS USED

In the MC simulation, at least 10,000 times new values of the random variables are selected and a new estimate of the final damage is foreseen. The results of the calculations are summarised in a single histogram of damage values. All kind of distributions (normal, lognormal, uniform etc.) and operations (multiplication, exponential functions, matrix calculations) can be managed.

The types of probability distributions used in this study are:

1. Normal 
   Normal distribution which is appropriate to describe the uncertainties of large samples that are stochastic events and symmetrically distributed around the mean. The probability density function is defined by the mean and the standard deviation. The normal distribution is especially appropriate if the data uncertainties are given as percentage of the standard deviation with respect to the mean, i.e. coefficient of variation (CV).
2. Log-normal 
   The log-normal probability distribution which is appropriate if a large number of quantities has to be represented, if no negative values are possible and if the variance is characterised by a factor rather than by a percentage. The 50%-percentile of a log-normal distribution corresponds to the mean of the corresponding normal distribution.

The lognormal distribution is calculated assuming the logarithm of the variable has a normal distribution. Many environmental impacts follow the lognormal model. Its modelisation is given by the expressions (4.1) and (4.2).

$$\mu = \exp\left(\xi + \frac{\theta^2}{2}\right) \tag{4.1}$$

$$\sigma^2 = \left[\exp\left(\theta^2\right) - 1\right] \times \exp\left(2\,\xi + \theta^2\right) \tag{4.2}$$

Where μ is the ordinary mean, and σ is the standard deviation of the lognormal distribution. ξ is the standard deviation of the Gaussian variable, and θ is its standard deviation.

It allows to calculate the geometric mean μ_g and the geometric standard deviation σ_g by expressions (4.3) and (4.4).

$$\mu_g = \exp(\xi) \tag{4.3}$$

$$\sigma_g = \exp(\theta) \tag{4.4}$$

These variables are very practical, corresponding to the mean and the coefficient of variation for the normal distribution: Moreover, they provide multiplicative confidence intervals such as:

- [μ_g / σ_g, μg σ_g] for a confidence interval of 68%
- [μ_g /σ_g^2, μg σ_g^2] for a confidence interval of 95%

4.6 Application of the framework to the Life Cycle Inventory of the electricity produced by a waste incinerator

In this section the presented framework for uncertainty assessment in Life Cycle Inventories is applied to the LCI of the electricity produced by the municipal solid waste incinerator, as introduced in chapter 3. It is tried to document well each step for further practical use:

1. Assigning probability distributions to the considered parameters
2. Assessing the uncertainties and variations in the computation of the LCI table
3. Determining the most relevant parameters in such a LCI by sensitivity analysis

1) Assigning probability distributions to the considered parameters

The predominant pollutants in the LCI were selected by a combined quantitative and qualitative approach. The quantitative selection consisted of a dominance analysis done on the basis of the results in the impact assessment carried out by the Eco-Indicator 95 method (Goedkoop, 1995). Figure 4.4 presents the contribution of the considered pollutants to the total environmental potential impact measured as Eco-indicator 95. Only emissions with a contribution higher than 1 % to the total environmental impact were selected for the uncertainty assessment. The considered pollutants were Cd, CO_2, HCl, Ni, SO_2, heavy metals and particles. Moreover, because of its carcinogenity and its consideration as primary air pollutants in the ExternE project (EC, 1998) As, CO and PCDD/Fs were also taken into account.

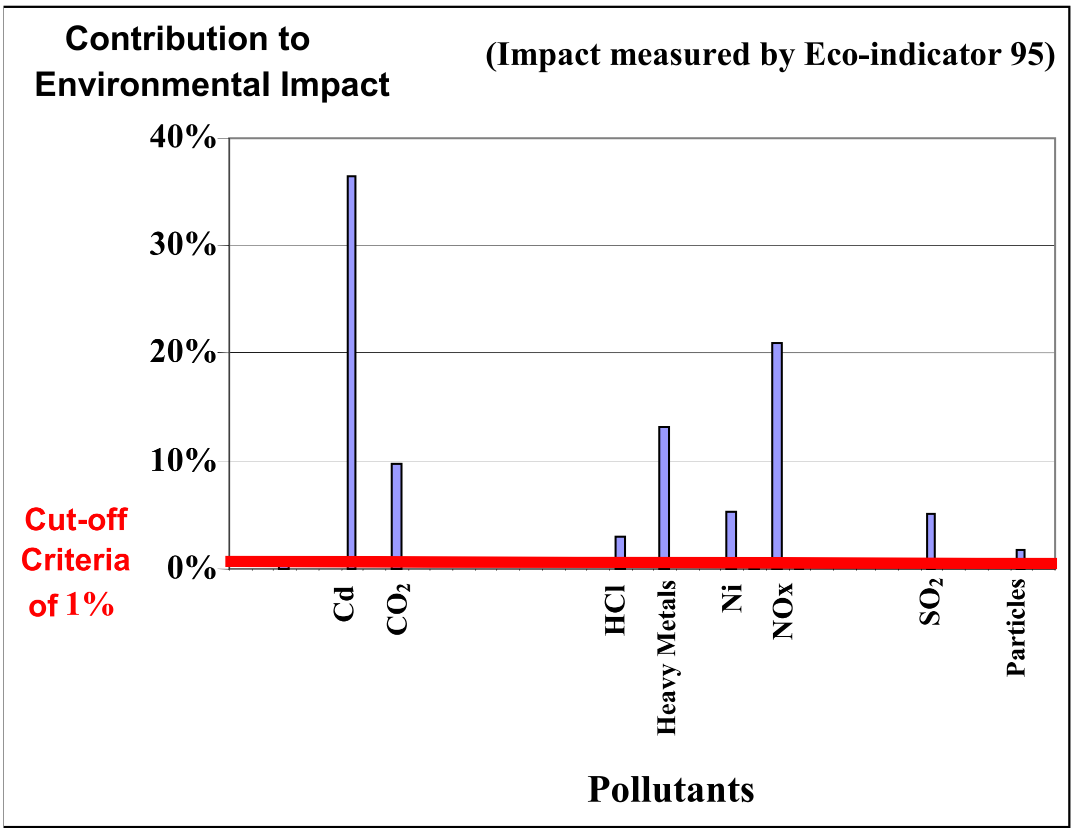


Figure 4.4: Selection of essential pollutants by dominance analysis

Only a proper determination of the probability distribution is possible if data are extensively available, as it is the case for the emissions measured, electricity production, etc. In these cases the probability distributions were calculated from experimental data provided by the LCA study (STQ, 1998) and by the director of the MSWI, either by the mean of a report (Nadal, 1999) or personally.

Based on a relevant number of measurements with its inherent variations the best fitting probability density function for the respective type of data was chosen (normal of lognormal distribution). The quality of the fitting was assessed by the Kolmogorov-Smirnov test for parameters with less than 30 measurements and by the Chi^2 Test for the parameters with more than 30 measurements. The software Crystal ball allows to carry out this fitting of probability distributions. The variation of the emissions was huge in our study because of the heterogeneous waste incinerated.

The concentrations of the incinerator process emissions together with their distribution type and deviations are presented in Table 4.1. Except for PCDD/Fs, the variations of the pollutant concentration emissions were fitted from experimental data by lognormal distributions with a geometric standard deviation (σ_g) between 1.5 and 3.4. It can be seen that the variation of the measurements of nickel and other heavy metal concentrations in the emissions are in general much higher than the one of the macropollutants SO_2, NOx and CO. These variations are huge because waste is a heterogeneous fuel.

Table 4.1: Site-specific data - Concentrations in the emissions of the incineration plant*

Parameter	Unit	Distribution	Former Situation 1	Current Situation 2	References
		Type	Mean Value (σ_g)	Mean Value (σ_g)	
As emissions	mg/Nm^3	lognormal	2.00E-02 (3.4)	5.60E-03 (3.4)	(STQ, 1998)
Cd emissions	mg/Nm^3	lognormal	2.00E-02 (1.7)	6.60E-03 (1.7)	(STQ, 1998)
CO emissions	mg/Nm^3	lognormal	4.00E+01 (1.5)	4.00E+01 (1.5)	(STQ, 1998)
PCDD/Fs emissions	pg/Nm^3	lognormal	2000 (2)	2 (2)	(STQ, 1998; Rabl & Spadaro, 1999)
HCl emissions	mg/Nm^3	lognormal	5.16E+02 (1.6)	3.28E+01 (1.6)	(STQ, 1998)
Heavy metals	mg/Nm^3	lognormal	4.50E-01 (2.5)	9.10E-02 (2.5)	(STQ, 1998)
Ni emissions	mg/Nm^3	lognormal	3.00E-02 (2.2)	8.40E-03 (2.2)	(STQ, 1998)
NO_x emissions	mg/Nm^3	lognormal	1.91E+02 (1.5)	1.91E+02 (1.5)	(STQ, 1998)
Particles emissions	mg/Nm^3	lognormal	2.74E+01 (2.1)	4.80E 00 (2.1)	(STQ, 1998)
SO_2 emissions	mg/Nm^3	lognormal	8.09E+01 (1.5)	3.02E+01 (1.5)	(STQ, 1998)

* CO_2 emissions are determined stochiometrically.
σ_g - Geometric standard deviation

Due to the lack of sufficient experimental data, the probability distributions for the PCDD/Fs concentrations in the emissions were considered to be lognormal with a geometric standard deviation of 2 according to the estimations published by Rabl and Spadaro (1999).

Also in the case of life cycle data taken from Frischknecht et al. (1996), site-specific data on transport processes as well as the inputs and outputs for local production and waste treatment processes, all with little information on the data quality, the uncertainty estimations are made according to literature (Weidema & Wesnaes, 1996; Meier, 1997). Table 4.2 shows the technical data of the incinerator.

Table 4.2: Site-specific data - Technical variables of the incineration plant

Parameter	Unit	Distribution Type	Former Situation 1 Mean Value (CV)	Current Situation 2 Mean Value (CV)	References
Electricity production	TJ	Normal	158.56 (8.08)	149.55 (6.94)	(Nadal, 1999)
Waste treated	t	Normal	153,467(5,024)	148,450 (5,024)	(Nadal, 1999)
Yearly working hours	h	Normal	8,280 (0.05)	8,280 (0.05)	(Nadal, 1999)
Flue gas volume	Nm^3/h	Normal	90,000 (0.05)	90,000 (0.05)	(Nadal, 1999)
Transport	tkm	Normal	4,100,000 (0.2)	4,100,000 (0.2)	Weidema & Wesnaes, 1996; STQ, 1998
Plastic proportion	%	Normal	13 (0.1)	13 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Electricity consumption	TJ	Normal	1.66 (0.1)	1.66 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Diesel	t	Normal	148.8 (0.1)	148.8 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Lubricant Oil	t	Normal	2.3 (0.1)	2.3 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Lime (CaO)	kg	Normal	0 (0)	921,000 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Water de-ionised	m^3	Normal	19,665 (0.1)	19,665 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Water refrigeration	m^3	Normal	5,175 (0.1)	5,175 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Water purified	m^3	Normal	7,360 (0.1)	7,360 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Unspecified Water	m^3	Normal	8,122 (0.1)	33,120 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Ashes treated	kg	Normal	590,000 (0.1)	3,450,000 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Scrap treated	kg	Normal	2,740,000 (0.1)	2,740,000 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998
Slag	t	Normal	42,208 (0.1)	42,208 (0.1)	Weidema & Wesnaes, 1996; STQ, 1998

CV - Coefficient of Variation with the exception of Electricity production and Waste treated that are expressed as normal standard deviation.

These data are site-specific data as waste treated, electricity production and consumption, transportation distances as well as material inputs and outflows. All these technical data show a normal distribution. The variations in the data on the waste treated and the electricity produced per year are described by their normal standard deviations, due to the fact that the deviations could be calculated out of a sufficient number of data. For the yearly working hours and the gas volume flow a coefficient of variation of 5 % was estimated according to the information given by the technical staff of the incinerator. The probability distributions for the other technical data had to be derived from literature (Weidema & Wesnaes, 1996; Meier, 1997). Hence, a normal distribution with a CV of 10 % was assumed for the site-specific inflows and outflows, while for the transport distances a normal distribution with a CV of 20 % was chosen because of the large uncertainties in the exact description of the waste transport.

A huge amount of the data used in the LCI are not directly related to the incinerator itself, but to the life cycles of associated inputs, outputs and transport processes as can be seen in the Figure 3.4. The data of these life cycles were not obtained in a site-specific manner in our study but by the use of the ETH-database (Frischknecht et al., 1996) where these data have been collected from a Swiss perspective on a European scale. It is evident that the transfer of these data to the Spanish situations causes definitely an uncertainty that according to Meier (1997) differs in a way depending on the considered pollutant. For information taken out of databases Meier (1997) proposes to assume classes of normal probability distributions with the following coefficients of variations:

1. For data obtained by stochiometric determination a CV of 2 %
2. For actual emission measurements or data computable in well-known process simulation a CV of 10 %
3. For well-defined substances or sum-parameters a CV of 20 %
4. For data acquired of specific compounds by an elaborated analytical method a CV of 30 %

For a better understanding of these estimates, in Table 4.3 it is specified which pollutant emission corresponds in the study to which class. Moreover, an example for the life cycle data for the input flow of energy consumption in Spain is presented. According to this scheme, CO_2 is the only environmental load that has been determined stochiometrically for all life cycle data. Due to this relative certain assessment method it has got a low CV of 2 %. CO, NOx and SO_2 were considered to be obtained by actual emission measurements or to be computed in well-known process simulations depending on multiple parameters. As there are more possibilities for errors a CV of 10 % has been assumed. For well-defined substances or sum parameters as heavy metals, PCDD/Fs and HCl a CV of 20 % has been proposed. Finally in the class for specific compounds with elaborated analytical methods the uncertainty is the highest with a CV of 30 %. In our study this CV was assumed for the life cycle database information on particles. The example in Table 4.3 shows normal mean values and its standard variations for the life cycle emissions of CO_2, NOx and HCl per 1 TJ of electricity produced in Spain.

Table 4.3: Uncertainties in the measurements of emissions in the ETH process modules according to Meier (1997) and Frischknecht et al. (1996)

Parameter Types	Distribution Type	Uncertainty (CV)
1. Substances determined stochiometrically (CO_2)	normal	0.02
2. Actual emission measurements or emissions of well-known processes depending on multiple parameters (CO, NOx, SO_2)	normal	0.10
3. Well-defined substances or sum-parameters (As, Cd, HCl, heavy metals, Ni, PCDD/Fs)	normal	0.20
4. Specific compounds with elaborated analytical methods (particles)	normal	0.30

CV - Coefficient of Variation

2) Assessing the uncertainties and variations in the computation of the LCI table

Following the procedure for uncertainty assessment proposed, a Monte Carlo simulation was run for each situation with these described probability distributions. The final result consists of two histograms for each selected pollutant, corresponding to the two situations studied, one with and the other without an advanced gas treatment system. Each simulation has been made in one separate run per situation. Because of the inherent variability of the Monte Carlo model, it is not possible to affirm that the set of values of the input variables used in the run of the current situation 1 are going to be the same in the former situation 2. Because of the generation of random numbers every run occurs in a different way. In order to verify the importance of this variability on the final outcome, both simulations have been also carried out in one run. The results obtained showed negligible variations.

In the Figure 4.5 the first LCI result is presented, it is the case of the heavy metals in the current situation with advanced acid gas treatment system. The mean life cycle emissions of the heavy metals per 1 TJ of electricity produced by the incinerator are 7.50E-01 kg with a normal standard deviation of 5.50E-01 kg/TJ. In the x-axis, it is possible to observe pollutant emissions per energy produced. The y-axis shows the probability of each value of the life cycle emissions. The total number of iterations carried out with the software Crystal Ball was 10,000. As the density distribution of the result is best adjusted by a lognormal density function, the geometric mean of 6.10E-01 kg/TJ and the geometric standard deviation of 1.92 have been calculated. On this basis a 68 % confidence interval from 3.18E-01 kg/TJ to 1.17E+00 kg/TJ was obtained.

Figure 4.6 to Figure 4.15 show the LCI results for all other pollutants in the current situation with advanced gas treatment. As they all can be adjusted well by a lognormal distribution they are all treated the same way in the computations for the 68 % confidence interval. The results were 3.37E-02 kg/TJ for As, 2.62E-02 kg/TJ for Cd and 2.43E+02 kg/TJ for CO (geometric standard deviation of 2.3, 1.67 and 1.37 respectively).

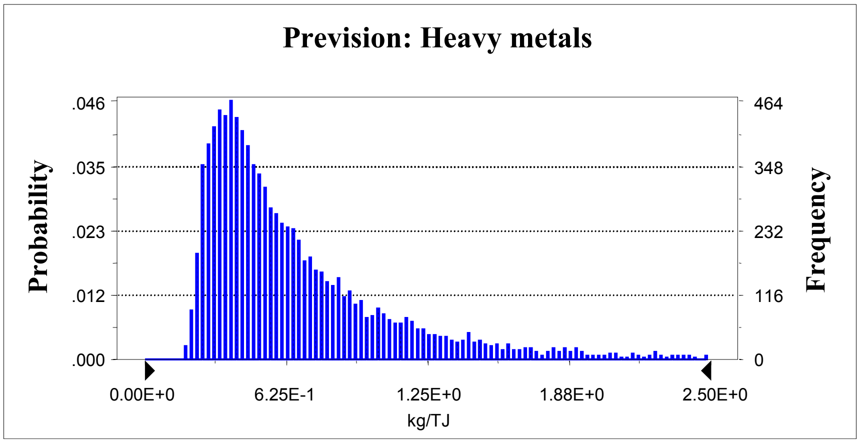


Figure 4.5: Monte Carlo Simulation LCI results for heavy metals in the current situation

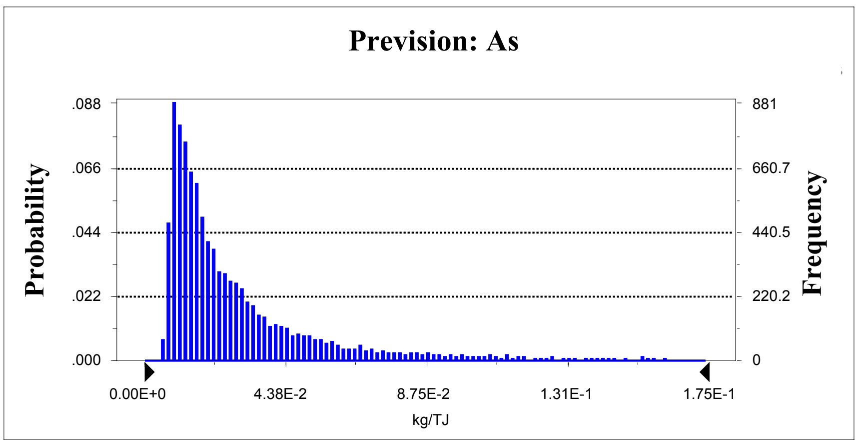


Figure 4.6: Monte Carlo Simulation LCI results for arsenic in the current situation

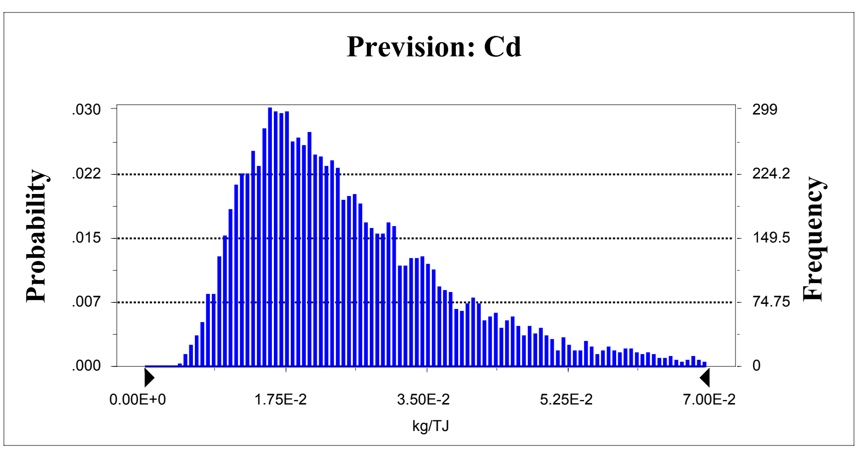


Figure 4.7: Monte Carlo Simulation LCI results for cadmium in the current situation

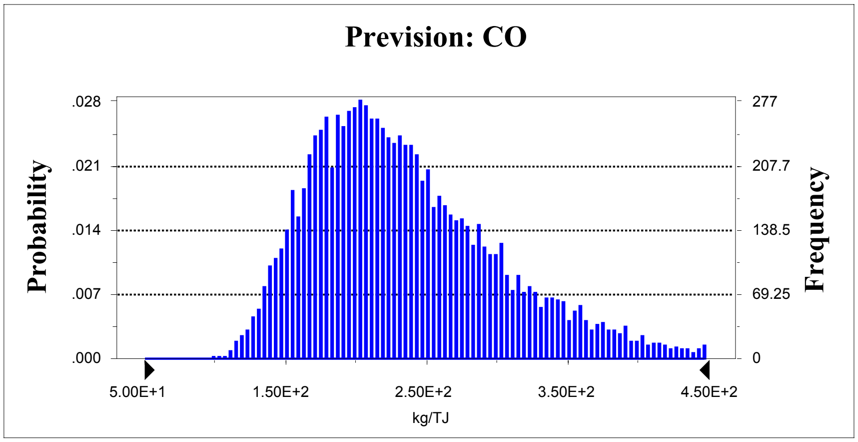


Figure 4.8: Monte Carlo Simulation LCI results for CO in the current situation

The CO_2 emissions of the incinerator are not determined by the measurements that are assumed to be lognormal distributed, but by multiplying the waste treated with the proportion of plastics in the waste (the only component with its origin in fossil fuels), taking into account a factor of 2.0691 kg CO_2 per kg plastic (see section 3.3.1) and dividing by the gas volume. Hence, for this pollutant a less lognormal but rather normal distribution is generated (Figure 4.9). The mean is 3.01E+05 kg/TJ (σ_g of 1.13).

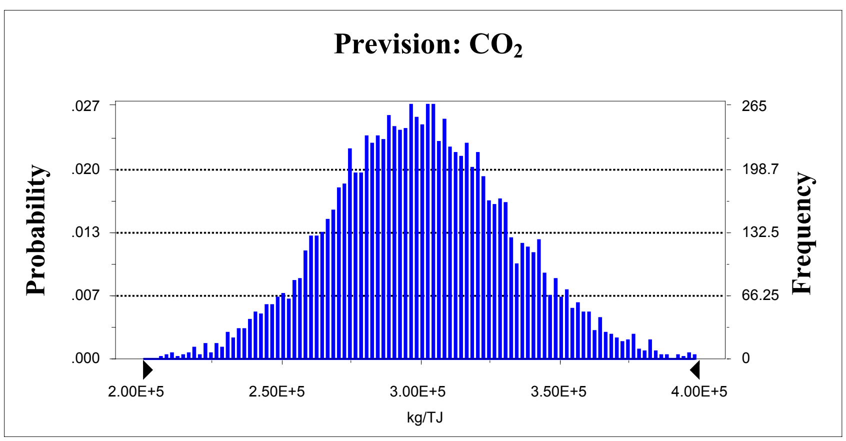


Figure 4.9: Monte Carlo Simulation LCI results for CO_2 in the current situation

The LCI result for the emitted particles, HCl, nickel and arsenic in the current situation are shown in Figure 4.10, Figure 4.11, Figure 4.12 and Figure 4.13 (particles: 1.62E+02 kg/TJ, σ_g 1.31; HCl: 1.39E+02 kg/TJ, σ_g 1.52; Ni: 6.35E-02 kg/TJ, σ_g 1.80; NOx: 1.04E+03 kg/TJ, σ_g 1.42). The shape of the LCI results for SO_2 in Figure 4.14 (2.12E+02 kg/TJ, σ_g 1.29) are similar to the ones of NOx and CO due to the same σ_g for the incinerator emissions.

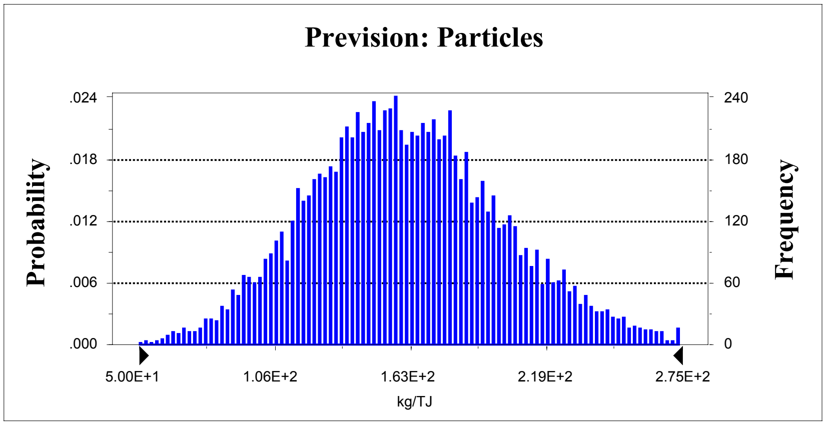


Figure 4.10: Monte Carlo Simulation LCI results for particles in the current situation

The shape of the distribution of the PCDD/Fs life cycle emissions in the current situation, Figure 4.15 (1.30E+04, σ_g 1.75) is in the same way as for all pollutants except particles very similar in the former situation without the advanced gas treatment. The LCI result by Monte Carlo simulation for the particles in the former situation is illustrated in the Figure 4.16 (1.50E+02 kg/TJ, σ_g 1.93). A clear change can be seen in the probability distribution from a lognormal to a rather normal one after the installation of the advanced acid gas removal system.

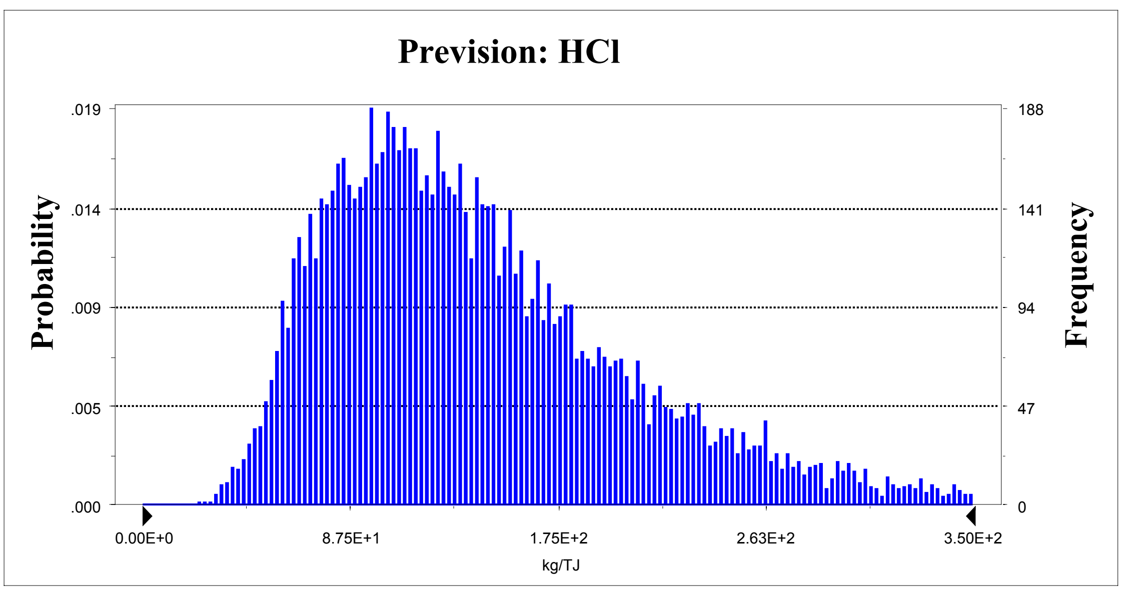


Figure 4.11: Monte Carlo Simulation LCI results for HCl in the current situation

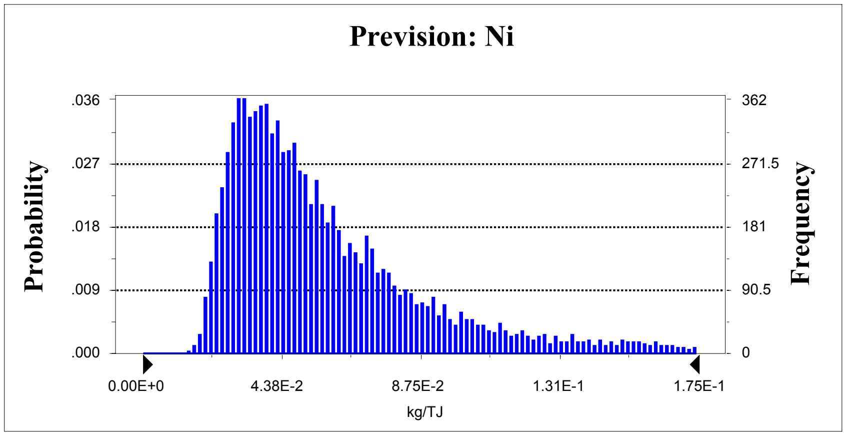


Figure 4.12: Monte Carlo Simulation LCI results for nickel in the current situation

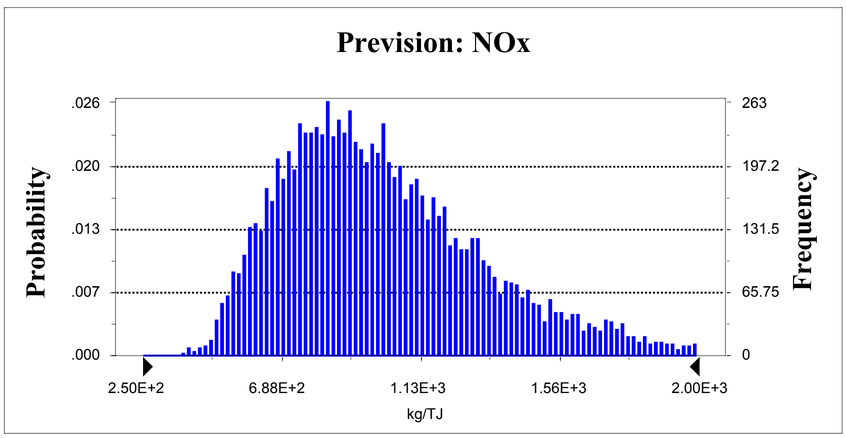


Figure 4.13: Monte Carlo Simulation LCI results for NOx in the current situation

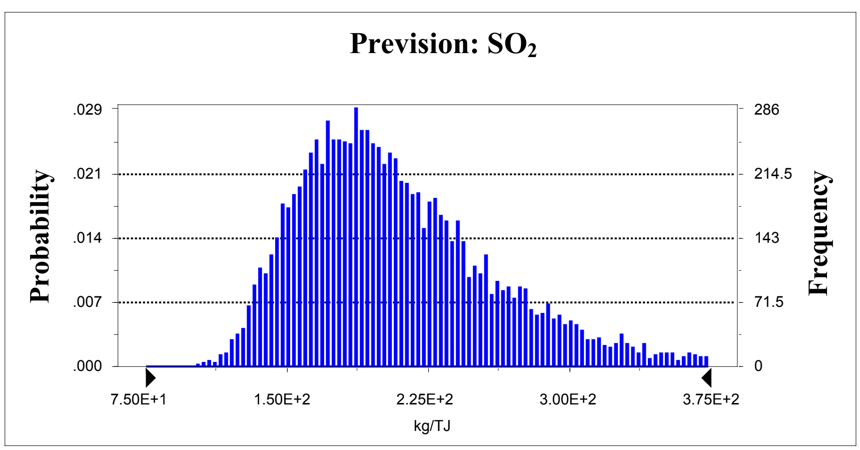


Figure 4.14: Monte Carlo Simulation LCI results for SO_2 in the current situation

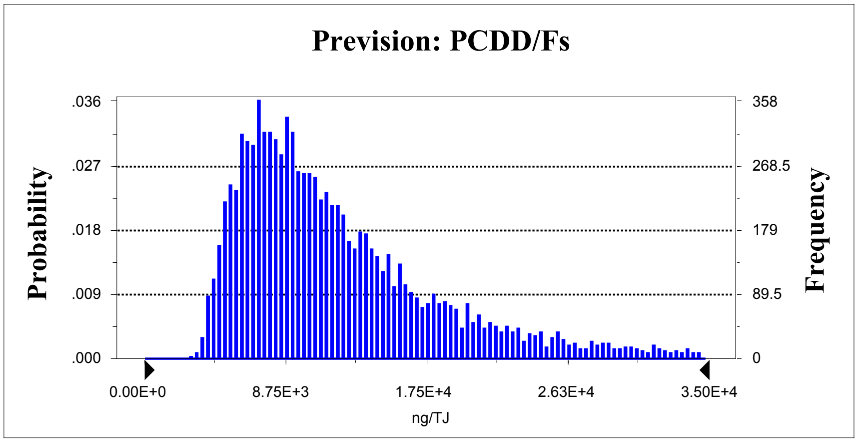


Figure 4.15: Monte Carlo Simulation LCI results for PCDD/Fs in the current situation

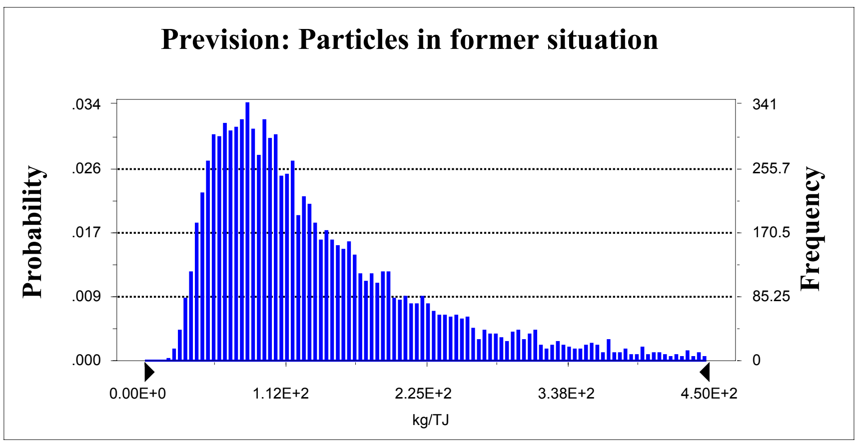


Figure 4.16: Monte Carlo Simulation LCI results for particles in the former situation

The mean values with confidence intervals for former and current situation for all studied pollutants are presented in Figure 4.17. Heavy metals were only considered as a sum parameter. For the PCDD/Fs, heavy metals, SO_2 and HCl a clean reduction can be observed, especially for the first one, while for CO_2, CO, particles and NOx no variation of the life cycle emissions per TJ electricity produced is found. In these cases changes were smaller than the given confidence intervals. Here, in an evident way, the detected uncertainty and variability interferes in the results and influences the interpretation of them.

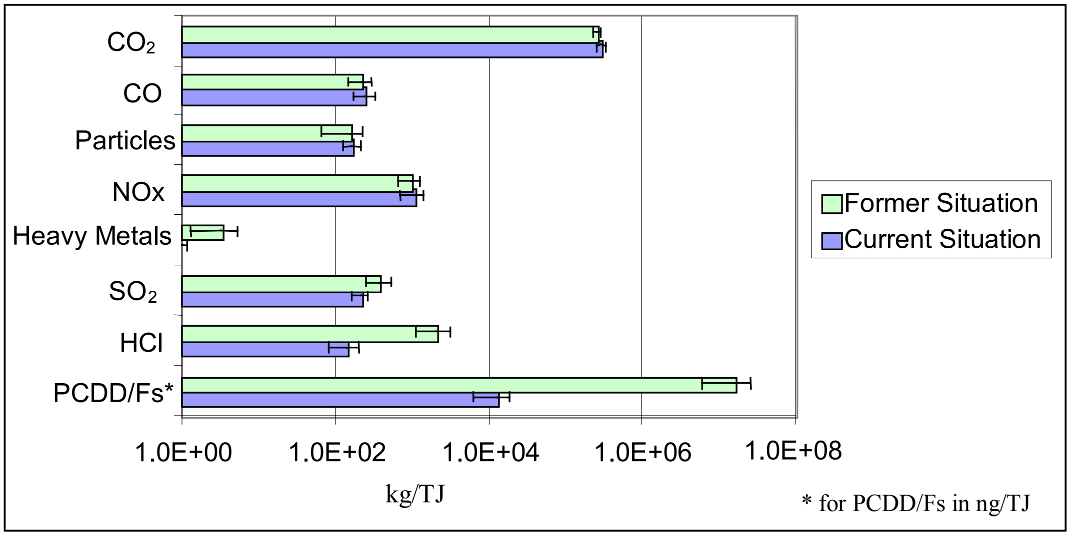


Figure 4.17: LCI results with confidence interval of 68 %

3) Determining the most relevant parameters in such a LCI by sensitivity analysis

The results of the sensitivity analysis for the PCDD/Fs are presented for the former and for the current situation (Figure 4.18 and Figure 4.19 respectively). In both situations the PCDD/Fs emitted by the incinerator process itself were the most important parameter (99,9 % and 99,6 % respectively). The same results were obtained for the other pollutants with percentages over 95 % with the exception of particles. Hence, only the results for particles are further described.

In the former situation the major emission of particles was due to the process of incineration. The effect of the other processes was practically negligible, as can be seen in Figure 4.20.

In the current situation the lime used in the advanced acid gas system (scrubber) has been added to the life cycle. This process generates a huge quantity of dust. Hence, as shown in Figure 4.21, this process contributes with 83.6 % to the variance of the life cycle particle emissions. The particles from the process of incineration itself present only 15.6 % and the other processes together less than 1 %.

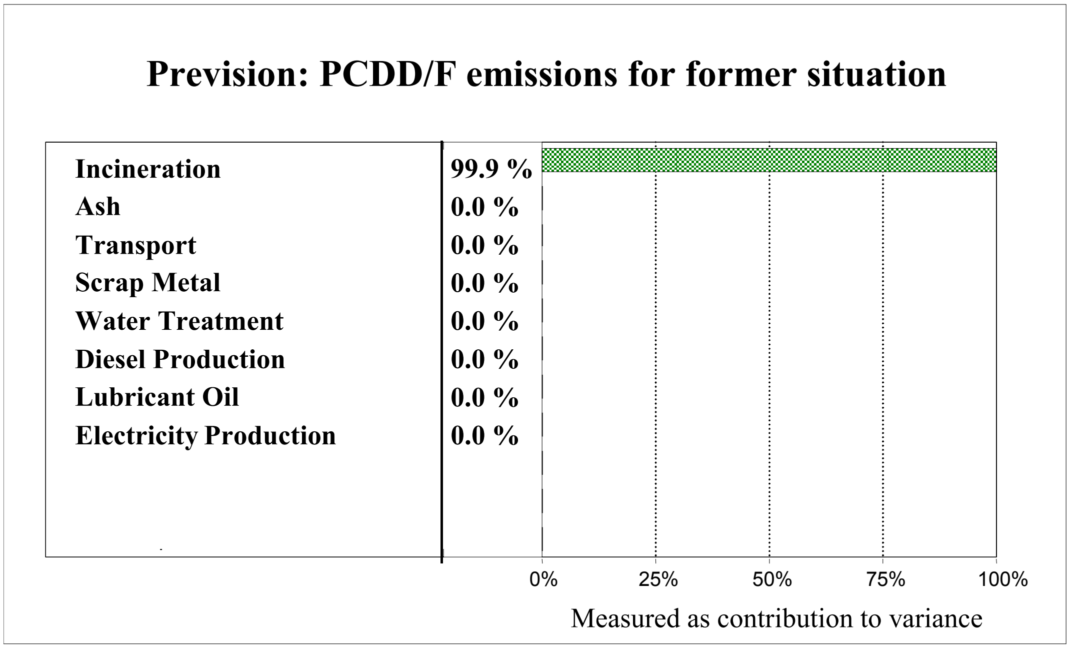


Figure 4.18: Sensitivity analysis of the LCI result for PCDD/Fs in the former situation

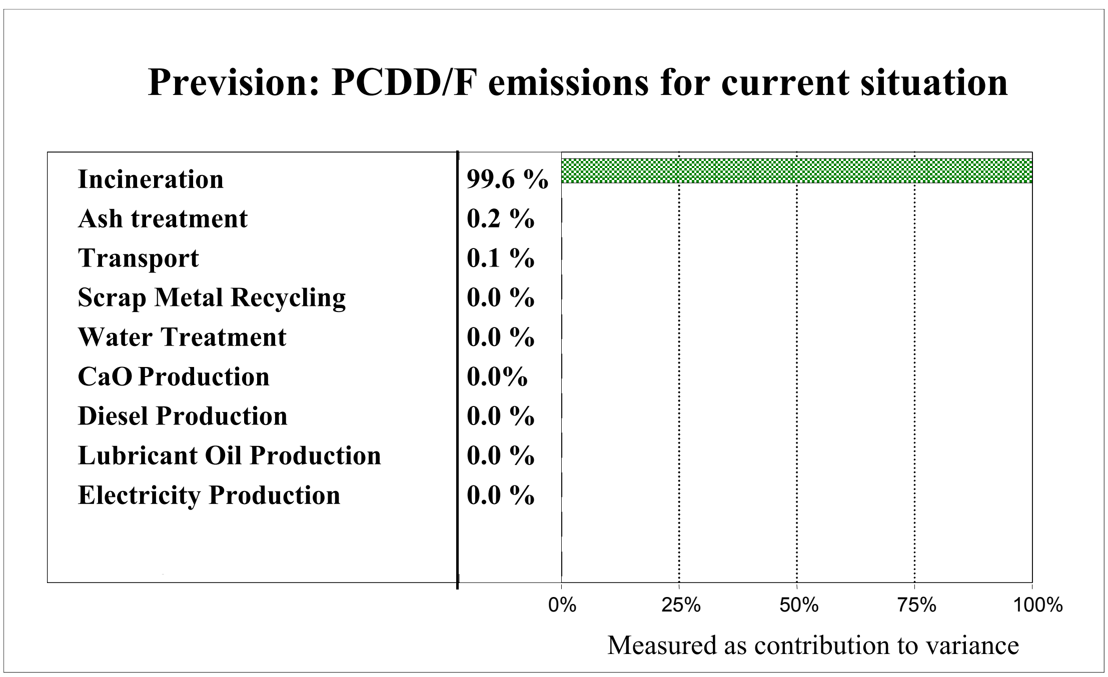


Figure 4.19: Sensitivity analysis of the LCI result for PCDD/Fs in the current situation

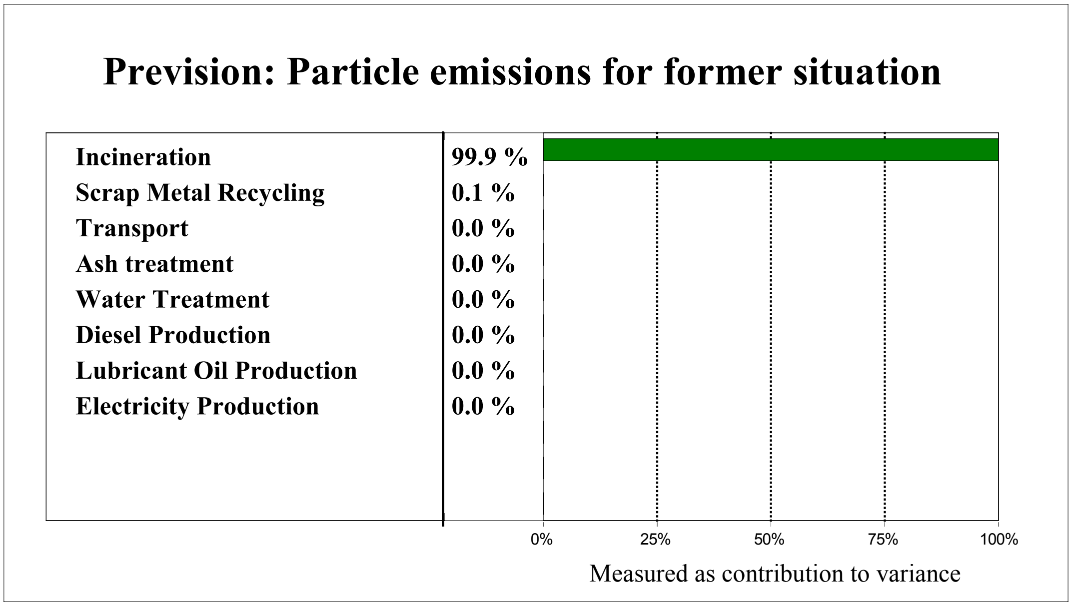


Figure 4.20: Sensitivity analysis of the LCI result for particles in the former situation

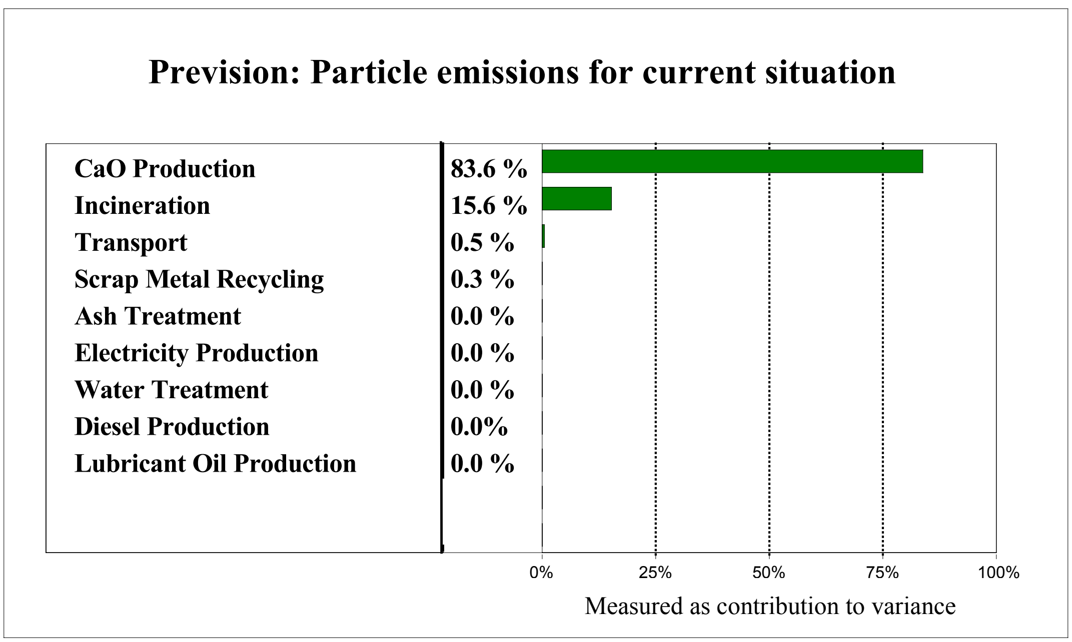


Figure 4.21: Sensitivity analysis of the LCI result for particles in the current situation

The advanced gas treatment system reduces the concentration of heavy metals and PCDD/Fs, particles, SO_2 and HCl in the emissions of the incinerator. The concentrations of the other pollutants as NOx and CO emissions are kept constant. Consequently, from an environmental risk assessment point of view there is a clear reduction of the risk to cause hazardous effects to human health and the environment in the area around the incineration plant. Also on the basis of a life cycle analysis it was possible to observe a clear reduction of the life cycle emissions, per TJ electricity produced, for the pollutants heavy metals, PCDD/Fs, HCl and SO_2, but not for the particles. Moreover, the absolute values show a very slight, even though not significant increase of the life cycle emissions, per TJ electricity produced, for CO_2, CO, and NOx. These life cycle emissions are higher because of the lime production process added to the life cycle, longer transport distances due to higher inflows and outflows and a lower efficiency in the production of the electricity (former situation 158.56 TJ/yr., current situation 149.55 TJ/yr.).

4.7 Application of the Framework to the Impact Pathway Analysis of the Waste Incinerator Emissions on a Local Scale

The framework for uncertainty assessment in IPA established in section 4.4.3 has been applied to the mentioned case study of local human health impacts due to the emissions of the municipal solid waste incinerator in Tarragona (see section 3.4.4) and is presented according to the following steps:

1. Assign probability distributions to the considered parameters
2. Assess the uncertainties and variations in the application
3. Determine the most relevant parameters in such an IPA by sensitivity analysis

1) Assign probability distributions to the considered parameters

As said by Rabl & Spadaro (1999) the probability distributions that are mainly used in environmental damage estimations are the normal distribution and the lognormal probability distribution. All normal distributions are symmetric and have bell-shaped density curves with a single peak. The lognormal distribution is calculated assuming that the logarithm of the variable has a normal distribution.

The proper determination of the probability distribution is only possible if measured data are extensively available, as in the case of emissions, the electricity production, working hours and the flow gas volume. If the parameters are based on little proper information literature values have to be applied to determine the probability distribution. This is the case for the emissions of PCDD/Fs, dispersion modelling results, dose-response and exposuire-response functions, population data and monetary valuation.

The necessary information for determining these probability distributions from measured data was taken from the LCA study of the STQ (1998) and from information given by the director of the MSWI (Nadal, 1999).

Probability distributions published in the literature were available from the ExternE project (EC, 1995; EC, 2000) and in particular from the publication on uncertainty analysis of environmental damages and costs by Rabl & Spadaro (1999). The estimate for the uncertainty of the dispersion model was taken from McKone & Ryan (1989). Further local information originates from the Public Health Plan of the Tarragona region (GenCat, 1997) and from a diagnosis on the socio-economic development of the Tarragona province by Soler (1999).

The way to calculate the probability distribution from a huge amount of experimental data shall be explained here with an example:

The software Crystal Ball facilitates the adjustment of sample points in a density function. The Figure 4.22 shows the variation of such sample points for different measurements of cadmium emissions. In the diagram 17 measurements are classified according to its range of concentration. The most frequent value is the 10 µg/Nm^3 that appears four times. However, there are also two samples with more than 35 µg/Nm^3. The presented variation has been adjusted with both the normal and the lognormal distribution. The different curves make evident that the lognormal distributions fits much better the variation of the measurements.

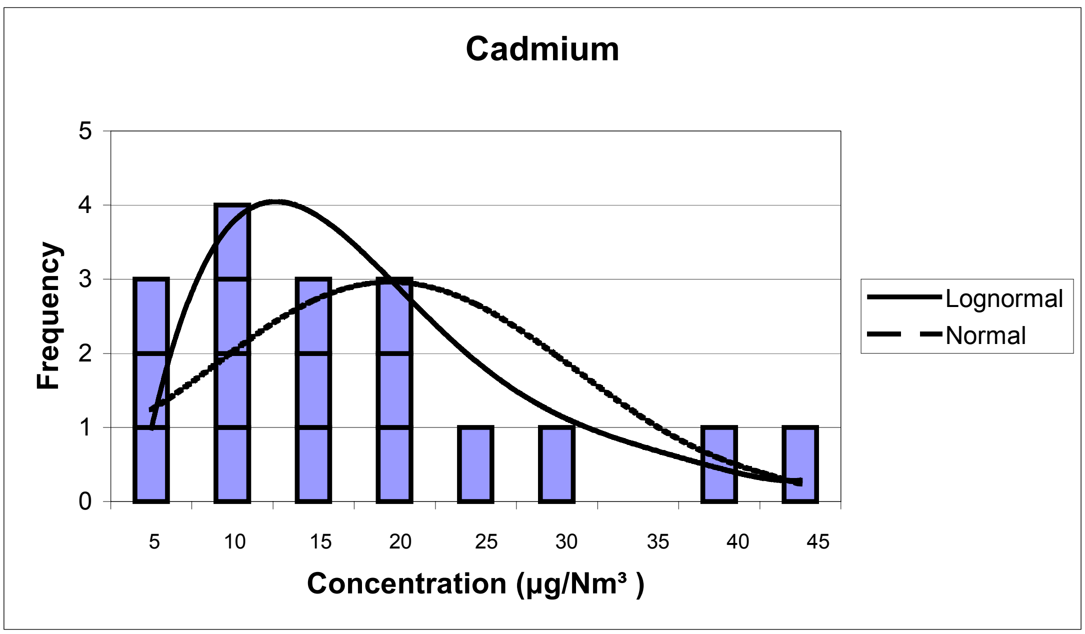


Figure 4.22 Lognormal distribution of cadmium emissions by adjustment of sample points in a density function

In the case study, the variation of the pollutant concentrations in the incinerator emissions is huge due to the heterogeneity of the incinerated waste. Its elementary composition varies strongly. There are often low cadmium concentrations, but sometimes they are high due to elevated cadmium amount in the waste. Thus the measured emissions are not constant over time and follow a lognormal distribution.

In Table 4.4 to Table 4.6 the probability distributions used in this study are summarised. A variable mean stands for a huge number of values that are not constant but differ depending on the grid and pollutant considered.

In Table 4.4 the technology and modelling parameters are presented together with its respective probability distributions and characteristics.

Table 4.4: Technology and modelling data (emission values for former situation 1)

Parameter	Units	Distribution	Mean	Dev.	Source
Electricity production	MW	Normal	5.02	(σ) 0.23	Nadal (1999)
Working hours per year	h	Normal	8,280	CV 0.05	Nadal (1999)
Flue gas volume	Nm^3/h	Normal	90,000	CV 0.05	Nadal (1999)
SO_2 (Emissions)	mg/Nm^3	Lognormal	81.13	(σ_g) 1.5	STQ (1998)
NOx (Emissions)	mg/Nm^3	Lognormal	191	(σ_g) 1.5	STQ (1998)
Particles (Emissions)	$mgNm^3$	Lognormal	28.57	(σ_g) 2.1	STQ (1998)
CO (Emissions)	mg/Nm^3	Lognormal	40	(σ_g) 1.5	STQ (1998)
As (Emissions)	$\mu g/Nm^3$	Lognormal	15.1	(σ_g) 3.4	STQ (1998)
Cd (Emissions)	$\mu g/Nm^3$	Lognormal	19.9	(σ_g) 1.7	STQ (1998)
Ni (Emissions)	$\mu g/Nm^3$	Lognormal	33.27	(σ_g) 2.2	STQ (1998)
PCDD/F (Emission)	ng/Nm3	Lognormal	2	(σ_g) 2	STQ (1998); Rabl & Spadaro (1999)
Flue gas Temperature	K	Point estimate	503	-	Nadal (1999)
Stack height	m	Point estimate	50	-	Nadal (1999)
Stack diameter	$.m^2$	Point estimate	1.98	-	Nadal (1999)
Anemometer height	m	Point estimate	10	-	Nadal (1999)
Geographical Latitude	º	Point estimate	41.19	-	Nadal (1999)
Geographical Longitude	º	Point estimate	1.21	-	Nadal (1999)
Elevation at site	m	Point estimate	90	-	Nadal (1999)
Incremental Immission concentration	mg/Nm^3	Lognormal	variable	(σ_g) 2	McKone & Ryan (1989)

CV - Coefficient of Variation σ – Normal standard deviation σ_g – Geometric standard deviation
Dev. - Deviatio

The technology parameters in Table 4.4 consist of the electricity production, working hours and specific characteristics of the incinerator (stack dimensions, geographical situation, etc.) and the emissions (concentration of pollutants, total volume, temperature, etc.). As modelling parameter the increment of the immission concentration is considered. According to the probability distributions obtained using Crystal Ball, the variations of the electricity production, the working hours and the flow gas volume are normal distributed and the emissions behave like cadmium in a lognormal way. The electricity production has a normal standard deviation of 0.23; working hours and flue gas volume have respectively a coefficient of variation of 0.05. The variations of the emissions are characterised by geometric standard deviations from 1.5 to 3.4. The parameters corresponding to the stack dimensions and the geographical situation are point estimates given by the director of the MSWI (Nadal, 1999). The incremental immission concentration is lognormal distributed with a geometric standard deviation of 2 according to the uncertainty estimates for the dispersion model by McKone & Ryan (1989).

In Table 4.5 the human health parameters are presented.

Table 4.5: Impact Human Health data

Parameter	Units	Distribution	Mean	Dev.	Source
Dose-response and exposure-response functions					
Chronic YOLL		Lognormal	0.00072	(σ_g) 2.1	IER (1998); Rabl & Spadaro (1999)
Acute YOLL		Lognormal	variable	(σ_g) 2.1	Rabl & Spadaro (1999)
Cancer		Lognormal	variable	(σ_g) 3	Rabl & Spadaro (1999)
Others		Lognormal	variable	(σ_g) 2.1	Rabl & Spadaro (1999)
Damage factors					
Chronic YOLL		Lognormal	1	(σ_g) 1.5	Rabl & Spadaro (1999)
Acute YOLL		Lognormal	1	(σ_g) 4	Rabl & Spadaro (1999)
Cancer		Lognormal	1	(σ_g) 1.6	Rabl & Spadaro (1999)
Others		Lognormal	1	(σ_g) 1.2	Rabl & Spadaro (1999)
% pop above 65 years	%	Point estimate	13		IER (1998); GenCat (1997)
% pop adults	%	Point estimate	57		IER (1998); GenCat (1997)
% pop children	%	Point estimate	24		IER (1998); GenCat (1997)
% pop asthma adults	%	Normal	4	CV 0.1	IER (1998); GenCat (1997)
% pop asthma children	%	Normal	2	CV 0.1	IER (1998); GenCat (1997)
% pop baseline mortality	%	Normal	0.864	CV 0.1	IER (1998); GenCat (1997)
Population	# hab	Normal	variable	CV 0.01	Soler (1999)

CV - Coefficient of Variation σ_g – Geometric standard deviation Dev. – Deviation

As can be seen in Table 4.5, uncertainty and variation are part of the public health data (like the population, the percentage of the children, adults and ancients, the percentage of asthmatics or the baseline mortality). The dose-response and exposure-response functions are characterised by a lognormal probability distribution provided by Rabl & Spadaro (1999). The mean value for chronic YOLL due to particles originates from IER (1998), the other means are varying in function of the respective pollutants. The definition and calculation of the damage factors (e.g. chronic YOLL; acute YOLL and cancer) involves another factor of uncertainty. The probability distributions for these factors, which are only used for aggregation and further multiplication (therefore equal to 1), have been taken from Rabl & Spadaro (1999) where they were supposed to be lognormal with a geometric standard deviation between 1.2 and 4. The description of the population properties has been identified to be a point estimate or normal distributed according to the values provided by IER (1998) and GenCat (1997). The possible variation of the number of habitants in each grid can be described by a normal distribution based on the study made by Soler (1999).

In the Table 4.6 the monetary valuation parameters are shown. All probability distributions used are lognormal and were taken from Rabl & Spadaro (1999).

Table 4.6: Monetary Valuation data

Parameter	Units	Distribution	Mean	Dev.	Source
Chronic YOLL	Euro	Lognormal	84,330	(σ_g) 2.1	Rabl & Spadaro (1999)
Acute YOLL	Euro	Lognormal	155,000	(σ_g) 2.1	Rabl & Spadaro (1999)
Cancer	Euro	Lognormal	1,500,000	(σ_g) 2.1	Rabl & Spadaro (1999)
Others	Euro	Lognormal	Variable	(σ_g) 1.2	Rabl & Spadaro (1999)

σ_g – Geometric standard deviation Dev. – Deviation

2) Assess the uncertainties and variations in the application

Following the framework proposed, a final result expressed in environmental damage costs due to the air emissions per kWh of electricity produced has been calculated for both situations considered. By the use of the obtained probability distributions for the essential parameters in a Monte Carlo Simulation, the result for environmental damage costs has been transformed from a concrete value into a probability distribution around a mean value. Due to the fact that the distributions of most parameters in Table 4.4 to Table 4.6 are lognormal and not normal, the final distribution of each result has got a lognormal distribution, too.

In the same way as in the application to LCI, each simulation has been made in one separate run per situation. Because of the inherent variability of the Monte Carlo model, it is not possible to affirm that the set of values of the input variables used in the run of the situation 1 are going to be the same in the situation 2. Because of the generation of random numbers every run occurs in a different way. In order to verify the importance of this variability on the final outcome, both simulations have been also made in one run, and it could be checked that the result is the same; the variations are negligible.

In the Figure 4.23 the first result is presented, it is the case of the incinerator with advanced acid gas treatment system. The mean environmental damage cost in this situation 2 is 0.87 mEuros per kWh (1E-3 €/ kWh). In the x-axis, it is possible to observe the environmental damage cost per energy output. The y-axis shows the probability of each cost value. The total number of iterations carried out with the software Crystal Ball is 10,000. A summary of all the results generated can be found in the Table 4.7. The geometric standard deviation for this result is 2.62.

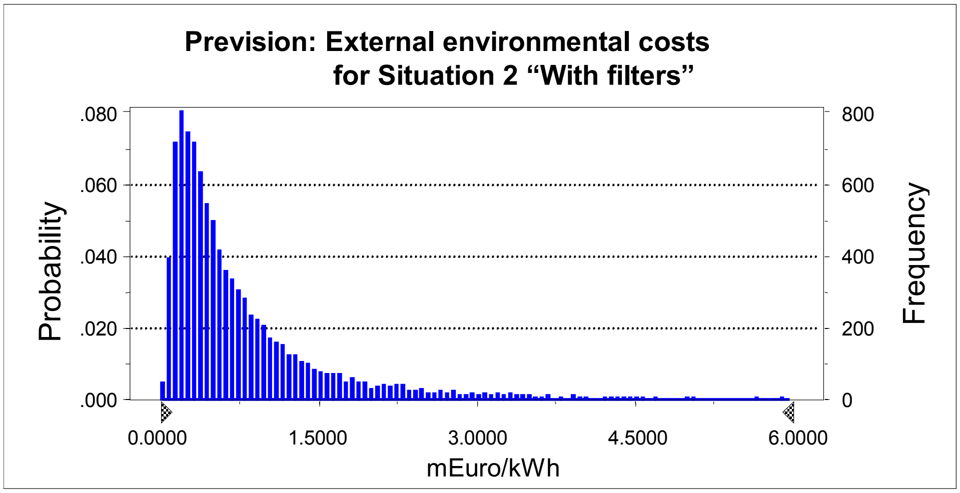


Figure 4.23: MC Simulation results for IPA of MSWI emissions in the current situation 2

Table 4.7: Statistical parameter describing the results of the environmental damage estimation in the situations 1 and 2 as external costs in mEuros per kWh (1E-3 €/ kWh)

Parameter	Situation 1 "Without filters"	Situation 2 "With filters"
Normal mean	3.73	0.87
Normal standard deviation	5.16	1.08
Geometric mean	2.19	0.55
Geometric standard deviation	2.81	2.62
Minimum	0.087	0.029
Maximum	221.7	74.4
Median	2.09	0.53
68% confidence interval		
Superior	6.15	1.44
Inferior	0.78	0.21

The second case is previous in time to the first one, when the incinerator did not have an advanced acid gas treatment system. The emissions of pollutants were more important, and consequently also the environmental damage cost is with 3.73 mEuros per kWh (1E-3 €/ kWh) much higher. The probability distribution of this result can be found in the Figure 4.24. The geometric standard deviation is with 2.81 nearly the same as in the other result presented because the only important change consists in the mean values of ten parameters, including the pollutants and the electricity production, which is less with an advanced acid gas treatment system installed. The overview of the values calculated is illustrated in Table 4.7.

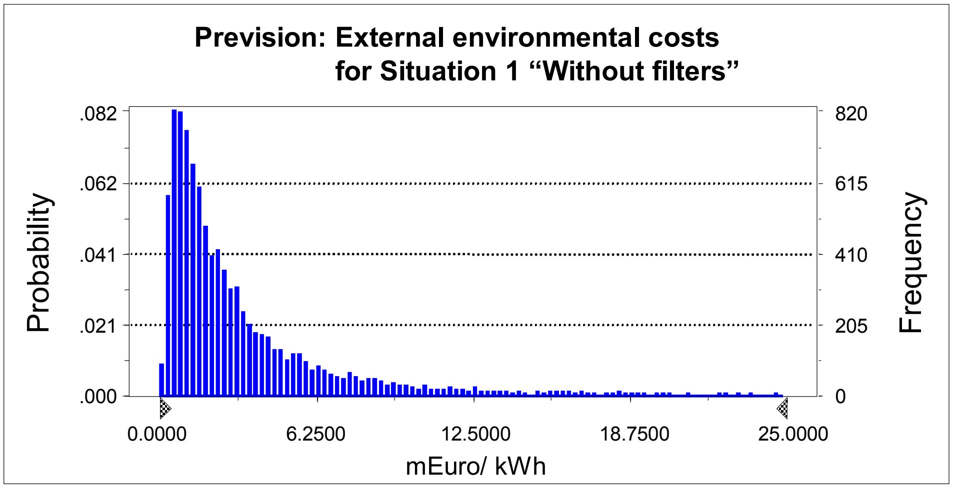


Figure 4.24: MC Simulation results for IPA of MSWI in the former situation 1

These results show that the uncertainty and variability calculated using Monte Carlo Simulation is less than by analytical methods due to the dynamic characteristics of this stochastic model. The presented results have a geometric standard deviation of less than 3, whereas the geometric standard deviation obtained by analytical methods is bigger than 4 according to Rabl & Spadaro (1999).

The Figure 4.25 illustrates the differences between the means of situation 1 and 2 and its confidence intervals of 68%. It is possible to observe a clear reduction of the damage cost. As can also be seen in Table 4.7, the 68 % confidence interval for the situation with advanced gas treatment system embrace the range from 0.21 to 1.44 mEuros per kWh while for the other situation the same confidence interval has the range between 0.78 and 6.15 mEuros per kWh. The inferior bound of the confidence interval is the maximum range of possible errors; for the result of the situation 1 the inferior bound is in the same order of magnitude as the mean for the situation 2. Hence, according to the results there are important uncertainties. However, a clear reduction of the damage cost can be foreseen within a confidence interval of 68 % when comparing the two different operation scenarios.

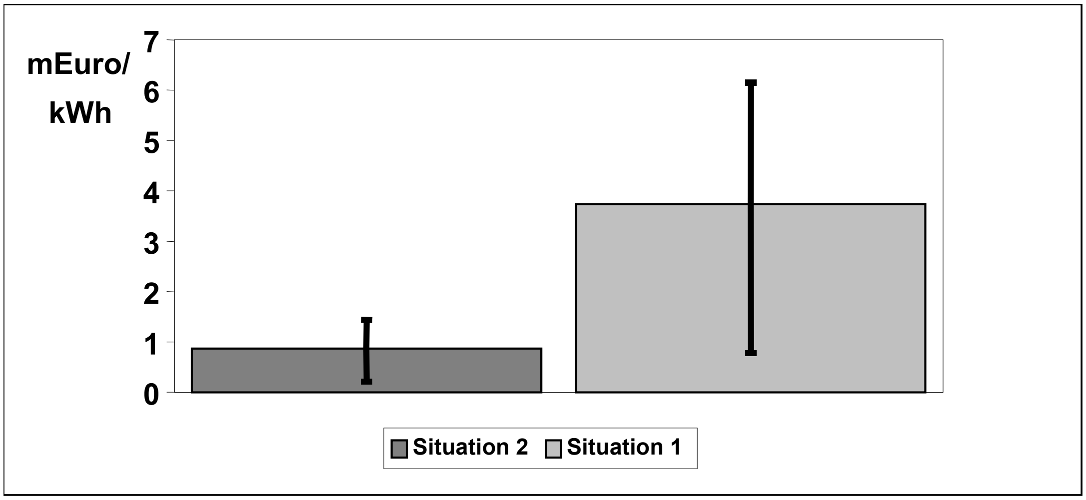


Figure 4.25: Comparison of the MC simulation results for the former situation 1 without advanced gas cleaning system and the current 2 with such an installation

3) Determine the most relevant parameters in such an IPA by sensitivity analysis

The results of the sensitivity analysis for the pollutants are presented in the Figure 4.26 for the second situation with advanced acid gas treatment. Some interesting results can be extracted. The graph shows all the pollutants and its contribution to the final result. Obviously, the emission of particles is the most important parameter, 91.5% of the total damage is caused by the particles. The NOx seems to be the second important pollutant, while the rest of them produce a negligible damage. The Figure 4.27 for the situation 1, i.e. without advanced gas treatment system, is very similar; the particle emissions contribute with more than 95 % to the total environmental damage cost. Because of the major emissions of particles, in the former situation 1 the mentioned percentage of NOx is practically negligible. Taking into account that the emission source is a municipal solid waste incinerator, with the mayor public preoccupation of producing dioxins, the result of the IPA for the case study shows that they only contribute little to the total human health damage by the air emissions.

The Figure 4.28 and Figure 4.29 present the results of the sensitivity analysis on the sources of health impacts to total damage costs. Most damage is caused by the lost of life expectancy, expressed as YOLL (Years of Life Lost). If the damage appears in a near-term, it is called acute, and if the damage turns up to appear in a long-term it is called a chronic impact. The chronic YOLL and the acute YOLL account together for more than 99% of the total environmental damage costs. Other parameters like hospital admission or cancer are by far less important. The Figure 4.29 with the results for the situation 1 without advanced acid gas removal system is a little bit different from the Figure 4.28 because of the major concentration of particles. The particles have more influence on the chronic YOLL, and an increase in its concentration produces the increase in the importance of the chronic YOLL.

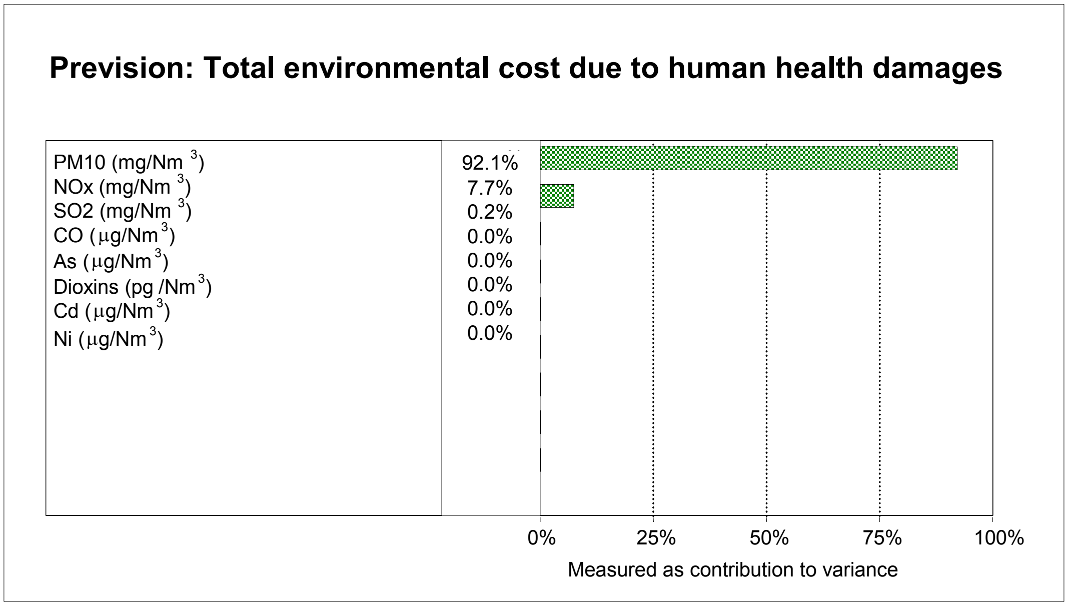


Figure 4.26: Sensitivity analysis for pollutants of the current situation 2

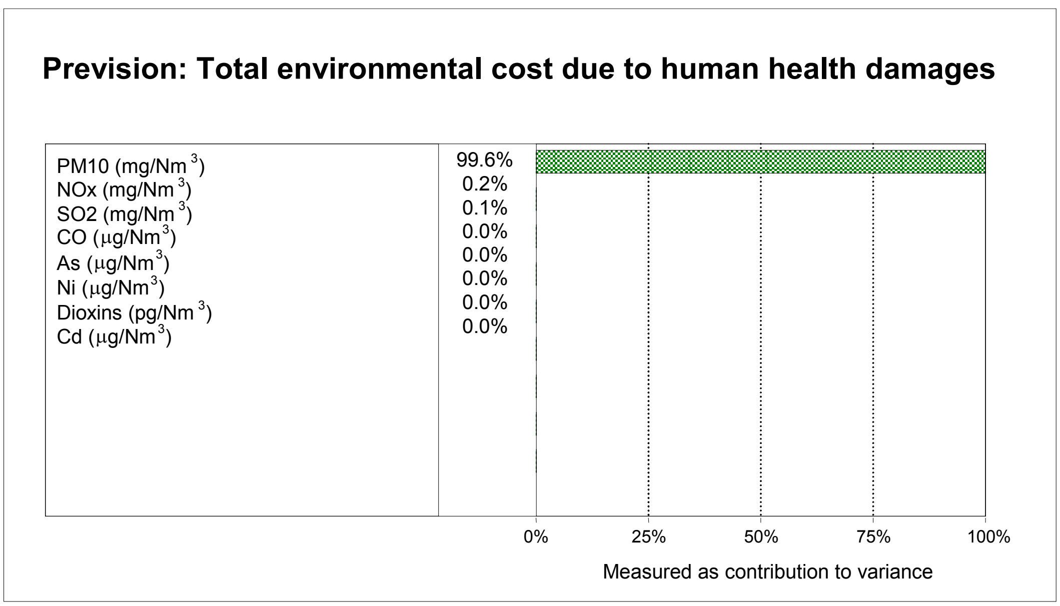


Figure 4.27 Sensitivity analysis for pollutants of the former situation 1

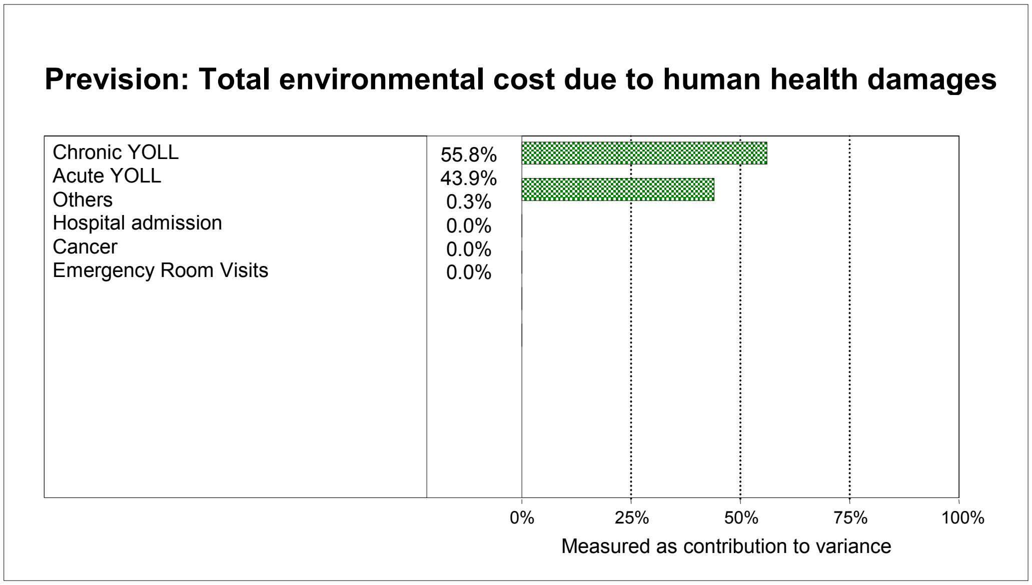


Figure 4.28: Sensitivity analysis for health impacts of the current situation 2

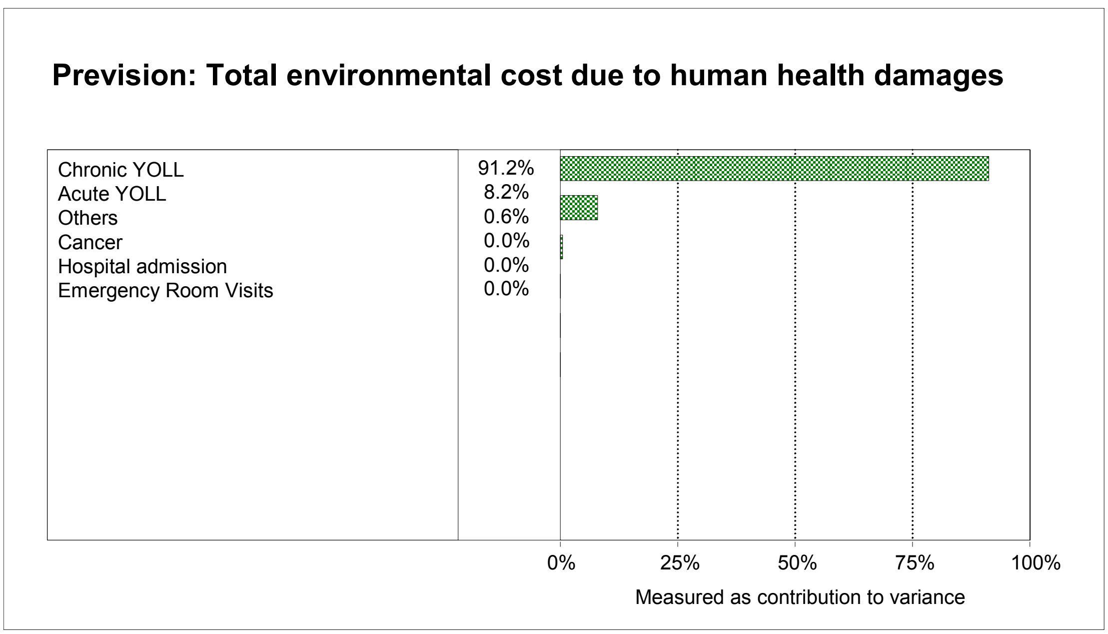


Figure 4.29: Sensitivity analysis for health impacts of the former situation 1

4.8 Comparison of the uncertainties in the Life Cycle Inventory and the Impact Pathway Analysis

In general, it can be said that the damage estimations carried out in the IPA are containing more uncertainties than the LCI due to the very important uncertainties related to the dispersion models, dose-response and exposure-response functions and the weighting schemes. That means that especially the effect analysis contains the parts with the highest uncertainties what corresponds to the parts that belong to the valueshere according to Hofstetter (1998).

In detail, the results for the LCI show a geometric standard deviation between 1.13 for CO_2 or 1.29 for SO_2 and 1.92 heavy metals or 2.3 for As. That means that the higher uncertainties are related to the variability in the waste input composition regarding trace elements. In comparison with the LCI results, the uncertainties in the environmental damage estimations in form of external costs are higher and sum up to a geometric standard deviation of 2.62 with filters and 2.81 without filters. These relatively high geometric standard deviations are less influenced by the emissions (except for As with a σ_g of 3.4, for Ni with a σ_g of 2.2 and for the particles with a σ_g of 2.1) than by the impact human health data, especially dose-response functions for cancer (σ_g 3) and damage factor of acute YOLL (σ_g 4). Moreover, monetary valuation for YOLL and cancer is an important source of uncertainties with a σ_g of 2.1 in the same way as the dispersion model with a σ_g of 2.

To overcome the incomparability related to the dispersion models it is proposed to use an internationally accepted reference model. Using in the future homogeneously dose-response and exposure-response functions approved by the WHO could have the same effect.

5 Methodology of Environmental Damage Estimations for Industrial Process Chains

Due to the problems observed in the current methods for environmental impact assessment in industrial process chains as described in chapter 2, a methodology has been developed that allows to estimate environmental damages for industrial process chains. Industrial process chains are understood here as chains of industrial processes with less than 100 processes. There is a need for a framework that allows to evaluate environmental damages in an accurate way as far as possible because nowadays used damage assessment methods generates different results than the evaluation of potential impacts. This is demonstrated for instance by the study undertaken by Spirinckx & Nocker (1999) that compare both approaches. This methodology is useful for certain Life Cycle Management applications, like end-of-life strategies and supply chain management. Possible applications are further discussed in section 5.6.

5.1 Challenges and strategy for a new methodology

The bases of life cycle inventories are the emissions of pollutants and the consumption of resources. In this methodology, the focus is on pollutant emissions and the damages that they may cause. After its emission pollutants are transported through the environment and cause a concentration increase. On its pathway they then can affect sensible receptors, as humans, and may so produce damages. The receptor density is clearly depending on local or regional geographic characteristics for non-global impact categories. These environmental damages can be evaluated an aggregated according to socio-economic evaluation patterns as indicators or external costs. The methodology makes a step out of the LCA framework and integrates other environmental tools, according to the idea of CHAINET (1998). Such a methodology is confronted with the following special challenges:

- Consider each or at least the main processes
- Find a compromise between accuracy and practicability
- Apply the damage functions as far as possible to the emissions in the respective continent, region or location

- Aggregate the damages by economic evaluation or other forms of weighting to a small number of indicators
- Show transparency, analyse uncertainties and sensitivity

First of all a general strategy is necessary with regard to the environmental damage estimations for industrial process. This strategy includes an approach to make the methodology more practicable. Starting with a conventional life cycle inventory, such a strategy can be described by:

1. Create an algorithm to consider site-specific aspects
2. Calculate the potential impact score
3. Estimate global damages by best available midpoint indicator(s)
4. Determine main media, pollutants and processes
5. Use fate models to obtain the concentration increment in the respective region (s)
6. Relate increments with dose-/ exposure-response functions and receptors
7. Dispose of methods for aggregation by accepted weighting schemes
8. Relate it to other environmental management tools

5.2 MATHEMATICAL FRAMEWORK

The basis for the general strategy described so far is a mathematical framework that enables to carry out the further steps. It includes, especially, a procedure that allows to know the quantity, situation and moment of the generation of the specific environmental interventions in the life cycle inventory.

As introduced in chapter 2, Castells et al. (1995) have presented an algorithm that uses an eco-vector. The different types of environmental loads like chemical substances can be classified in categories according to their environmental impacts, and then different impact potentials can be calculated by the characterisation factors presented in Heijungs et al. (1992). For example CO_2, CH_4, CFC-11 and others can be aggregated in the Global Warming Potential (GWP) and be expressed as CO_2-equivalents. The different chemicals are characterized by a specific weighting factor, i.e. the eco-vector $\mathbf{e_v}$ is converted into a weighted eco-vector $\tilde{\mathbf{e}}_{\mathbf{v}}$. See expression (5.1), where e_v^i is the specific [EL/kg] of the EL_i, $\tilde{e}_v^i$ is the specific weighted [EL_{eq}/kg] of the EL_i, and λ^i is the specific weighting factor of the EL_i.

$$\tilde{\mathbf{e}}_{\mathbf{v}} = \tilde{e}_v^i \quad = \quad e_v^i \ \lambda^i \qquad (5.1)$$

According to the goal of the general strategy the eco-vector algorithm has to be changed in order to make possible the assessment of the actual impacts caused by a specific process in its particular environment. Considering the example of a process chain with three processes, each process (PR) consumes raw material (RM) or an intermediate product (IP) and generates emissions and/ or waste (SO_2, PM10, Cd, etc.) to obtain the functional unit, the product. When the life cycle inventory analysis is applied to the three processes an eco-vector with the environmental interventions for all the three processes is obtained, illustrated in Figure 5.1. It is evident that the following LCIA generates only potential impacts because all site-specific information is lost when the LCIA is carried out.

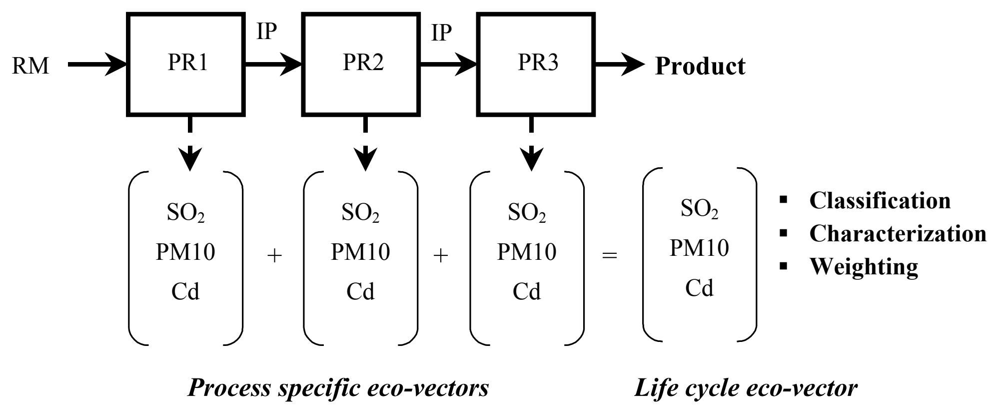


Figure 5.1: Life Cycle Inventory Analysis according to the eco-vector principle (IP – Intermediate Product, PR – Process, RM – Raw Material)

Another approach to carry out the environmental assessment of a functional unit consists in an analysis of the actual impacts of each process, according to a site-specific damage estimation concept (Figure 5.2). Consequently, the environmental loads of each process are also accounted, but the evaluation is carried out for each process in its specific region. Each assessment contains the three consecutive procedures of fate analysis, application of impact factors and weighting across the impact categories. The results of each process assessment can be summed up if they are expressed in monetary units or by the same indicators. A damage profile is provided.

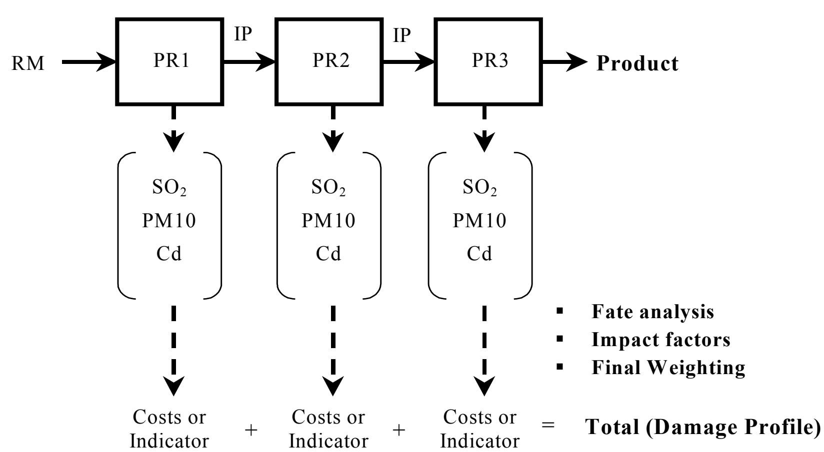


Figure 5.2: Determination of damage estimations by site-specific assessment (IP – Intermediate Product, PR – Process, RM – Raw Material)

In order to express this different method in an algorithm that is based on the same principles as introduced before, the eco-vector has to be transformed into an eco-technology matrix $\mathbf{E_M}$. In the eco-technology matrix, similar to the technology matrix mentioned by Heijungs (1998), there are M columns for M linear processes, and there are N rows for the N environmental loads as of the eco-vector. In the example there are three columns for the three processes and there are three rows for the three environmental loads SO_2, PM10 and Cd. See expression (5.2) for an illustration. It is evident that the environmental loads have to be in the same order for all the processes and that a certain environmental intervention always has to be expressed in the same unit.

$$
\begin{array}{ccccc}
\textbf{Process 1} & \textbf{Process 2} & \textbf{Process 3} & \textbf{Process ...} & \textbf{Process m} \\
\mathbf{e_{v1}} = \begin{pmatrix} SO2 \\ PM10 \\ Cd \\ X_{...} \\ X_n \end{pmatrix} &
\mathbf{e_{v2}} = \begin{pmatrix} SO2 \\ PM10 \\ Cd \\ X_{...} \\ X_n \end{pmatrix} &
\mathbf{e_{v3}} = \begin{pmatrix} SO2 \\ PM10 \\ Cd \\ X_{...} \\ X_n \end{pmatrix} &
\mathbf{e_{v...}} = \begin{pmatrix} SO2 \\ PM10 \\ Cd \\ X_{...} \\ X_n \end{pmatrix} &
\mathbf{e_{vm}} = \begin{pmatrix} SO2 \\ PM10 \\ Cd \\ X_{...} \\ X_n \end{pmatrix} \\
\Downarrow & \Downarrow & \Downarrow & \Downarrow & \Downarrow
\end{array}
$$

$$
\mathbf{E_M} = \begin{pmatrix}
SO2_1 & SO2_2 & SO2_3 & SO2_{...} & SO2_m \\
PM10_1 & PM10_2 & PM10_3 & PM10_{...} & PM10_m \\
Cd_1 & Cd_2 & Cd_3 & Cd_{...} & Cd_m \\
X_{...,1} & X_{...,2} & X_{...,3} & X_{...,...} & X_{...,m} \\
X_{n,1} & X_{n,2} & X_{n,3} & X_{n,...} & X_{n,m}
\end{pmatrix} \downarrow \textbf{N environmental loads} \qquad (5.2)
$$

$$\xrightarrow{\textbf{M processes}}$$

In analogy to the multiplication of the eco-vector by a weighting vector, the eco-matrix can be multiplied by another matrix, the weighting or damage-assigning matrix $\mathbf{W_M}$, which contains the fate analysis, the impact factors and the final weighting. This matrix assigns to each process the damage cost or another damage indicator caused by one specific EL_i. In $\mathbf{W_M}$ there are N columns for the N environmental loads; and there are M rows for the M linear processes. In this way an M x M matrix is obtained, the weighted eco-technology matrix $\mathbf{WE_M}$, expression (5.3).

$$\mathbf{W_M} \;*\; \mathbf{E_M} \;=\; \mathbf{WE_M} \qquad (5.3)$$

The main magnitude will be the trace D of $\mathbf{WE_M}$, expression (5.4), where $E_M{}^{ij}$ is the specific [EL/kg] of the EL_i in the process j, $W_m{}^{ji}$ is the specific damage factor for the EL_i in the process j, and D^j is the environmental cost or another indicator of the process j. D expresses either the total environmental damage cost of the life cycle, if $\mathbf{W_M}$ represents a weighting by costs, or the damage indicator for the life cycle, if $\mathbf{W_M}$ represents a weighting by an indicator.

$$D = \mathrm{tr}\,(\mathbf{WE_M}) = \sum_{ji} (W_M^{ji} * E_M^{ij}) = \sum_{j}^{M} [\sum_{i}^{N} W_M^{ji} * E_M^{ij}]$$

$$= \sum_{j}^{M} D^{jj} \; \lambda \sum_{j}^{M} D^{j} \qquad (5.4)$$

The algorithm is illustrated in the expression (5.5) for the example of three processes and two environmental loads SO_2 y Cd.

$$M \downarrow \overset{N \rightarrow}{\begin{pmatrix} W_M^{1,\,SO2} & W_M^{1,\,Cd} \\ W_M^{2,\,SO2} & W_M^{2,\,Cd} \\ W_M^{3,\,SO2} & W_M^{3,\,Cd} \end{pmatrix}} \bullet N \downarrow \overset{M \rightarrow}{\begin{pmatrix} E_M^{SO2,\,1} & E_M^{SO2,\,2} & E_M^{SO2,\,3} \\ E_M^{Cd,\,1} & E_M^{Cd,\,2} & E_M^{Cd,\,3} \end{pmatrix}} = M \downarrow \overset{M \rightarrow}{\begin{pmatrix} D^{1} & D^{12} & D^{13} \\ D^{21} & D^{2} & D^{23} \\ D^{31} & D^{32} & D^{3} \end{pmatrix}} \qquad (5.5)$$

$$\text{with} \quad \begin{cases} D^1 = W_M^{1,\,SO2} \bullet E_M^{SO2,\,1} + W_M^{1,\,Cd} \bullet E_M^{Cd,\,1} \\ D^2 = W_M^{2,\,SO2} \bullet E_M^{SO2,\,2} + W_M^{2,\,Cd} \bullet E_M^{Cd,\,2} \\ D^3 = W_M^{3,\,SO2} \bullet E_M^{SO2,\,3} + W_M^{3,\,Cd} \bullet E_M^{Cd,\,3} \end{cases} \quad \Rightarrow \quad D = D^1 + D^2 + D^3$$

The values of the weighted eco-matrix $\mathbf{WE_M}$ in the diagonal D^1, D^2 and D^3 represent the environmental damage cost or damage indicator of the process 1, 2 and 3. If the values of $\mathbf{WE_M}$ are not in the diagonal, as D^{12}, and given the assumption of linearity, they represent the environmental damage cost or damage indicator of the corresponding process, but in a different region, e.g. the damage cost or damage indicator that the process 2 would cause in the region 1. Consequently it is possible to compare the effects of a certain process in different regions. Each component D^{kj} of the weighted eco-matrix is obtained by expression (5.3), written in the components of the matrix it is presented in expression (5.6), where k stands for the region k and j for the process j.

$$D^{kj} = \sum_{i}^{N} (W_M^{ki} * E_M^{ij}) \qquad (5.6)$$

The weighting matrix components for the EL_i corresponding to the different processes j and k are equal if the processes are situated in the same region. Indeed, that is the case for the global impacts as for the GWP where this simplification allows to work with one eco-vector for all the processes and one weighting vector for all regions, expression (5.7). In the case of global weighting factors the weighting matrix has the same components for all processes.

$$W_M^{ji} = W_M^{ki} \quad \Longleftrightarrow \text{region (j) = region (k)} \qquad (5.7)$$

A special topic that has to be considered is the question of mobile processes. If the process is a moving one there may be different regions involved so that the expression (5.8) holds true. By choosing the size of the region the number of regions to consider for the corresponding mobile process are determined.

if there is a mobile process with sufficient transport km

$$\Longrightarrow \text{exists at least 1 i ; } D^{kj} \neq D^{lj} \ \mathrm{V}\ k \neq i \qquad (5.8)$$

This mathematical framework delivers a tool for introducing site-specific aspects in the life cycle approach. The matrix algebra provides an elegant and powerful technique for the derivation and formulation of different tools in a life cycle perspective.

5.3 METHODOLOGY FLOWCHART

5.3.1 OVERVIEW OF THE METHODOLOGY FLOWCHART

Having the mathematical framework at our disposal the next question is to find a way to determine the eco-technology matrix and the weighting or damage-assigning matrix. It is proposed to base the environmental damage estimations of industrial process chains on the results of a conventional LCI Analysis and one or more LCIA methods to answer the environmental management problem of interest.

In this way one or more impact scores are calculated and this information can be used for a selection process in the form of a three-fold dominance analysis. In the three phases of the dominance analysis the main medium, processes and pollutants have to be identified in a combination of quantitative and qualitative evaluation. In principle, such an evaluation should be carried out for each of the selected impact score, since each one considers one particular type of environmental impact or weighting scheme. Then the obtained relevant processes and pollutants are spatially differentiated according to the site or region that should be taken into account in the environmental impact assessment. Finally the eco-technology matrix is elaborated for the predominant processes and pollutants in the assigned sites and regions.

In a next step, the fate & exposure and consequence analysis is carried out with different levels of detail for each process that has been identified to be relevant. The results of the fate & exposure and consequence analysis are the input for the damage assigning matrices.

In the case of global indicators like GWP it is not necessary to perform a fate & exposure and consequence analysis with different levels of detail. Hence, these indicators can be used directly in the damage-assigning matrix. Especially the global warming potential and the ozone depletion potential are considered as global indicators. According to Bare et al. (2000) and Udo de Haes & Lindeijer (2001) these potentials are important for the sub-Area of Protection Life Support Systems which belongs to the Area of Protection (AoP) Natural Environment and might be seen as having intrinsic value in their own right. The Life Support Functions concern the major regulating functions, which enable life on earth (both human and non-human). These particularly include the regulation of the earth climate, hydrological cycles, soil fertility and the bio-geo-chemical cycles. In the same way, depletion of the sub-AoP natural resources (abiotic, biotic and land) can be taken into account if the decision-maker considers these potentials as important.

The flowchart of Figure 5.3 gives an overview of the procedure to generate the eco-technology matrix and the damage-assigning matrix. The multiplication of the matrixes yields the damage profile of each considered alternative and interesting information for the optimisation of process settings. In the case of using different impact scores the same damage endpoints considered in the fate & exposure and consequence analysis related to these scores have to be summed up.

The damage profile can be divided in damages to human health (mortality, cancer and morbidity), man-made environment, natural environment and global indicators (GWP, etc.). The application of a weighting and aggregation scheme, determined in the goal and scope definition avoids a multi-criteria analysis of a huge amount of impact parameters, as for instance emergency room visit (ERV), asthma attacks, maintenance surface for paint, and yield loss of wheat, that are the result of the site-specific environmental evaluations.

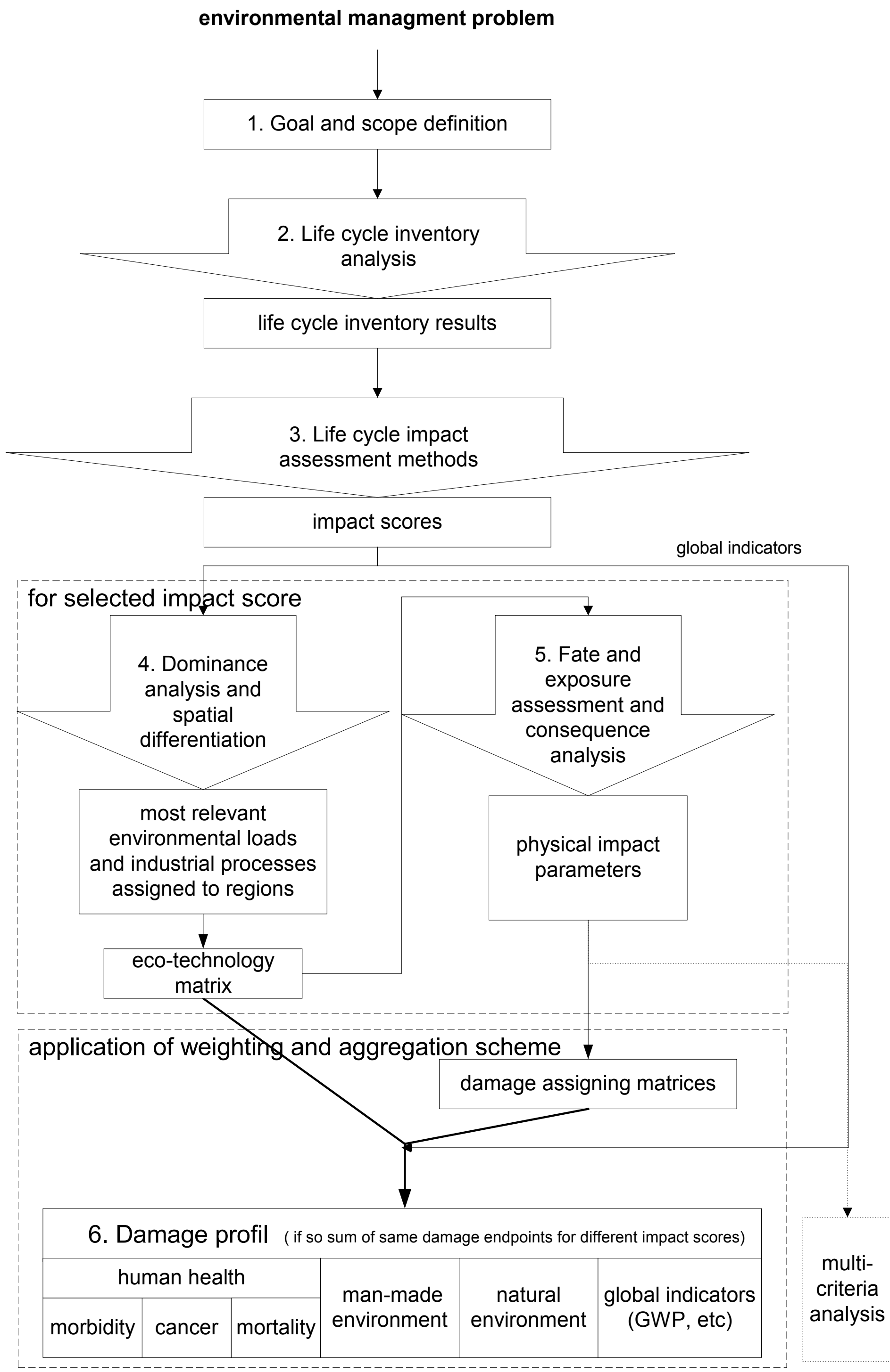


Figure 5.3 Overview of the procedure to generate the eco-technology matrix and the damage-assigning matrix

5.3.2 GOAL AND SCOPE DEFINITION

In the goal and scope definition (Figure 5.4) the decision-maker determines the cornerstones of the environmental damage estimations for industrial process chains that in his opinion best fit to answer the environmental management problem of interest. Of course he has to do this taking into account budget restrictions.

In the goal definition it has to be decided which situations or scenarios will be assessed and compared. Here situations are referring to existing process chains while scenarios means process chain options for the future (Pesonen et al., 2000).

In the scope definition the decision-maker has to select the functional unit (e.g. 1 TJ electricity or 1 kg waste treated), the initial system boundaries (e.g. 1 % contribution to functional unit) and the LCIA characterisation potentials (e.g. GWP, AP, NP, etc.) and/ or single index methods (e.g. Eco-indicator 95, EPS, etc.). These requirements correspond to those for LCAs according to the ISO 14040 series.

However, in the methodology of environmental damage estimations for industrial process chains more information is necessary to outline the study. These decision points are obligatorily the initial cut-off criteria (e.g. site-specific 5% and literature values 1%) and the weighting and aggregation scheme. Optionally it has to be decided if an uncertainty analysis should be included, if accidents should be considered and if the eco-efficiency of the process chain should be calculated.

Initial cut-off criteria have to be defined for the dominance analysis. These cut-off criteria serve to determine which media, processes and pollutants have to be further studied in the fate & exposure and consequence analysis and in which way, e.g. site specific or by literature values.

For the weighting the decision-maker can follow the general decision tree presented in Figure 5.5. There are different options to evaluate the environmental loads. It depends on the worldview of the decision-maker which options he prefers.

For the environmental loads that cause a global impact GWP, ODP and other global indicators can be calculated. In that case these potentials have environmental relevance in the form of life support functions and depletion of natural resources if this is considered a problem. First the decision-maker has to select the environmental impacts he considers relevant for the environmental management problem under study, then he has to decide if according to the knowledge that is available for him the damages related to these potentials are estimable. In the case that he thinks that these damages cannot be estimated with acceptable reliability he has to make up his mind for each global indicator if he prefers to monetise the potential impact using abatement costs or to express it directly as physical impact potential, e.g. CO_2-equivalent. In the case that he believes the damages to be estimable they can be assessed in conjunction with the other environmental loads.

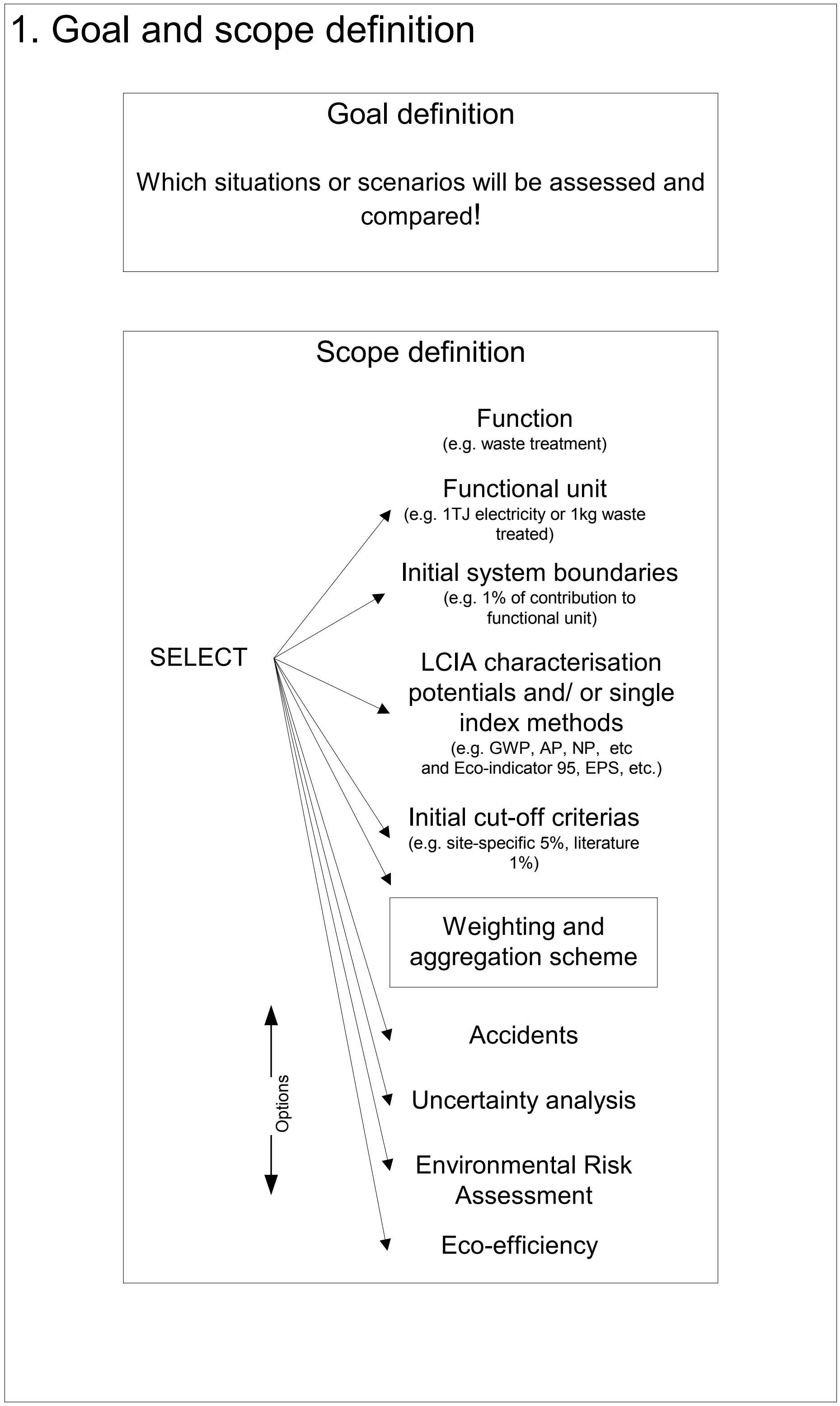


Figure 5.4: Goal and scope definition

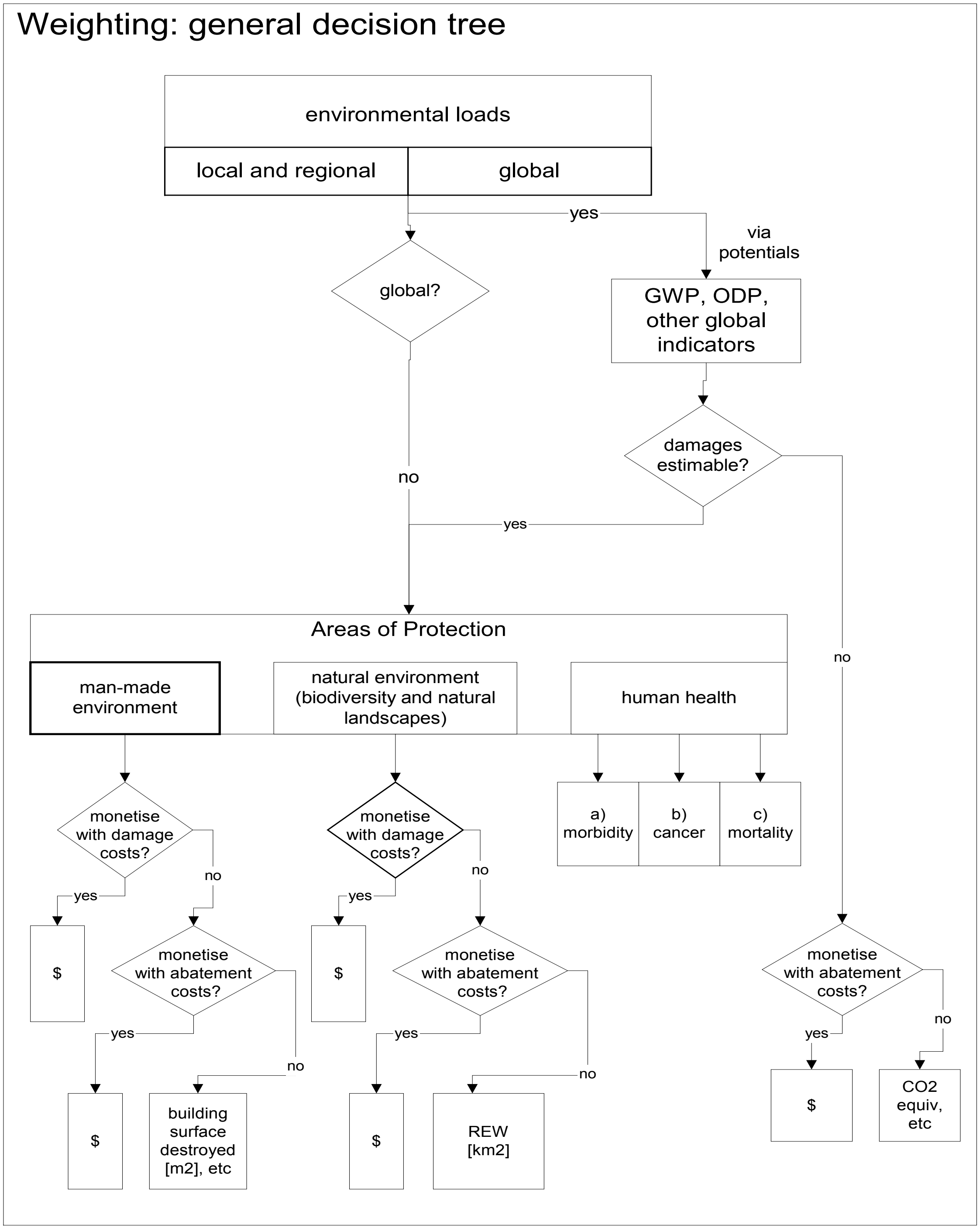


Figure 5.5: General decision tree for weighting

The other environmental loads may cause local and regional damages. These damages can be divided into the AoPs published by Udo de Haes et al. (1999): man-made environment, natural environment and human health. In the case of man-made environment and natural environment the following questions to decide on are the same. Due to the complexity of the weighting options for the AoP human health another decision tree has been drawn and is presented in Figure 5.6.

For the AoPs man-made environment and natural environment the first question is whether the damage should be monetised in the form of environmental external costs, e.g. according to the ExternE approach (EC, 1995). If the decision-makers does not like this type of weighting he has to say if he prefers to monetise the impacts using abatement costs or to express it directly as physical impact parameter; that is in the case of man-made environment for example in maintenance surface for paint [m^2] and yield loss of wheat [t] or for the natural environment the REW in [km^2] as described in chapter 2.

For all the damages on human health the first questions are the same as for the other AoPs; that means whether monetisation and if so by damage or external environmental costs or by abatement cost or internal environmental costs. In the case of fatal effects, furthermore it has to be decided if the monetisation of the damages should be done based on YOLL or directly the VSL. Besides, due the existence of international accepted damage indicators by the WHO in the form of DALY (Murray & Lopez, 1996; Hofstetter, 1998) and YOLL (Meyerhofer et al., 1998) other types of weighting across the different damage endpoints are available. Hence, in the case of damages that cause morbidity the decision-maker has to select among the assessment by DALY or physical impact parameters as for instance emergency room visit, asthma attacks, restrictive activity days, etc. For cancer a selection has to be made among DALY, YOLL and the physical impact parameter cases of cancer. In the case of morbidity the choice is among DALY, YOLL and the physical impact parameter cases of death. Finally, in the case of site-specific assessment the individual risk can be evaluated, as mentioned in chapter 2. These different weighting options are summarised for the decision-maker in a table where only fields have to be filled in. This table named weighting of impacts forms part of Figure 5.7 in which four decision tables are presented that are relevant for weighting and aggregation. Two tables concern the discount rate for monetisation and the cultural theory for DALY, the last one concerns the decisions related to the aggregation of damages.

In the table for the weighting of impacts one entry has to be made for each damage class. The damages classes are the AoPs man-made environment, natural environment and human health as well as all the so-called global indicators like GWP and ODP, which could be related to the sub-AoPs life support functions and resources if resource depletion is considered as an environmental problem.

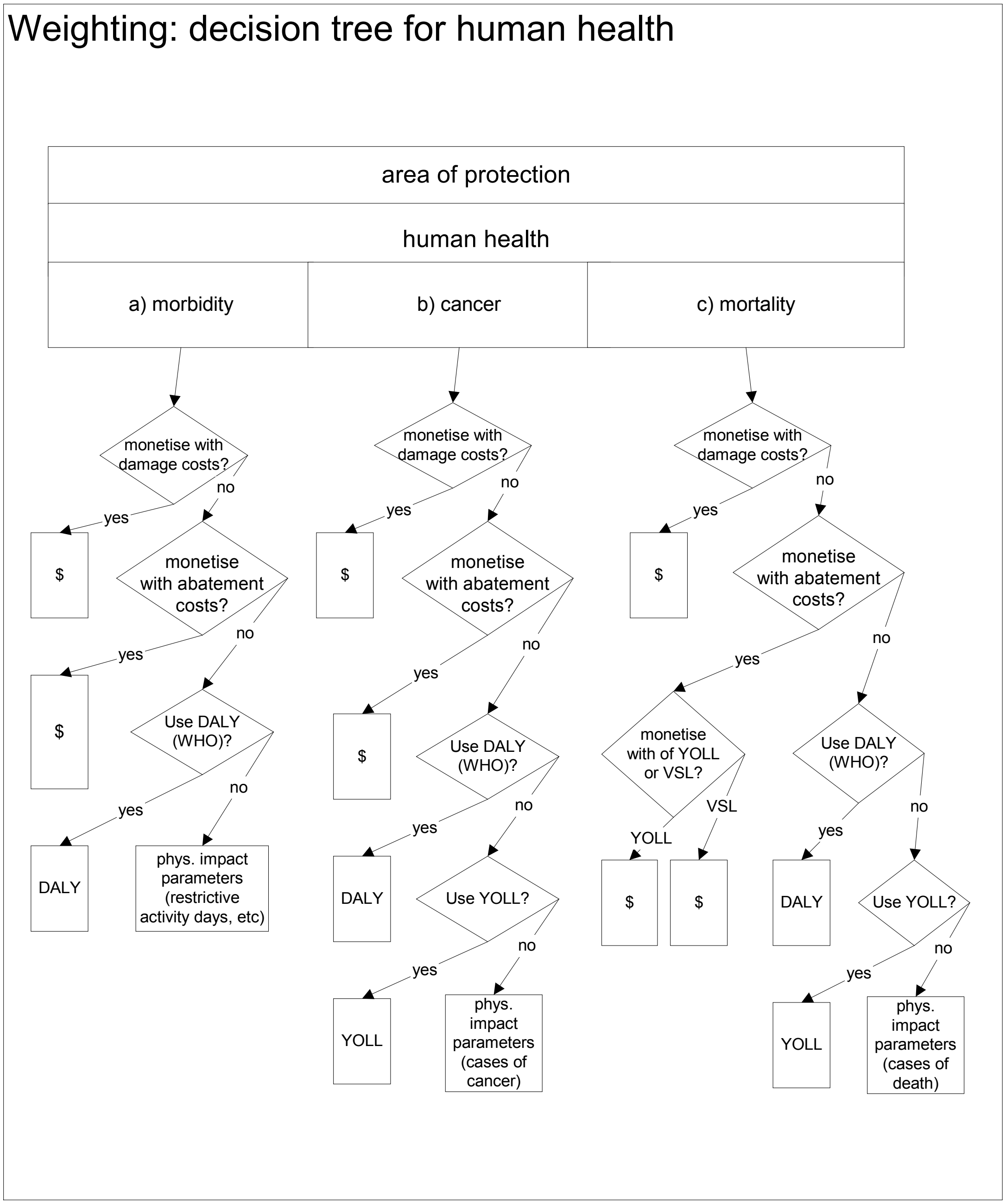


Figure 5.6: Decision tree for weighting for the AoP human health

Weighting and aggregation: decision tables

Weighting of impacts

(for each damage class you need one entry)

damage class (areas of protection, global indicator)		monetisation		Usage of DALY or YOLL	Usage of physical impact parameters
		damages	abatement		
man-made environment		yes			
human health	morbidity	yes			
human health	cancer	yes			
human health	mortality	yes			
natural environment (bidodiversity and natural landscapes)					REW
GWP and other global indicator					GWP

Discount rate for monetisation

select one		
0 %	3 %	10 %
	yes	

Cultural perspective for DALY

select one		
Hierachist	Egalitarian	Individualist

Aggregation decisions of damages

(only possible, if the classes have the same weighting resp. unit)

damage class (areas of protection, global indicator)		<u>intermediate aggregation</u> in the damage-assigning matrix	<u>final aggregation</u> in the damage profil
		create groups	*to show unity*
man-made environment		A	
human health	morbidity	A	
human health	cancer	A	
human health	mortality	A	
natural environment (bidodiversity and natural landscapes)			
GWP, ODP and other global indicators			

Figure 5.7: Decision tables for weighting and aggregation

Human health is divided in morbidity, cancer and mortality. In principle, the monetisation can be done for all the damage classes, either by damages or abatement costs. DALY can only be used for the damages to human health. While for monetisation and DALY no more than the preferred option has to be selected, in the case of using physical impact parameters the selected parameters should be mentioned here.

For the monetisation a discount rate has to be defined. Although in principle any rate can be chosen here 0 %, 3 % and 10 % are proposed according to the standard values used in the ExternE project (EC, 1995). In the case of using DALY one cultural perspective has to be selected. According to Hofstetter (1998) three archetypes are quite well representing human socio-economic perceptions. These are hierarchist, egalitarian and individualist.

Finally, it has to be decided in which way the damage classes are aggregated. Of course this is only possible if the classes have the same weighting unit, e.g. monetary values or DALYs. In principle, there are two options for aggregation. One option is to aggregate directly in the damage matrix, called intermediate aggregation, what is less laborious due to less matrix operations. The other option is to do a final aggregation reducing the number of components in the damage profile, what makes the steps more transparent, but bears the risk to be confusing. In any case, the final result will be the same. Also groups according to certain criteria as for instance AoP can be created to show its unity.

In the Figure 5.7 default selections are presented as illustration of a typical case for a study of environmental damage estimations in industrial process chains. While it is generally accepted that damages to the man-made environment can be best evaluated by external environmental costs, here the decision has been made that this can also be done for the damages to the AoP human health. The natural environment will be assessed by REW and as global indicator only the GWP is chosen. The default discount rate is 3 %. An intermediate aggregation is selected for damages to man-made environment and those to human-health.

Apart from the described selection of clear weighting schemes in order to obtain meaningful indicators is has to be acknowledged that, in principle, also the determination of which dose-response and/ or exposure-response functions to use, and even which dispersion model to apply, implies indirect value choices that especially in the case of the dose-response and exposure-response functions can have very important influences on the final result. So it is recommended also on this point to be transparent and to check what are the preferences of the decision-maker; for instance it can be said that, in general, internationally accepted standard values have a high level of reliability.

All the presented criteria will be exemplified later on in this chapter through the application in a case study that has as one of its primary interest to demonstrate the functionality of the weighting and aggregation scheme.

5.3.3 Life Cycle Inventory analysis

After the goal and scope definition follows the Life Cycle Inventory analysis in the same way as in a LCA according to ISO 14040. An overview of the LCI analysis with its options is given in Figure 5.8.

If a situation of an existing process chain is assessed the measured data (environmental loads) from the core process(es) and those obtained from up- and down-stream processes, e.g. by questionnaires, can be used to feed the LCI spreadsheet model or software tool that contains a more or less elaborated database with information for the background processes.

If a scenario of a process chain options for the future is to assess the data can be generated by a model of the core process(es) and a linear adaptations of current data obtained from up- and down-stream processes. The model can be a modular model as described in chapter 3 or a process simulator.

In the case that in the goal and scope definition the consideration of accidents was chosen, potential environmental loads through the accidents have to be generated by simulations with the corresponding analysis of the undesired events or accidents (AICHE, 1985; Aelion et al., 1995), as mentioned in chapter 2.

The proper LCI can be created either by a spreadsheet model, e.g. Castells et al. (1995), or by a commercial software tool, e.g. TEAM, as presented in chapter 2 and applied in chapter 3. Important in the LCI model or software is the incorporated database especially for background processes like electricity, e.g. Frischknecht et al. (1996).

Another optional element is the uncertainty analysis. This analysis can for instance be carried out by Monte Carlo simulation, as described in chapter 4. By the use of probability distributions for the essential factors in a Monte Carlo Simulation (La Grega, 1994), the inventory results can be transformed from a concrete value into a probability distribution around a mean value.

In principle, the proposed methodology does not need a complete LCI analysis according to ISO 14041 as basis. What is needed are life cycle inventory data about the process chain under study. This data can also be obtained by streamlining LCAs or simplified LCI approaches (Curran & Young, 1996). An important part of such methods is an iterative screening procedure (Fleischer & Schmidt, 1997), which includes similar elements as those used in the methodology presented in this study by the way of an iterative dominance analysis in order to identify the priorities. Evidently, a dominance analysis can also be applied directly in the LCI analysis when collecting data.

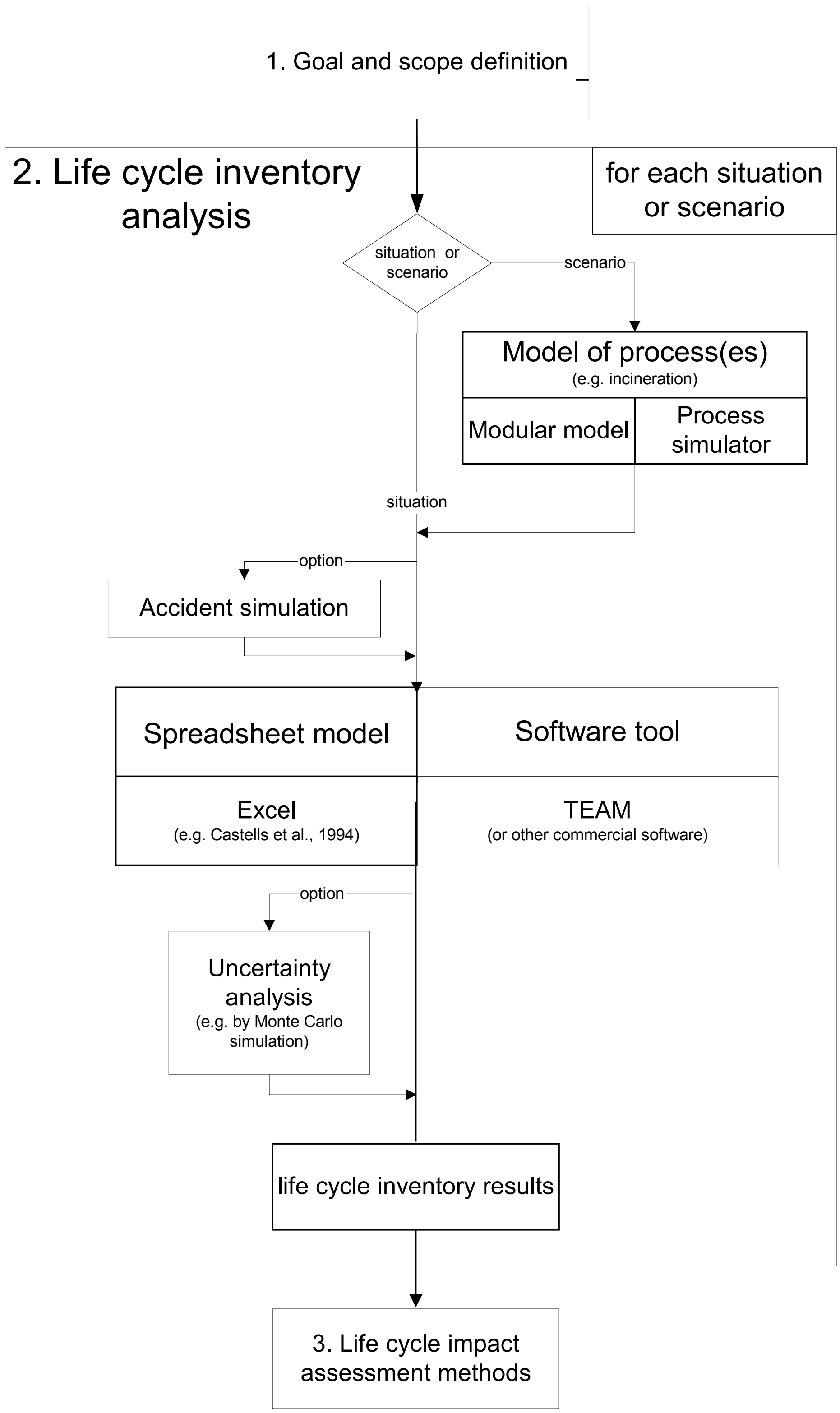


Figure 5.8: Life Cycle Inventory analysis

5.3.4 LIFE CYCLE IMPACT ASSESSMENT METHODS

In the next step following the ISO 14040 framework for LCA, one or more Life Cycle Impact Assessment methods are applied to the LCI results. In the goal and scope definitions the LCIA method(s) have been selected. In Figure 5.9 an overview of the usage of Life Cycle Impact Assessment methods is given, Schematically the main options are shown:

- Midpoint potentials (e.g. GWP and HTP)
- Midpoint based weighting methods (e.g. Eco-indicator 95 and EDIP)
- Direct weighting methods (e.g. Tellus and EcoScarity)
- Endpoint orientated methods (e.g. Eco-indicator 99 and EPS)

More details about these methods can be found in the chapters 2 and 3.

The global indicators that have been selected in the weighting scheme are considered separately. They are obtained in the characterisation step in both options where midpoints are calculated. Each global indicator feeds directly in the damage profile. If required they are first monetised by abatement costs.

The midpoint related LCIA methods allow to calculate the environmental potential of the respective impact category in the characterisation step. All presented LCIA methods except the midpoint potentials permit to obtain a single index to measure the environmental impact performance. The midpoint based weighting methods require carrying out normalisation and then a weighting step. The direct weighting methods omit the characterisation and the normalisation step. Eco-indicator 99 as endpoint-orientated method consists of models in the ecosphere and in the valuesphere (see chapter 2 for further details), but does not contain explicitly midpoint results.

The results of the LCIA methods are called impact scores in Figure 5.9. These scores allow to compare the situations or scenarios on a midpoint level or endpoint-orientated level, but not in the most accurate way that is still feasible with regard to actual impacts and the consideration of spatial differentiation. Therefore one or more selected impact scores are used in a dominance analysis in order to estimate more in detail the environmental damages of the main processes and pollutants of the studied process chain.

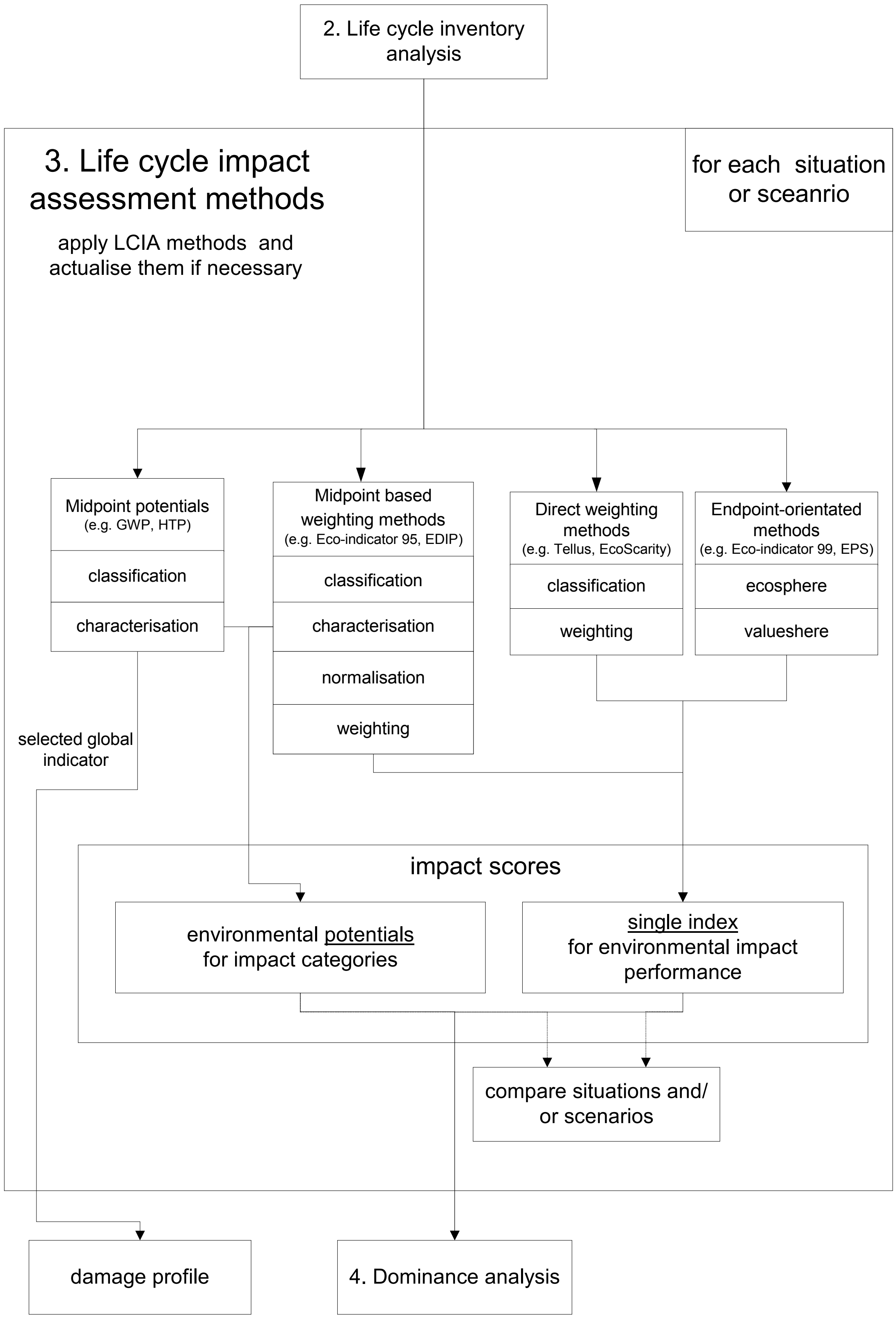


Figure 5.9: Usage of life cycle impact assessment methods

5.3.5 DOMINANCE ANALYSIS AND SPATIAL DIFFERENCIATION

Before determining the predominant processes and pollutants of the studied process chain for each selected impact score, it has also to be checked which media are mostly affected by the emissions of the process chain. Afterwards, the environmental loads and process have to be spatially differentiated. Therefore, in Figure 5.10 together with a general overview of this selection procedure the dominance analysis for media and the spatial differentiation is presented.

In the dominance analysis for media (i.e. air, water and soil), first all emissions considered in the selected impact score are assigned to its medium. Then the contribution of each medium to the total impact score is presented graphically. Finally a decision has to be made which media, emitted to, will be considered for further assessment. In this decision also qualitative arguments can be used.

The dominance analysis for processes and pollutants is applied to the predominant media and is structured in a similar way for processes and emissions (Figure 5.11). First for each process and pollutant the percentage of the total of the selected impact score is calculated, then the defined initial cut-off criteria are applied. For processes and pollutant respectively the percentages and cut-off criteria are represented graphically to visualise the data. The graphical presentations may suggest redefining the respective cut-off criteria in order to get a manageable number of processes that can be assessed further in detail. In principle, this is an iterative procedure to find the optimum. It is up to the decision-maker to decide if one or more impact scores are to be considered and in which relation to another. That means also that the dominance could be carried out with different cut-off criteria for different impact scores, for instance 5 % for the human toxicity potential and 10 % for the acidification potential.

In the case of the selection of processes, it has also to be determined which processes should be assessed by site-specific and site-dependent factors (in general obtained by project-related impact assessment studies) and which in a process-specific and/ or region-dependent way (in general by values published in the literature).

In a next step, the most relevant pollutants and industrial processes (differentiated in site-specific and site-dependent as well as in process-specific and/ or region-dependent) are determined. Also in this decision qualitative arguments can be used. In a sustainable perspective also social and economic aspects are important. Hence, it seems evident that for instance PCDD/Fs have to be considered in a case study on waste incineration due to their relevance in discussions in society although their percentage of the total impact score is less than the lowest selected cut-off criterion.

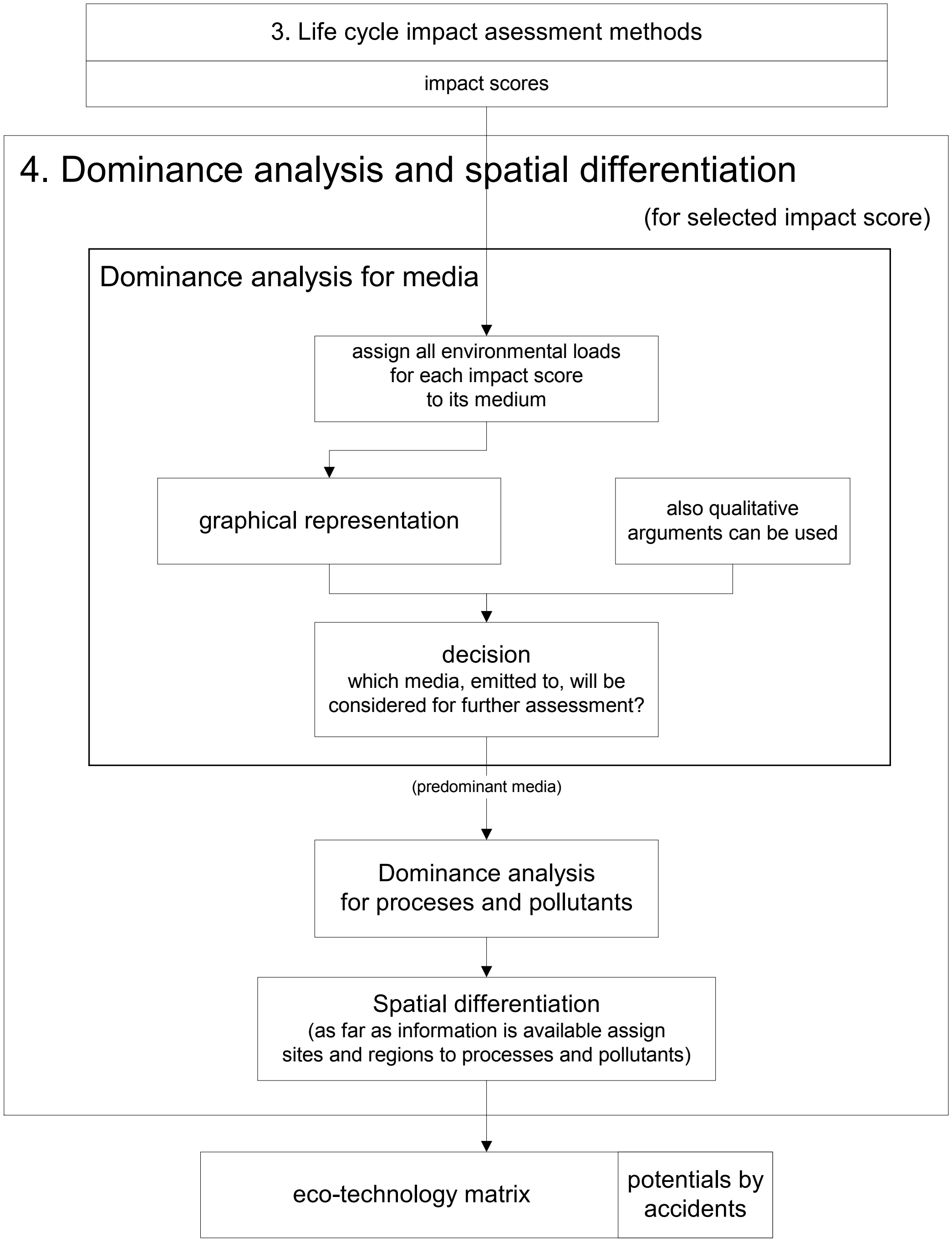


Figure 5.10: Dominance analysis for media and spatial differentiation

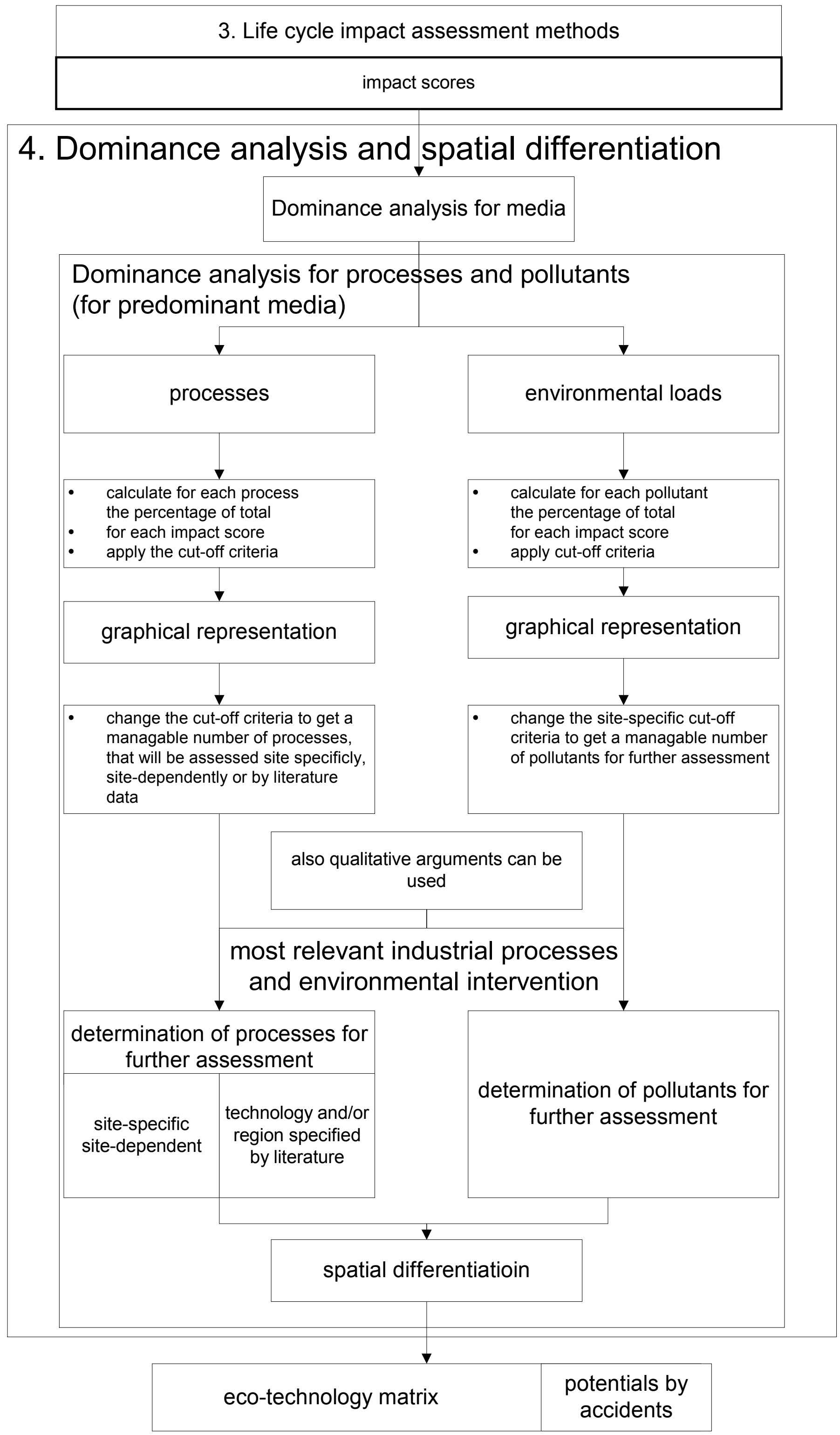


Figure 5.11: Dominance analysis for processes and pollutants

Finally the identified predominant pollutants and regions have to be assigned to sites or regions. In how far this is possible depends especially on the information that is available about the location of a particular process. Hence, it has to be taken into account that the site might be unknown; in this case only the region can be assigned. The spatial scale of the pollutants depends on their residence time in the respective medium. A lot of background processes whose LCI data are normally taken from databases are broadly spatially distributed. Here the question is to determine the most adequate size for a region. For example in the case of electricity production, in general the LCI data for the electricity mix of a country is taken.

A problem that occurs in the assignation of sites is that, actually, often it is not the site itself that most influences the environmental damages, but the emission height. This can be concluded by the results for the site-dependent impact factors obtained in chapter 6. Therefore, instead of regions essentially we make a differentiation according to classes that have similar characteristics with regard to the emission situation in the way as outlined in chapter 6. However, in this methodology we call them regions due to the fact that this term illustrates much better the idea behind spatial differentiation.

Consistent with expression (5.7) the world is the corresponding region for pollutants that causes a global environmental impact like CO_2, and in agreement with expression (5.8) in the case of mobile processes, i.e. transports, it has to be decided if the environmental loads can really be assigned to only one region or if they have to be differentiated among two or more regions if the distance is sufficient long.

The determined processes assigned to sites and regions are the M processes and the chosen pollutants are the N pollutants of the eco-technology matrix. The eco-matrix can contain always a part of potential environmental loads if accidents have been considered.

5.3.6 FATE & EXPOSURE ASSESSMENT AND CONSEQUENCE ANALYSIS

The level of detail in the fate & exposure assessment depends on the determined importance of the respective process. The few processes that contribute most to the overall environmental impact should be assessed in a site-specific if possible; other important processes can be evaluated by corresponding region-dependent or technology-dependent impact assessment factors published in the literature, e.g. Krewitt et al. (2001). For airborne pollutants due to transport processes an evaluation by site-dependent impact assessment by statistically determined factors for generic classes seems to be most adequate. Nigge (2000) has proposed such a method, in this study it was further developed and applied to the case study. It is explained in chapter 6. However, it still has to be considered as an approach in development.

Figure 5.12 gives an overview of the fate & exposure and consequence analysis with the different levels of detail. The results of the fate & exposure and consequence analysis are the basis for the damage-assigning matrices.

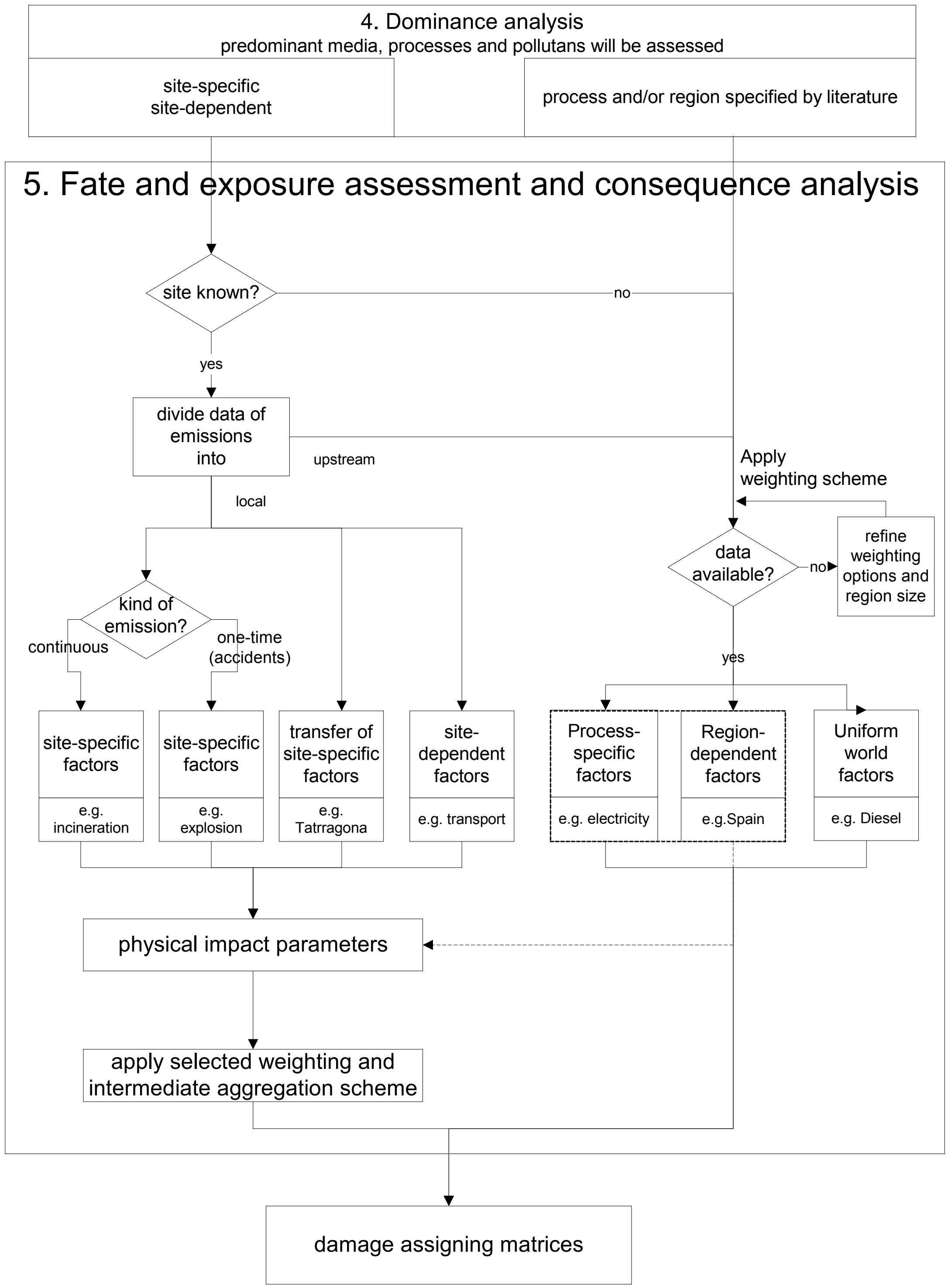


Figure 5.12: Fate & exposure assessment and consequence analysis

For the processes that have been identified to be most important a site-specific or site-dependent assessment is carried out if the site is known. If this is not the case the corresponding process has to be treated as those processes that have been determined to be evaluated process- and/ or region specified by literature values. In the case the site is known the data about the emissions in the LCI have to be divided into the upstream-related data that have to be evaluated by literature values and the foreground process-related or local emissions.

Only the obtained local emission data can be further assessed. If potential emissions due to accidents are taken into account in the LCI and the eco-technology matrix it has to be checked in each case, which kind of emissions it is (continuous or one-time) for a site-specific assessment. An example of a site-specific assessment of continuous emissions is IPA (and the ERA) carried out for the MSWI in Tarragona as described in chapter 3. An example of one-time emissions is that from an explosion. If once a site-specific impact assessment has been carried out in a region and in this way site-specific factors have been estimated the results can be transferred to another process in the same region, using a transfer factor if necessary for the stack height as proposed by Rabl et al. (1998). Such a transfer is for instance the use of results of the site-specific impact assessment of the MSWI of Tarragona for another process in the region of Tarragona, e.g. for the ash treatment situated in Constanti a few kilometres away from the municipality of Tarragona.

If site-dependent impact assessment factors according to the approach outlined in chapter 6 are available then these factors allow to estimate the environmental damages due to airborne emissions in the way of an adequate trade-off between accuracy and practicability. This holds true especially for transport [tkm] due the fact that transport can be considered as a number of industrial processes that take place (at one time after the other) in different regions.

Through the site-specific and site-dependent impact assessments physical impact parameters are obtained that can be converted in indicators or environmental costs by the application of the selected weighting scheme. Moreover, if laid out in the goal and scope definition, the intermediate aggregation can take place.

For the processes where literature values should be used the first action is to check if the wished data are available in the literature. It this is not the case the weighting options of the scope definition and/ or the size of the region have to be refined. Due to the fact that the assessment of site- and region-dependent damage endpoints is an issue that more or less started in the mid-nineties there are not so much data published. Hence, it is quite possible that for certain regions processes and pollutants determined indicators or cost types are not available in the literature.

If data are available then it has to be seen for each process if a classification is possible according to technology and/ or region. If one or both options are possible then technology and/ or region-dependent factors from the literature are used to estimate the corresponding damage. For instance data on external costs are available for electricity production [kWh] in the region of Spain. Another example are the mentioned region-dependent impact factors, e.g. in YOLL, published by Krewitt et al. (2001) for different European countries and some world regions. If a classification according to technology and/ or region is not possible then uniform world factors have to be applied. For instance, Rabl et al. (1998) have published a uniform world model for air emissions. An example of a process that is difficult to classify is the diesel production and the related process chain that takes place all over the world.

Depending on the selected weighting scheme and the available data in the literature physical impact parameter, damage indicators or environmental costs are obtained. The physical impact parameters can be summed up directly with those obtained in the site-specific and site-dependent assessment. Damage indicators and environmental costs can be gathered together according to the selected intermediate aggregation scheme.

Options for site-specific impact assessment are explained in Figure 5.13 for the medium air. In principle it can also be applied to other media. For example Schulze (2001) presents site-orientated impact assessments for the medium water in relation to LCAs for detergents, using the integrated assessment model GREA-TER in an adapted version that is valid for products instead of chemical substances.

Based on the data of local air emissions site-specific factors are calculated for the predominant pollutants. These factors can first be expressed in the form of physical impact parameters before being weighted and aggregated according to the scheme chosen in the goal and scope definition.

The fate & exposure analysis can be carried out in a generic or detailed way. The generic way means to use an integrated impact assessment model, e.g. EcoSense (described in chapter 2). Such an integrated impact assessment model consists of a Gaussian dispersion model for the pollutant transport near the emission point (i.e. approximately <= 100 km) and another transport model for the long-range pollutant transport (i.e. approximately > 100 km). In the case of EcoSense 2.0 the models included are ISCST-2 and WTM. The integrated impact assessment model EcoSense includes also an elevated number of dose-response and exposure-response functions that can be used for the consequence analysis. The level of detail in the database of an integrated impact assessment model is of course limited, e.g. the resolution of population densities is not as detailed as it could be when using a geographic information system as demonstrated in chapter 3.

In the case of a more detailed assessment, only the long-range transport model of the integrated impact assessment model is used (e.g. WTM in EcoSense). For the transport near the emission point an independent Gaussian dispersion model is applied (e.g. ISCST-3 in BEEST) and more detailed geographic data like in ERA provided by a geographic information system are employed (e.g. population densities taken from the GIS MiraMon). In Figure 5.13 it is proposed to carry out the consequence analysis for the damages due to long-range transport within the integrated impact assessment model and to do it separately, e.g. on a spreadsheet, for the damages generated by the emissions near to the emission point. Then the physical impact parameter of both assessments can be summed up. This is a practical proposal, in principle the consequence analysis can also be carried out together.

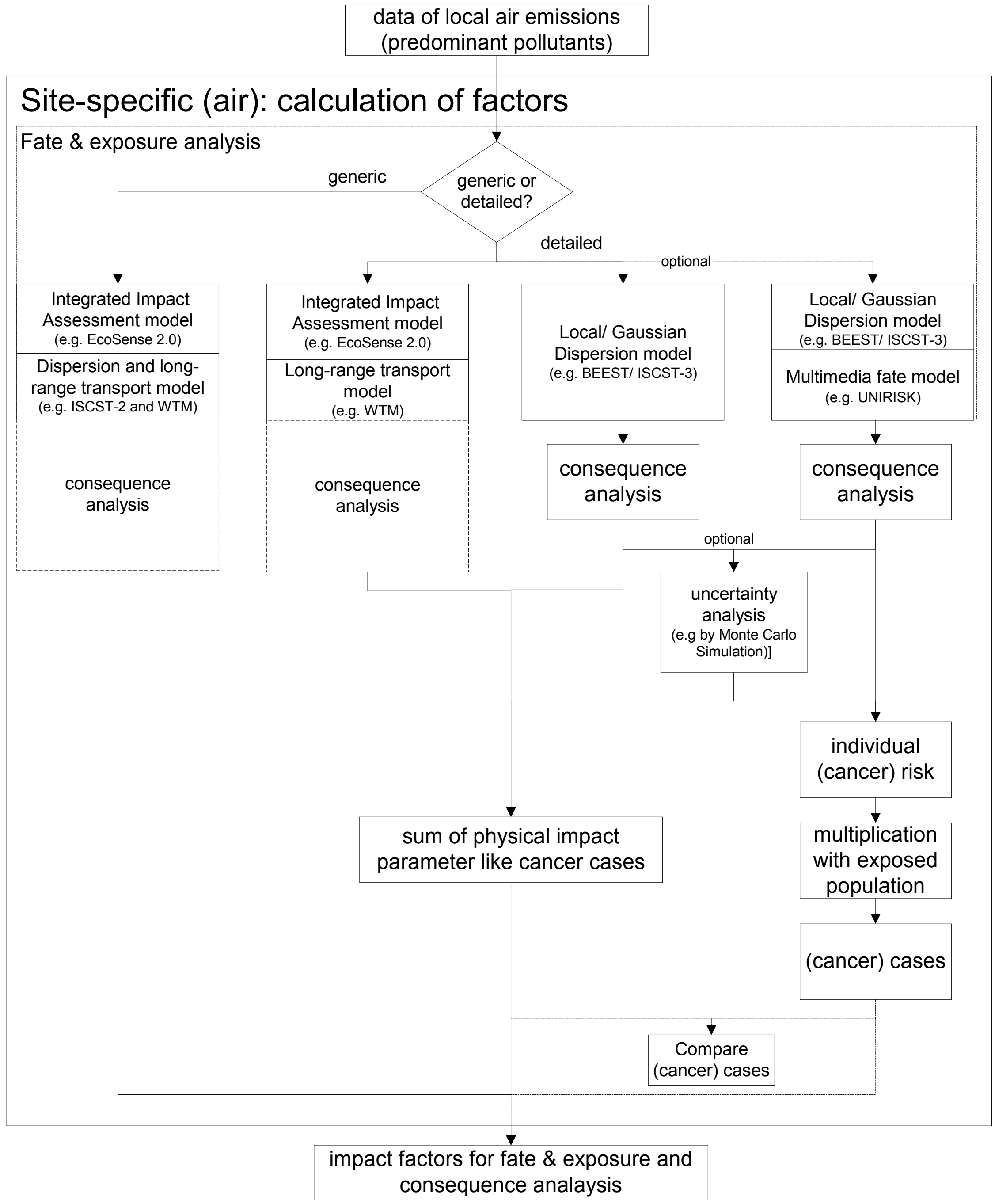


Figure 5.13: Site-specific impact assessment for air emissions

When calculating the site-specific impact factors a lot of work is done to complete at the same time an ERA, as described in chapter 3. The only additional step is the application of the multi-media fate model (e.g. UNIRISK). Or we can say it also the other way round: If an ERA has been carried out in the way as done in chapter 3 for one of the identified predominant processes of an industrial process chain, then it is quite easy to compute the impact factors necessary for a quite accurate environmental damage estimation of industrial process chains.

The realisation of an ERA is seen in the Figure 5.13 as an optional element because of the laborious work of multi-media modelling and the limited importance of the other pathways, based on the results obtained in the ERA applied to the MSWI in Tarragona (chapter 3). A correction factor could be used to take into account the other pathways as proposed by Mayerhofer et al. (1998).

Also in the case of the environmental risk assessment a consequence analysis has to be carried out. Due to the high level of detail in this type of study also thresholds can be considered, so that in the case of human health assessment the consequence analysis is not restricted to cancerigenic effects and respiratory diseases (see chapter 2 for further explanations) but also other types of toxic effects can be considered.

When carrying out ERA in the way as done in chapter 3 the individual risk, e.g. to get cancer due to the increment of a certain pollutant in the atmosphere is calculated. The multiplication with the absolute number of population exposed allows to obtain an estimate of the damage in the form of physical impact parameters as for instance cancer cases. These calculated impact parameters like cancer cases can also be compared with those provided by the application of the IPA on a local scale. Such a comparison can provide the above-mentioned correction factors.

Another optional element is the uncertainty analysis that can be carried out e.g. by Monte Carlo simulation according to the framework proposed in chapter 4. In the same way as for the LCI results the outcomes of the fate & exposure analysis can be transformed from a concrete value into a probability distribution around a mean value.

Apart from site-specific impact assessments in this work the focus for the fate & exposure and consequence analysis has been on site-dependent impact assessment as an adequate trade-off between accuracy and practicability. The whole method is largely explained in chapter 6. The other options of the fate & exposure and consequence analysis (Figure 5.12) do not need further explications since they are similar to the site-specific impact assessment or consist only in the application of published values for impact indicators.

5.3.7 DAMAGE PROFILE

In Figure 5.14 the last part of the obligatory steps for the methodology of environmental damage estimations for industrial process chains is presented. In principle, this flowchart consists of an illustration of the developed mathematical framework.

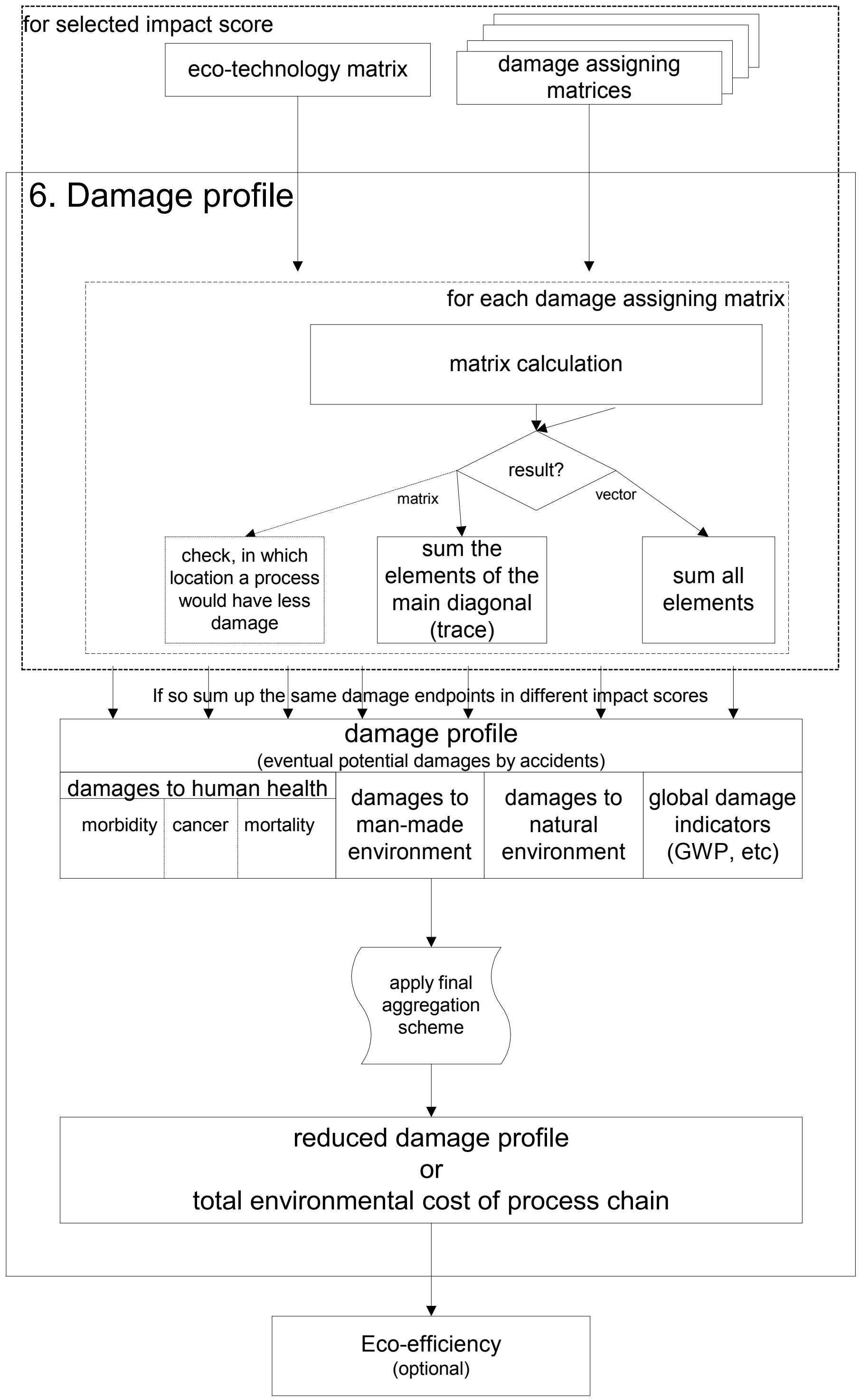


Figure 5.14: Damage profile

For each selected impact score the eco-technology matrix is multiplied with the damage-assigning matrices. For each damage-assigning matrix the result can be another matrix or a vector. It will be a vector for the case of global impacts. In that case the elements of the vector just have to be summed up. In the case of the matrix a sum has to be made of the elements of the main diagonal, the trace. The matrix allows to check, in which location a process would have caused less damage.

The sum obtained by each matrix calculation provides a damage endpoint per impact score that then forms the damage profile. If the same damage endpoints have been estimated in different impact scores they have to be summed up. The damage profile might contain potential damages in the case accidents have been simulated. In principle, depending on the selected intermediate aggregation scheme, the damage profile can be broken up into damages to the AoPs human health (morbidity, cancer, mortality), man-made environment and natural environment as well as the so-called global damage indicators, which could be related to the sub-AoPs life support functions and resources if resource depletion is considered as an environmental problem.

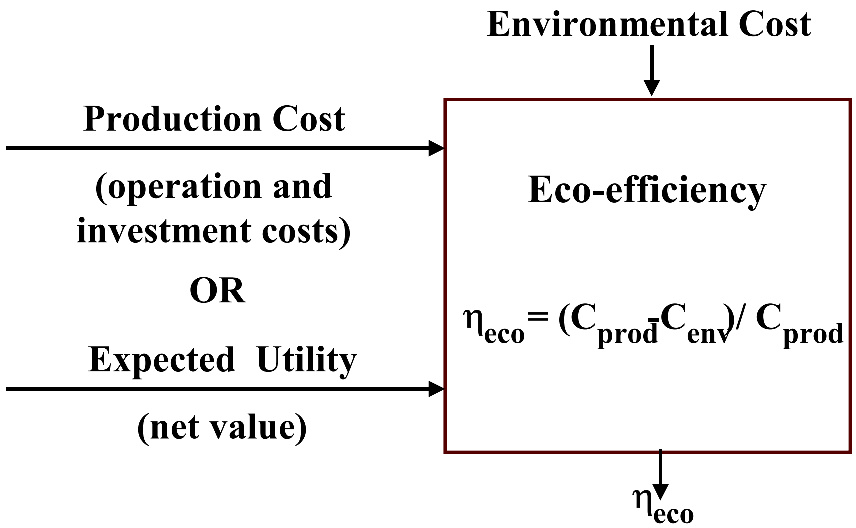
If the damage profile is consisting of different damage endpoints their number can be further reduced applying the selected aggregation scheme. In this way finally a reduced damage profile is obtained or an estimation of the total environmental cost of the process chain under study.

5.3.8 ECO-EFFICIENCY

A further optional element is the calculation of the eco-efficiency of the industrial process chain for which the environmental damages have been estimated. The concept of eco-efficiency has been proposed as expression of sustainability for economic activities (see chapter 2). Eco-efficiency has been defined as the delivery of competitively priced goods and services while progressively reducing environmental impacts.

Hence, it has been suggested to measure eco-efficiency η_{eco} by the coefficient of the difference between production costs C_{prod} and external environmental costs C_{env} to the production costs C_{prod}. While the production costs are easily to obtain, the environmental costs are not so visible. Nevertheless, they can be estimated by the presented methodology of environmental damage estimations for industrial process chains.

The expression (5.9) is applicable to the final result of the environmental damage estimations for industrial process chains if this is expressed in a monetary unit. Figure 5.15 illustrates the procedure to calculate the eco-efficiency according to this expression. Instead of the production costs (operation and investment costs) also the expected utility or net value could be used for the determination of the eco-efficiency.



(5.9)

Figure 5.15: Eco-efficiency of an industrial process chain

5.4 ANALYTICAL TOOLS AND TECHNICAL ELEMENTS USED IN THE PRESENTED METHODOLOGY

The methodology permits various linkages with other environmental management tools and concepts. For instance, the impact parameters for local impacts permit to carry out risk assessments for single processes; and in combination with an analysis of the internal costs, the eco-efficiency of the process chain can be determined.

Moreover, the whole methodology is a combination of different analytical tools that in general have been developed for other applications. First of all, the LCA methodology is to mention that in principle has been developed for the environmental assessment of product systems. LCA is an important element for the steps goal and scope definition, LCI analysis, LCIA methods and for providing site-, region- technology dependent impact factors The next tool to look at is the Impact Pathway Analysis (IPA) that it the fruit of a project to assess the externalities of electricity production. IPA is crucial for the step fate & exposure and consequence analysis, including the weighting and aggregation schemes. Furthermore, Environmental Risk Assessment (ERA) is to cite that has got its origin in the assessment of the behaviour of chemical substances in the environment. It is of course relevant in the fate & exposure and consequence analysis. It has not only influenced IPA, but also the LCIA methods. Other methods that are indirectly involved are Cost-Benefit Analysis (CBA), accident investigation and process simulation.

Finally, several technical elements are behind the methodology and its flowchart. Here only the main technical elements are outlined. The terminology is based on Dale & English (1999). Due to the LCA part a functional unit has to be defined and in the LCI analysis allocation models have to be used. In the fate & exposure analysis fate and transport models (Gaussian, long-range transport, multimedia) are applied. In the consequence analysis dose-response and exposure-response functions are employed. Laboratory exposure and animal test are often the basis for dose-response functions and epidemiological studies the basis for exposure-response functions. In order to make the weighting transparent decision trees have been established. Value – evaluation models are the basis for the weighting factors used. One of such models is the cultural theory in combination with the DALY concept. Related to this is the socio-economic impact assessment as done with the presented different methods to evaluate external costs (chapter 2) like contingent valuation. ERA uses individual risk or population risk based on the lifetime average dose. Accidents simulation needs the help of event and fault trees. Process simulation is in principle an engineering model. Eco-efficiency could be calculated with the net present value of an expected utility. The uncertainty analysis can be carried out with Monte Carlo simulation.

The whole methodology will be exemplified in the following section through the analysis of a case study that has as primary interest to demonstrate the functionality of the presented framework.

5.5 APPLICATION OF THE METHODOLOGY TO THE WASTE INCINERATION PROCESS CHAIN

The methodology has successfully been applied with interesting results to the case study on the process chain related to waste incineration. The information that can be obtained by the developed methodology might be crucial in the future for decisions on further improvement of exiting and new waste management systems.

The presented algorithm is applied to the life cycle inventory of the electricity produced by the municipal solid waste incinerator of Tarragona/ Spain.

1) Goal and scope definition

Two operating situations of the MSWI are compared: the situation in 1996 and the current situation after the installation of an advanced gas removal system. More data on the MSWI are presented in chapter 3.

The function of the MSWI is to treat the household waste of the surroundings of Tarragona. However, the produced TJ of electricity is chosen as the functional unit, as done in the existing LCA study. The study comprises in its system boundaries all the processes from the municipal waste disposal in containers to the landfill of the final waste (Figure 5.16). As LCIA method the midpoint based weighting method with single index, Eco-indicator 95

(Goedkoop, 1995), is used. The method is based on the characterisation factors presented by Heijungs et al. (1992) and uses equal scores of distances to science-political targets (for more information see chapter 2 and 3).

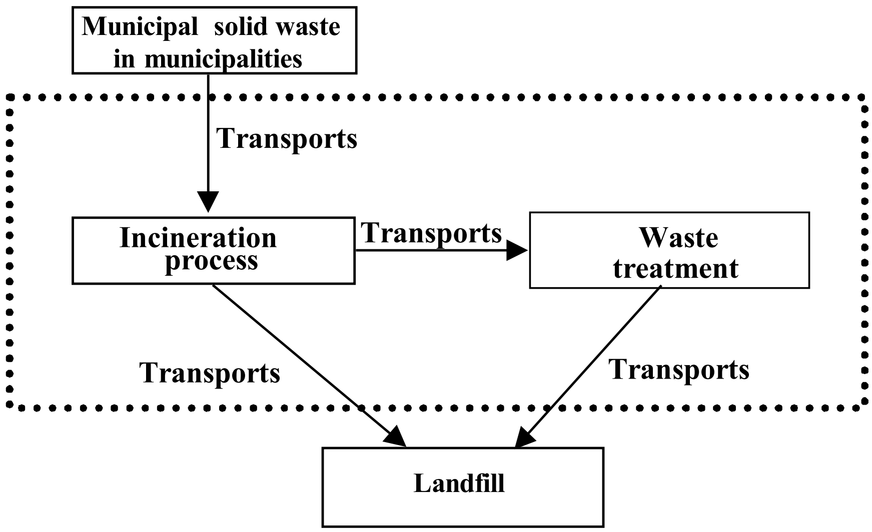


Figure 5.16: Boundaries of the studied system

In the dominance analysis, for reasons of resource economy, only all processes with a contribution greater than 10 % will be selected for site-specific impact assessment by a particular study. However, the remaining processes, with more than 1 %, should be evaluated by the transfer of available site-specific damage data or the use of values published in the literature to estimate its environmental damages.

The selected weighting and aggregation scheme corresponds to the one represented as an example in Figure 5.7. That means that three indicators have been selected for the weighting of impacts. For the AoP, human health and man-made environment, the external environmental costs (EEC) according to the European Commission (1995) have been used.

Considering monetary evaluation of environmental damages, there exists a lot of criticism. For this reason, special attention is paid to the arguments why to monetise these damages. The impossibility to sum up the non-economic impact endpoints necessarily implies a value judgement. As most decisions have to confront the reality of the market place, hence the most useful measure is the cost of the damages. This information allows society to decide how much should be done for the protection of the environment by the public institutions and how much of the damage cost should be internalised so that a functional unit is consistent with the market. Further information on this topic can be found in chapter 2 and the huge externality studies for electricity production carried out in parallel in EU (EC, 1995) and USA (ORNL, 1995).

For the damage evaluation of the AoP natural environment (biodiversity and landscape) there exist no acceptable economic method. Therefore, the evaluation has to be carried out through an ecological damage parameter. In the present study, the parameter that has been applied is the Relative Exceedance Weighted (REW) ecosystem area where the critical load of a pollutant is exceeded (UN-ECE, 1991); this parameter is also further explained in chapter 2.

The global damages that might occur in the future due to the emission of greenhouse gases are highly uncertain to forecast and to monetise. Therefore, the climate change has been expressed in the form of the GWP as in the LCIA (Albritton & Derwent, 1995).

The potential occurrence of accidents is not considered in this case study. Uncertainty analysis for the LCI and the site-specific environmental impact assessment of the MSWI emissions are described in chapter 4. An environmental risk assessment for the same plant has been carried out and the results are presented in chapter 3. Based on the results of the environmental damages estimations of the waste incineration process chain the eco-efficiency will also be calculated.

2) LCI analysis

The Life Cycle Inventory analysis is described in chapter 3. Here the existing results are used for creating the eco-technology matrix of the environmental damage estimation for industrial process chains.

3) LCIA method

The detailed results of the application of the LCIA method Eco-indicator 95 can be found in chapter 3, where also a comparison of the results for the two situations based on the Eco-indicator 95 is done. In the presented methodology the impact score is further applied for the dominance analysis.

4) Dominance analysis and spatial differentiation

In the current case study the predominant medium, to which the emissions are emitted, is clearly air. The predominant pollutants are those that have been selected by dominance analysis for the uncertainty analysis in chapter 4 (see Figure 4.4).

Figure 5.17 presents the contribution of the considered processes in the LCI analysis to the total environmental impact potential measured as Eco-indicator 95. It is evident that in this case study only the incineration process contributes with more than 10 % to the total environmental impact potential. Hence, it is the only process that will be assessed in a site-specific way by a particular study. The corresponding site is Tarragona.

The other industrial processes with more than 1 % contribution will be considered in two ways: On one hand the data obtained from the IPA study of the incineration process in Tarragona are considered to be valid for all processes in the Tarragona region. On the other hand, the remaining processes have to be evaluated using damage information of similar situations obtained from the literature. The processes that contribute with more than 1% and less than 10 % to the total environmental impact are spatially differentiated in the following way: The processes "production of CaO", and "treatment of ashes" take place in the

Tarragona region In the LCI analysis data for the "electricity generation" are used from the so-called Spanish mix. The environmental impacts of the "transport" and "scrap-metal recycling" are also depending on the Spanish region.

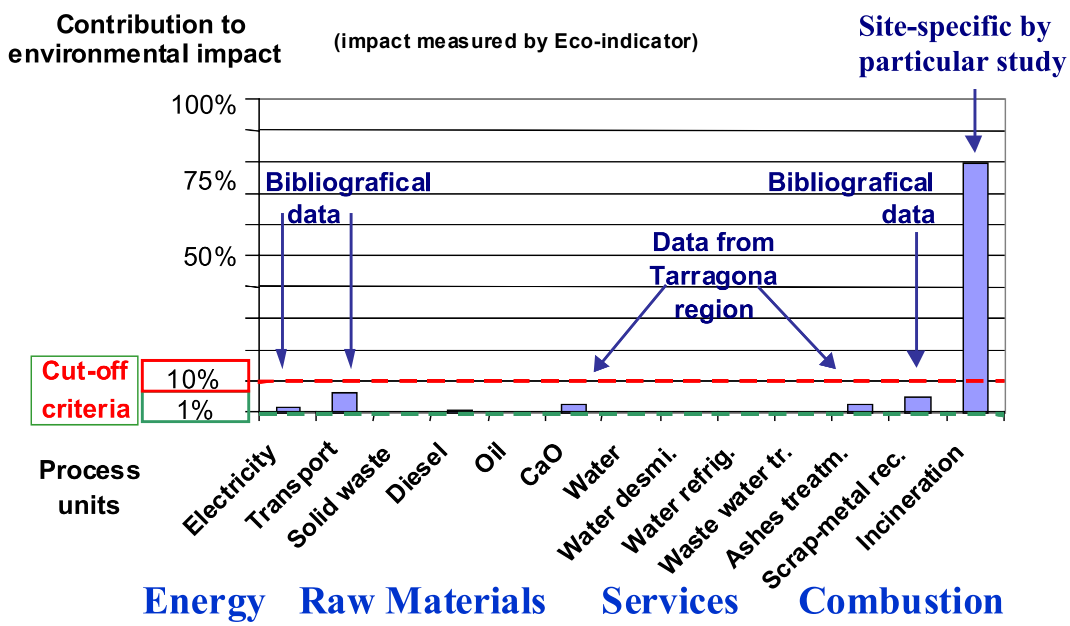


Figure 5.17: Dominance analysis for the relevant processes for site-specific damage assessment

The results of the inventory analysis of the current situation for the relevant processes and the selected environmental loads are presented in Table 5.1. This table includes the eco-technology matrix with sub-matrixes: the environmental loads from kWh to NOx correspond to the first matrix for the economic damage parameter, the second matrix for the ecological damage parameter consists only of SO_2 and NOx, and the third matrix for the global damage parameter includes the other loads from CO_2 to trichloromethane.

Table 5.1: Eco-matrix with life cycle inventory data for selected environmental loads, divided in those assessable by environmental costs and those responsible for ecosystems damages and global warming, for the current situation 2

Relevant Process	**Electricity**	**Transport**	**CaO**	**Ash treatment**	**Scrap-metal recycling**	**Incineration**
Region	Spain	Tarragona/ Catalonia	Tarragona/ Catalonia	Tarragona	Madrid/ Spain	Tarragona
kWh	2,96E+03	0,00E+00	0,00E+00	0,00E+00	0,00E+00	0,00E+00
tkm	0,00E+00	2,64E+04	0,00E+00	0,00E+00	0,00E+00	0,00E+00
kg PM10	2,08E+00	8,45E+00	1,11E+02	4,73E+00	6,03E+00	2,82E+01
kg As	3,12E-04	2,93E-03	3,74E-04	8,66E-04	2,09E-03	3,29E-02
kg Cd	6,97E-04	4,28E-04	4,74E-05	1,41E-04	3,06E-04	3,87E-02
Kg Ni	1,29E-03	5,56E-03	1,31E-03	2,42E-03	3,98E-03	4,93E-02
kg VOC	4,74E-01	1,68E+01	5,84E-01	4,86E+00	1,21E+01	0,00E+00
ng Dioxins TEQ	8,05E+01	7,32E+02	3,83E+02	7,57E+02	5,22E+02	1,17E+04
kg CO	3,36E-01	2,92E+01	2,46E+00	1,16E+01	2,09E+01	2,35E+02
kg SO_2	1,86E+01	1,93E+01	1,16E+01	1,17E+01	1,38E+01	1,77E+02
kg NO_2	3,30E+00	9,31E+01	5,76E+00	3,17E+01	6,65E+01	1,12E+03
kg CO_2	1,48E+03	9,13E+03	8,02E+03	5,76E+03	6,53E+03	2,25E+05
kg dichlorometh.	9,90E-07	3,17E-06	5,64E-07	1,11E-06	2,26E-06	0,00E+00
kg HALON-1301	2,27E-05	1,09E-03	3,61E-05	3,21E-04	7,80E-04	0,00E+00
kg meth.	4,51E+00	1,41E+01	7,57E+00	8,19E+00	1,01E+01	0,00E+00
kg N_2O	1,27E-02	1,00E+00	3,55E-02	2,75E-01	7,16E-01	0,00E+00
kg tetrachlorometh.	3,03E-07	1,03E-05	6,41E-06	3,78E-06	7,36E-06	0,00E+00
kg trichlorometh.	1,55E-08	1,10E-06	7,06E-07	4,03E-07	7,83E-07	0,00E+00

5) Fate & exposure and consequence analysis

In a next step, the corresponding three damage-assigning matrixes for the selected three indicators are established. It is tried to give a particular value to each environmental load for a specific region or site because the respective indicator depends for each environmental load on the characteristics of the respective region or site. If there are no region-dependent or process-specific damage estimates available for a pollutant, a general model, the Uniform World Model (Rabl et al., 1998), is used in the case of environmental external costs.

The largest damage-assigning matrix is elaborated for the environmental damage cost; the matrix for the current situation is presented in Table 5.2. For the electricity process the corresponding region is Spain, the evaluation is done for the environmental load kWh (technology-.dependent) through the project data published by CIEMAT (1997). The transport is assessed by literature sources for lorry metric ton km [tkm] in South Europe (Friedrich et al., 1998), since data for Spain were not available. The processes CaO supply, ash treatment and incineration are located in the Tarragona region. The weighting factors of the Tarragona region for all environmental loads accept VOC are obtained by the IPA for the incineration process described in chapter 3. The VOC values as well as the evaluation of the scrap-metal recycling are taken from the Uniform World Model (Rabl et al., 1998), a model developed to provide medium factors obtained out of a huge amount of externality studies for use in economic damage assessment if no regional values are available.

Table 5.2: Damage-assigning matrix for External Environmental Cost (EEC) in €

Relevant Process	Region and Source	kWh	tkm	kg PM10	kg As	kg Cd	kg Ni	kg VOC	PCDD/Fs ng TEQ	kg CO	kg SO_2	kg NO_2
Electricity	Spain (1)	0,040	0,00	0	0,0	0	0,0	0,00	0,00	0,00	0	0
Transport	South Europe (2)	0,00	0,31	0	0,0	0	0,0	0,00	0,00	0,00	0	0
CaO	Tarragona, site-specific transfer	0,00	0,00	23	3,0	27	61	0,73	2,6E-08	1,03	13	11
Ash treatment	Tarragona, site-specific transfer	0,00	0,00	23	3,0	27	61	0,73	2,6E-08	1,03	13	11
Scrap-metal recycling	Uniform World Model (3)	0,00	0,00	14	156	19	2,6	0,73	1,7E-05	0,0022	13	19
Incineration	Tarragona, site-specific	0,00	0,00	23	3,0	27	61	0,73	2,6E-08	1,03	13	11

(1) CIEMAT, 1997; (2) Friedrich et al., 1998; (3) Rabl et al., 1998

Due to the lack of literature data, the matrix of the ecological damage indicator REW is not yet fully established. Information is only available for the processes in the Tarragona region. Nevertheless, the matrix for the current situation is presented in Table 5.3 to illustrate the complete framework.

Table 5.3: Damage-assigning matrix for Relative Exceedance Weighted (REW) area in km^2

Relevant Process	Region and Source	kg NO_2	kg SO_2
Electricity	-		
Transport	-		
CaO	Tarragona, site-specific factor	1,31E-05	1,36E-06
Ash treatment	-		
Scrap-metal rec.	Tarragona, site-specific factor	1,31E-05	1,36E-06
Incineration	Tarragona, site-specific	1,31E-05	1,36E-06

In the case of the global damage indicator the situation is slightly different. As the GWP changes from one environmental load to another, but the impact is site-independent, indeed, in Table 5.4 there is no weighting matrix but a vector (for the current situation).

Table 5.4: Damage-assigning matrix for Global Warming Potential (GWP) in kg CO_2-equivalents

Relevant Process	Region and Source	kg CO_2	kg dichlorometh.	kg HALON-1301	kg meth.	kg N_2O	kg tetrachlorometh.	kg trichlorometh.
All processes	World (1)	1	15	4900	11	270	1300	25

(1) Albritton & Derwent, 1995

6) Damage profile

The multiplication of the damage-assigning matrixes with the eco-technology matrix (the respective parts of the inventory table) yields the damage profile (see Table 5.5 for the current situation), i.e. the ensemble of the three damage parameters, per functional unit. In the weighted eco-technology matrix or damage matrix for the current situation, the external environmental cost per functional unit is estimated as 28,200 €/ TJ electricity what is in the range of other externality studies for waste incineration (Rabl et al., 1998; CIEMAT, 1997). That is the sum of the diagonal corresponding to the sum of the regions considered in the life cycle study, here called life cycle region. It gets clear that the damage generated by the functional unit would be higher if all the processes were in the Uniform World Model and less if they were all in the Tarragona region (situated at the Mediterranean coast). In the case of the ecological damage parameter, only few accurate values in the weighted eco-matrix are known. For the REW ecosystem area the diagonal elements sum up to 0.0155 km^2 per TJ electricity. That is a relative small value because the studied region is not sensitive to acidification. For the GWP a weighted eco-vector is obtained. The sum of the vector components yields to $2.57*10^5$ kg CO_2-equivalents per TJ electricity. This result is similar for electricity generation by fossil fuels (CIEMAT, 1997; Frischknecht et al. 1996).

Table 5.5: Damage Profile for the current situation 2 (with advanced acid gas treatment system)

REGIONS	Electricity	Transport	CaO	Ash treatm.	Scrap-metal rec.	Incineration	*TJ Electricity*
EEC in €							
Spain	119	0	0	0	0	0	-
South Europe	0	8.184	0	0	0	0	-
Tarragona	322	1.470	2.765	610	1.050	15.020	*21.237*
Tarragona	322	1.470	2.765	610	1.050	15.020	*21.237*
Uniform World Model	328	2.130	1.827	816	1.522	23.740	*30.363*
Tarragona	322	1.470	2.765	610	1.050	15.020	*21.237*
Life Cycle Region	*119*	*8.184*	*2.765*	*610*	*1.522*	*15.020*	**28.220**
REW in km2							
Spain	-	-	0	0	-	0	-
South Europe	-	-	0	0	-	0	-
Tarragona	-	-	9,12E-05	4,31E-04	-	1,50E-02	*1,55E-02*
Tarragona	-	-	9,12E-05	4,31E-04	-	1,50E-02	*1,55E-02*
Uniform World Model	-	-	0	0	-	0	*1,55E-02*
Tarragona	-	-	9,12E-05	4,31E-04	-	1,50E-02	*1,55E-02*
Life Cycle Region	-	-	*9,12E-05*	*4,31E-04*	-	*1,50E-02*	**1,55E-02**
GWP in kg CO_2-equiv.							
World	1,54E+03	9,56E+03	8,11E+03	5,93E+03	6,83E+03	2,25E+05	**2,57E+05**

The damage profile of the current situation can now compared to the situation of the MSWI in 1996 without advanced acid gas removal system, as a different process design option. The damage profile for the situation in 1996 is presented in Table 5.6.

Table 5.6 Damage profile for the situation in 1996 (without advanced acid gas treatment system)

REGIONS	Electricity	Transport	CaO	Ash treatm.	Scrap-metal rec.	Incineration	*TJ Electricity*
EEC in €							
Spain	114	0	0	0	0	0	114
(South) Europe	0	7.812	0	0	0	0	7.812
Tarragona	308	1.400	0	100	1.001	20.890	23.699
Tarragona	308	1.400	0	100	1.001	20.890	23.699
Uniform World Model	313	2.030	0	133	1.450	28.260	32.187
Tarragona	308	1.400	0	100	1.001	20.890	23.699
Life cycle Region	114	7.812	0	100	1.450	20.890	**30.365**
REW in km^2							
Spain	-	-	0	0	-	0	0,0000
(South) Europe	-	-	0	0	-	0	0,0000
Tarragona	-	-	0	7,03E-05	-	0,0150	0,0151
Tarragona	-	-	0	7,03E-05	-	0,0150	0,0151
Uniform World Model	-	-	0		-	0	0,0000
Tarragona	-	-	0	7,03E-05	-	0,0150	0,0151
Life cycle Region	-	-	0	7,03E-05	-	0,0150	**0,0151**
GWP in CO_2-equiv.							
World	1,47E+03	9,11E+03	0,00E+00	9,69E+02	6,51E+03	2,14E+05	**2,32E+05**

While the Eco-indicator 95 shows a reduction of 60% in the total score between the former and the current situation (chapter 3), the environmental damage estimations show less reduction. The external environmental costs decreases 10%, but the REW augments 3 % and the GWP increases 10 %. These both damage indicators do not show reduction due to the increased transport and the reduced energy efficiency in the MSWI process chain after the installation of the advanced gas treatment system, as explained in chapter 3, what affect the emissions of NOx and CO_2, which are crucial for these damage factors.

A comparison of the external environmental costs with the Eco-indicator 95 for the plant before and after the installation of the advanced gas removal system is given in Figure 5.18. It can be seen that the environmental external cost estimations give much more weight to the transport processes. The transport contributes with approximately 25 % to the external environmental costs before the installation of the advanced gas treatment system and with 30 % afterwards, while according to the Eco-indicator 95 the transport adds less than 10 % in the current situation and less than 5 % in the former situation In contrast, the incineration process is much more predominant in accordance with the Eco-indicator 95 methodology. In the former situation more than 90 % of the Eco-indicator 95 are attributed to the incineration and still in the current situation it counts for 80 %, while the external environmental cost estimations only assign 70 % for the situation in 1996 to the incineration process and in the current situation even less, namely a little more than 50 %. This different relation between the transport and the incineration process explains also why the external environmental costs have decreased less than the Eco-indicator 95 score. Another important process in the current situation is the CaO production. Moreover, in Figure 5.18 it can be seen that the selected relevant processes make really up to nearly 100 % of the total impact.

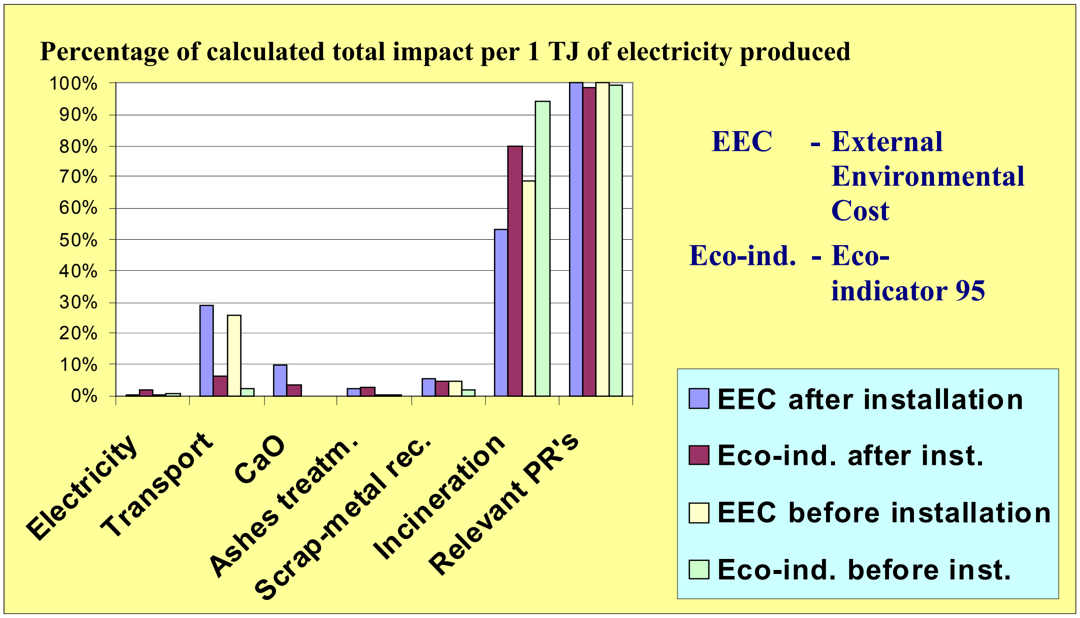


Figure 5.18: Comparison of external environmental cost with Eco-indicator 95 for plant before and after the installation of advanced acid gas removal system

Eco-efficiency

Based on the environmental cost estimations obtained in the previous sections the eco-efficiency of the process chain can be calculated. According to expression (5.9) in Figure 5.15 as additional element we need therefore the production costs.

According to information from SIRUSA (Nadal, 1999) the production costs are the following:

- Former situation: 32,000 €/ TJ
- Current situation: 37,000 €/ TJ

These costs are based on the yearly total operation costs of the MSWI, including the financial costs that they pay for the investments made. The production costs are higher than the market price of 0,07 €/ kWh, the deficit is paid by price for the waste treatment, what can be considered as a service the MSWI is doing to the society.

Applying of the presented formula the following eco-efficiency for the process chain is obtained:

- Former situation: $\eta = 5\ \%$
- Current situation: $\eta = 25\ \%$

The production of electricity by waste incineration is not very eco-efficient, if avoided environmental charges for the waste treatment are not considered, as it is done in the existing LCA of the MSWI process chain in Tarragona (see chapter 3 for further explications).

Nevertheless the eco-efficiency increases significantly by the installation of an advanced acid gas removal system.

5.6 POSSIBLE APPLICATIONS OF THE PRESENTED METHODOLOGICAL FRAMEWORK

Application patterns of LCA have been studied by Frankl & Rubik (1999). In this section we will provide suggestions of possible applications of the presented methodology. We consider that the developed framework is of interest for the following stakeholders:

- Public administration (to justify taxes and to optimise waste management strategies)
- Financial entities (to get to know possible risks of payments)
- Companies (to contribute to a system of continuous environmental improvement, to choose between sites of a process chain with an environmental perspectives and to demonstrate the diminution of environmental damages by the inversion in abatement technologies in a life cycle viewpoint)
- Consumers and general community (to have the possibility to obtain products and services of minor pollution and to learn about the environmental damages behind industrial process chains)

5.6.1 PUBLIC ADMINSTRATION

Table 5.7 shows more in detail the possible applications for public administration.

Table 5.7: Applications for the public administration

Application	Example	Optional element
Technology assessment	Energy production, waste treatment, transport	None
Green tax	Electricity, transport or all other types of products and/or services	Using external environmental costs
Public policy planning	Future scenario assessment of energy production, waste treatment, transport	Possibly eco-efficiency
Environmental justice	Waste incineration, land fill, cracker	None
End of life management	Waste incineration, land fill, integrated waste management planning (similar to Wizard by Ecobilan, 1999), but on a more detailed level with regard to spatial aspects (second generation)	Possibly eco-efficiency

Applications are among the mentioned green tax by using external environmental costs, technology assessment in general, public policy planning and, in particular, environmental justice and end of life management. The idea behind the application "end of life management" is to develop a somehow second generation of integrated waste management software, like Wizard (Ecobilan, 1999), that considers also the setting problem, which is quite related to environmental justice. By combining site-specific aspects with life cycle considerations new plans for waste management can really assess the transport processes and their routes in relation to the waste treatment facilities and their sites and the respective environmental damages in an integrated manner.

5.6.2 FINANCIAL ENTITIES

The possible applications for financial entities (bancs and assurances) are presented in Table 5.8.

Table 5.8: Applications for financial entities (bancs and assurances)

Application	Example	Optional element
Planning of new plants or changes	Waste incineration, land fill, cracker	Possibly accidents
New insurance of existing plant	Waste incineration, land fill, cracker	Possibly accidents
Business strategic planning	Future scenario assessment	Possibly eco-efficiency

Main applications are the assessment of risk by the new plants or changes in existing ones, and the respective insurance for plants as well as the evaluation of strategic business plans. Important optional elements are accident simulations and questions of eco-efficiency.

5.6.3 PRODUCTION AND OTHER SERVICE COMPANIES

Table 5.9 represents the numerous possible applications for production and other service companies.

Table 5.9: Applications for production and other service companies

Application	Example	Optional element
Planning of new plants	Waste incineration, land fill, cracker	Possibly accidents
EMAS: to show that the best solution is chosen (improvement of performance and liability by extension of EMAS)	Change of process in chemical industry	Possibly eco-efficiency
Documentation of environmental profile's improvement towards the public	Advanced gas treatment system in waste incineration plant	None
Process improvement	Change of process in chemical industry, additionally flue gas cleaning in waste incineration	Possibly eco-efficiency
Determination of most problematic processes	Production with a huge amount of complex steps	Especially dominance analysis, possibly accidents
Chain responsibility (assessment of supplier and waste treatment companies and their technology)	Supply chain management, avoiding use of undesired substances and occurrence of accidents	Especially dominance analysis, possibly accidents
Determination of most problematic emission and medium emitted to	Identification of points for environmental improvement: e.g. waste reduction in chemical industry plant	Especially dominance analysis
Claim for subventions	Trade association of waste management industry	Possibly uncertainty analysis
Marketing strategies by communication of environmental profile to consumers	Waste incineration, chemical industry plant	None
Business strategic planning	Future scenario assessment	Possibly eco-efficiency
End of life management	Waste incineration, land fill, integrated waste management planning (similar to Wizard by Ecobilan, 1999), but on a more detailed level with regard to spatial aspects (second generation)	Possibly eco-efficiency
Optimisation of setting (new plant)	Waste incineration, land fill, chemical industry	Possibly ERA

A lot of applications are proposed in Table 5.9 for which it could be of interest for companies to use the developed methodological framework. Especially important are the integrated end of life management and in general the parallel optimisation of the life cycle and the settling problem from an environmental point of view, as described in section 5.6.1. Furthermore, all the questions related to the chain responsibility with the assessment of supplier and waste treatment companies and their technology are important. The latter includes the evaluation of accident risks with regard to processes like overseas petrol transport for which several big companies were declared responsible in the last years.

A part from typical production companies the methodological framework is certainly of relevance for the waste treatment companies who want to document the environmental profile's improvement towards the public. Moreover, these companies would probably be interested in gaining the confidence of the public concerning new plants or changes in the overall waste management plan. The developed methodology can be of great assistance in this task by the generation of an important amount of quite objective, relevant information that should allow to find a convincing solution for all interested parties.

5.6.4 CONSUMERS AND SOCIETY IN GENERAL

The possible applications for consumers and the society in general are shown in Table 5.10.

Table 5.10: Applications for consumers and society in general

Application	**Example**	**Optional element**
Education and communication as potential common ground for discussion	Increased information available by all considered applications	Possibly all optional elements
Eco-labelling and/ or environmental product declarations	The system would be a quite accurate way to obtain relevant results about the environmental damages caused by product systems, but at the moment it seems not to be very practicable for this purpose. Nevertheless, in the future this information might be available due to the advances of the information technologies.	Possibly uncertainty analysis

The main application consist in the education and communication as potential common ground for discussion about environmental damage estimations due to certain problematic industrial process chains, like the waste incineration that has been part of the public discussion on environmental aspects for the last decade.

Another potential application could be in the future eco-labelling and/ or environmental product declarations since the developed methodology would be a quite accurate way to obtain relevant results about the environmental damages caused by product systems, but at the moment it seems to be not very practicable for this purpose. Nevertheless, in the future this information might be available due to the advances of the information technologies.

6 SITE-DEPENDENT IMPACT ASSESSMENT BY STATISTICALLY DETERMINED GENERIC CLASSES

We present here a provisional approach to overcome the disadvantages of the site-generic and site-specific methods. The approach presented here is one of the recently developed site-dependent impact assessment methods that can be considered as a trade-off between exactness and feasibility. As in site-specific approaches fate, exposure and effect information are taken into account, but indicators are calculated that are applicable for classes of emissions sites rather than for specific sites. That is the trade-off between the accurate assessment of the impacts and the practicability of spatial disaggregation for impact assessments in a life-cycle perspective. We note the developing nature of these methods, but we regard them as the most promising alternative for future estimations of environmental damages in industrial process chains. We adapt the approach in a way so that it perfectly fits into the methodology presented in the previous chapter. A flowchart for site-dependent impact assessment is proposed and the algorithm of the methodology is applied using the calculated site-dependent impact indicators.

6.1 RECENTLY DEVELOPED APPROACHES FOR SITE-DEPENDENT IMPACT ASSESSMENT

The consideration of spatial differentiation in LCIA was proposed first by Potting & Blok (1994). However, it took time until developments for site-dependent impact assessment have been made in an operational way especially for acidification and eutrophication, such as Potting (2000) and Huijbregts & Seppälä (2000). Moreover, several approaches have been presented for human health effects due to airborne emissions. Exemplary damage factors for a number of European countries are provided by Spadaro & Rabl (1999). Potting (2000) establishes impact indicators that take into account different release heights, population density, substance characteristics such as atmospheric residence time and dispersion conditions. The release height is statistically linked to several industrial branches. Typical meteorological data for four zones within Europe are used, but the issue of local dispersion conditions is not addressed, and no operational guidance for the determination of population densities based on sufficiently detailed data is provided. Moriguchi & Terazono (2000) present an approach for Japan where the meteorological conditions are set to be equal for all examples. Nigge (2000) presents a method for statistically determined population exposures per mass of pollutant that considers short-range and long-range exposure separately and allows to take into account dispersion conditions and population distributions using sufficiently detailed data in a systematic way. Impact indicators are derived that depend on the settlement structure class and the stack height. However, a general framework that is valid also for other receptors is not proposed, the dispersion conditions are considered to be equal for all classes in the case study and a quite simple dispersion model is used.

In this study a general applicable framework for site-dependent impact assessment by statistically determined receptor exposures per mass of pollutant is proposed which addresses some of the mentioned shortages. Receptor density and dispersion conditions are related to form a limited number of representative generic spatial classes, as suggested by Potting & Hauschild (1997) and recommended by Udo de Haes et al. (1999). The basis for the classification is statistical reasoning, assuming no threshold (Potting, 2000). For each class and receptor incremental receptor exposures per mass of pollutant can be calculated. Finally, the incremental exposures are transformed into damage estimations. The general framework was applied to the case of population exposures due to airborne emissions in the Mediterranean region Catalonia/ Spain. A differentiation was made with regard to dispersion conditions, stack height and atmospheric residence time, a sophisticated dispersion model was applied and a geographic information system (GIS) was used.

6.2 General framework for site dependent impact assessment as basis for statistically determined generic classes

Udo de Haes (1996) distinguished the following dimensions of impact information that are relevant for Life Cycle Impact Assessment:

- Effect information
- Fate and exposure information
- Background level information
- Spatial information

The first two dimensions directly refer to the cause-effect chain. The last two dimensions can be interpreted as additional conditions related to the processes in the considered chain.

Based on Udo de Haes' (1996) proposal, Wenzel et al. (1997) suggested the relation of dimensions of impact information and levels of sophistication presented in Figure 6.1. The first dimension in this figure is in accordance with the proposal of Udo de Haes (1996). Only the exposure information has been removed. In this way, the second dimension covers all information that is connected to the source (emission, distribution/ dispersion, and concentration increase). The third dimension comprises the third dimension from Udo de Haes (1996) and, in addition, all other types of information about the receiving environment and/ or target system (background concentration, exposure increase, sensitivity of the target system, etc.).

With each of the three dimensions, the characterisation modelling addresses different levels of sophistication (see section 2.3.9). In the effect analysis the sophistication increases from the use of legal threshold standards via the application of NOEC to the integration of slope in the impact factors. These factors may include none, some or comprehensive fate information. The same holds true for the information on the target system.

Dimensions of impact information	**Levels of sophistication**				
1. Information on the effect of a pollutant	standard	→	NOEC	→	slope
2. Pollutant fate information	none	→	some	→	full
3. Target information	none	→	some	→	full

Figure 6.1: Dimensions of impact information and levels of sophistication in Life Cycle Impact Assessment (Wenzel et al, 1997)

Category indicators for toxicity, but also for other endpoint-orientated impacts, are generally calculated by multiplying the emitted mass M of a certain pollutant p with a fate and exposure factor F and an effect factor E, i.e. the slope of the dose-response and exposure-response functions (Nigge, 2000). In the general case where the transfer of the pollutant across different environmental media or compartments (e.g. air, water, soil) needs to be considered, the category indicator incremental damage ΔD_p^{nm} [damage (like cases for cancer, a for YOLLs and DALYs or € for external environmental costs)], characterising the effect in the compartment m of the pollutant p emitted in the initial compartment n, is given in expression (6.1).

$$\Delta D_p^{nm} = E_p^m \cdot F_p^{nm} \cdot M_p^n \tag{6.1}$$

where

M_p^n mass of pollutant p [kg] being emitted into the initial medium n (air, water or soil)

F_p^{nm} fate and exposure factor for substance p emitted into the initial medium n and transferred into medium m, in the form of [$m^2 * a/ m^3$] or [$m^3 * a/ m^3$] depending on the compartments considered, and taking into account the propagation, degradation, deposition, intermedia transfer and food chain or bioconcentration routes

E_p^m effect factor [damage/ $m^2 *$ (µg/ m^3) $*$ a)] representing the severity of the impact due to the substance p in medium m (air, water, soil or food chain).

The ratio $\Delta D_p^{nm} / M_p^n$ is called damage factor [damage/ kg] (Hofstetter, 1998).

The release and target compartments are linked by the different fate and exposure routes. For example, an emission to the compartment air can have impacts in the target compartments air (inhalation), soil (via deposition) and water (absorption). The pollutants can be transported farther to other target compartments on other routes (soil-plant etc.). Principally, there is a large variety of possible routes between release and target compartments.

Expression (6.1) does not explicitly consider the distribution and the number of receptors affected by the pollutants. Depending on the impact category, the receptor may be among others human population, material surface, crop yield and sensitive ecosystem area or also fish population. The effective receptor density $\rho_{eff,r}^m$ is introduced into the expression in order to relate the distribution of receptors r to the distribution of the respective pollutant in the environment. Expression (6.1) then reads as follows in expression (6.2)

$$\Delta D_{p,r}^{nm} = E_{p,r}^m \cdot \rho_{eff,r}^m \cdot F_p^{nm} \cdot M_p^n \tag{6.2}$$

where

$\Delta D_{p,r}^{nm}$ incremental damage caused [damage] due to pollutant p being emitted into the initial medium n (air, water or soil) on the receptor r in the target compartment m (air, water, soil or food chain)

$\rho_{eff,r}^m$ effective density of receptor r [receptors/ m^2] in target medium m (air, water, soil or food chain); that is for the receptor human population [persons/ m^2] and for material surface [m^2 maintenance surface/ m^2], while this means for the fish population in the compartment water [fishes/ m^3], i.e. we do not have to consider a density, but a concentration

$E_{p,r}^m$ effect factor [damage/ receptors $*$ (µg/ m^3) $*$ a)] representing the severity of the impact due to the substance p in medium m (air, water, soil or food chain) on receptor r.

In the context of this framework the incremental receptor exposure (ΔRE) is then defined as the product of the number of receptors exposed to a certain concentration during a certain period of time, as shown in expression (6.3).

$$\Delta RE_{p,r}^{nm} = \Delta D_{p,r}^{nm} / E_{p,r}^{m} = \rho_{eff,r}^{m} \cdot F_{p}^{nm} \cdot M_{p}^{n} \tag{6.3}$$

where

$\Delta RE_{p,r}^{nm}$ incremental receptor exposure during a certain period of time [receptors ∗ (µg/ m³) ∗ a] due to pollutant p being emitted into the initial medium n (air, water or soil) on the receptor r in the target compartment m (air, water, soil or food chain); that is, for example, for airborne emissions with the receptor human population [persons ∗ µg/ m³ air ∗ a] and for the receptor material surface [maintenance surface ∗ µg/ m³ air ∗ a] and for water with the receptor fish population [fishes ∗ mg/m³ water ∗ a].

If the dose-response or exposure-response function is linear or the emission source contributes only marginally to the background concentration the incremental damage becomes independent of the time pattern of the emission. It only depends on the total mass emitted. For a detailed mathematical derivation see Nigge (2000) and for the general idea of marginality and linearity see Potting (2000). For the purpose of this study it is assumed that the incremental damage $\Delta D_{p,r}^{nm}$ is independent of the time pattern of the emission. The incremental damage can then be calculated by expression (6.4) using an incremental receptor exposure per mass of pollutant emitted $I_{p,r,i}^{nm} = \Delta RE_{p,r,i}^{nm} / M_{p,i}^{n}$. The index i refers to an emission situation rather than to an emission site only. The emission situation is determined by the emission site and by the source type like the stack height for emissions to air or the release depth into the lake or river for emissions to water.

$$\Delta D_{p,r,i}^{nm} = E_{p,r}^{m} \cdot M_{p,i}^{n} \cdot I_{p,r,i}^{nm} \tag{6.4}$$

where

$\Delta D_{p,r,i}^{nm}$ incremental damage caused [damage] due to pollutant p being emitted into the initial medium n (air, water or soil), at the emission situation i, on the receptor r in the target compartment m (air, water, soil or food chain)

$M_{p,i}^{n}$ mass of pollutant p [kg] being emitted into the initial medium n (air, water or soil) at the emission situation i

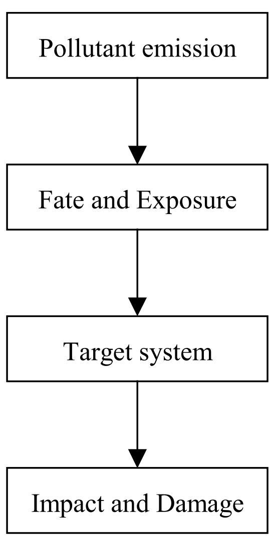
$I_{p,r,i}^{nm}$ incremental receptor exposure per mass of pollutant emitted [receptors * (µg/ m³) * a/ kg] due to the amount of pollutant p being emitted into the initial medium n (air, water or soil), at the emission situation i, on the receptor r in the target compartment m (air, water, soil or food chain)

The mass of pollutant emitted M, the fate and exposure factor F, the effect factor E and the effective receptor density ρ_{eff} used in the presented framework are directly related to the causality chain as illustrated in Figure 6.2. Such a comprehensive framework is the basis for using high levels of sophistication in the different dimensions of impact information corresponding to the scheme described in Figure 6.1.

This impact information should be based on a certain spatial differentiation with regard to the processes in the chain and include a minimal amount of additional data on the corresponding geographic situation. In accordance with Potting (2000) we believe that the relevance of life cycle impact assessment can be enhanced by the inclusion of a few general site-parameters in the assessment procedure and we support to call this site-dependent impact assessment.

For the effect analysis we propose to use the dose-response and exposure-response functions described in section 2.2.1. The fate information should be obtained by using pollutant dispersion and long-range transport models and/ or multimedia fate models. The target information needed corresponds to the receptor density that describes the sensitivity of the target, but we do not consider that background information is necessary assuming that residual risk is what we want to address in LCIA and that linear dose-response and exposure-response functions exist, at least for priority pollutants (see for a further discussion of this issue section 2.2.1, Crettaz et al. (2002), Nigge (2000) and Potting (2000).

Causality chain **Descriptors**



$M^{n}_{p,i}$ = mass of pollutant p being emitted into the initial medium n (air, water or soil) at the emission situation i (location, source type)

F^{nm}_{p} = fate and exposure factor for substance p emitted into the initial medium n and transferred into medium m, partitioning between compartments, dilution, immobilisation, removal/ degradation, environmental concentration increase (without consideration of background level)

$\rho^{m}_{eff,r}$ = effective density of receptor r in target medium m (air, water, soil or food chain); that is the sensitivity of the system

$E^{m}_{p,r}$ = effect factor representing the severity of the impact due to the substance p in medium m (air, water, soil or food chain) on receptor r, including concentration- effect curves (the dose- and exposure-response functions) as well as weighting schemes (e.g. monetisation and DALY) to express different types of impacts in the same way as damage

Figure 6.2: Relation of the factors used in the presented framework with the general cause-effect chain for the environmental impact of an emitted compound (developed based on Jager et al, 1994)

Considering only one pollutant p and one receptor r as well as one release compartment n and one target compartment m, then $I^{nm}_{p,r,i}$ = I_i (= ΔRE_i/ M) the incremental receptor exposure per mass of pollutant emitted [receptors * (μg/ m^3) * a/ kg], representing the concentration increment multiplied with the receptors during a certain time period divided through the mass of pollutant. In two-dimensional polar coordinates (r,φ) around the emission situation i within a suitable cartographic projection of the earth's surface this can be written as expression (6.5) (Nigge, 2000).

$$I_i = \frac{1}{Q}\int_0^R r \int_0^{2\pi} \Delta c_i(r,\varphi)\,\rho_i(r,\varphi)\,d\varphi\,dr \qquad (6.5)$$

where

$Q = M_i/T$	constant emission rate [kg/ a] with M_i as mass of one pollutant [kg] being emitted at the emission situation i and T as the duration of the emission [a]
r	radius [m]
R	outer boundary of the modelling area [m]
$\Delta c_i(r,\varphi)$	concentration increment at a receptor point with the polar co-ordinates (r, φ) for the emission situation i [µg/ m^3]
$\rho i(r, \varphi)$	receptor density at a receptor point with the polar co-ordinates (r, φ) for the emission situation i [receptors/ m^2]

Generally speaking the integration of expression (6.5) should be carried out over the whole planet. As most of the pollutants even in the air compartment do not disperse over the whole planet due to their residence time and dispersion characteristics in the environment and since the calculation effort shall be kept appropriate, R is chosen in a way so that most of I_i is covered by the area. Another limitation is the spatial range of the models chosen which often do not allow the calculations to be extended over a certain limit. The source characteristics influence the choice of R as well. The higher the stack in the case of emissions to air the farther the pollutant is transported and therefore the greater R has to be chosen.

The idea of the methodology further developed and applied in this study is to statistically define classes of emission situations i, the impact of which differs significantly from class to class but for which the deviation of impact between the emission situations covered by each class is small. The overall number of classes shall be kept small to enable an easy handling.

On the one hand, the neglect of the spatial distribution around the emission point i of the receptor density ρ(r,φ) is the main reason for the discordance between the potential impact results and the actual impacts. To be precise that means for example that in conventional LCIA potentials the impact is the same for an air pollutant over the ocean than for one in a big city. On the other hand, the corresponding dispersion conditions in the respective medium and the resulting concentration increment $\Delta c_i(r,\varphi)$ are relevant for the occurrence of damages. In order to relate these main factors for the estimation of environmental damages in a process chain-perspective the present method proposes to form representative classes of receptor density and dispersion conditions. This classification has to be based on statistical reasoning. For each class receptor incremental exposures per mass of pollutant ΔRE have to be calculated.

In a next step, the receptor incremental exposures per mass of pollutant ΔRE can be converted into damage estimates through an effect analysis based on dose-response and exposure-response functions and, if desired by the decision-maker, by the application of a weighting scheme to express different types of impacts in the way of an aggregated damage.

6.3 DEFINITION OF GENERIC CLASSES FOR HUMAN HEALTH EFFECTS OF AIRBORNE EMISSIONS

For the case of human toxicity impacts of airborne emissions the receptors are the persons of a population and the release as well as the target compartment is air. Hence, $\rho^{m}_{eff,r}$ corresponds to the population density and the receptor incremental exposures $\Delta RE^{nm}_{p,r,i}$ is then called population incremental exposures to airborne pollutants ΔPE, expressed in units of [persons µg/ m^3·a]. PE is also called the pressure on human health (Nigge, 2000).

As lot of laws and regulations with respect to emission limitation and pollution prevention exist for pollutants emitted into the air and solely transported by the air, this study is confined to the air as only release and target compartment. Considering as receptor only human beings, for one pollutant, expression (6.4) can then be simplified to expression (6.6).

$$\Delta D_i = E \cdot M_i \cdot I_i \tag{6.6}$$

where

- ΔD_i incremental damage caused [damage] due to the exposure to one pollutant being emitted at the emission situation i
- E effect factor [damage/ persons * (µg/ m^3) * a)] representing the severity of the impact due to one pollutant
- M mass of one pollutant [kg] being emitted at the emission situation i
- I_i incremental population exposure per mass of one pollutant emitted at the emission situation i [persons * (µg/ m^3) * a/ kg]

Expression (6.5) can be divided into two integrals accounting for the short-range dispersion and the long-range transport to the outer boundary of the modelling area R:

- $I_{i,near}$ short-range contribution to the incremental receptor exposure per mass of one pollutant emitted at the emission situation i [persons * (µg/ m^3) * a/ kg]
- $I_{i,far}$ long-range contribution to the incremental receptor exposure per mass of one pollutant emitted at the emission situation i [persons * (µg/ m^3) * a/ kg]

The reason for this procedure is the fact that the concentration increment is usually highest within the first kilometres around the stack. Therefore, the impact indicator is very sensitive to the receptor density close to the stack. The population density can vary strongly within only a few kilometres. As an example may serve the fact that the population density often rapidly decreases from a big city to the countryside. The population exposure therefore is subject to drastic changes within a few kilometres as well. The long-range contribution, however, only depends on the average receptor density of the region where the pollutants are transported to and is not very much subject to changes on a local scale because the concentration increment is small due to dilution on transport and because the concentration does not change very much with the distance. Long-range contributions are known well studied for pollutant like SO_2 and NOx (due to their importance for the acidification in regions far away from the emission source). Strong differences for the long-range exposure are likely to appear between densely inhabited areas such as Western and Middle Europe and scarcely inhabited regions like Scandinavia. Therefore, as a good approximation country averages for $I_{i,far}$ seem to be appropriate.

A major problem with deriving $I_{i,near}$ is the fact that $\Delta c_i(r,\varphi)$ depends very much on the meteorological conditions, especially the wind direction that can vary significantly for the different emission sites within a few kilometres. It is therefore desirable to eliminate φ in order to simplify expression (6.5). By Nigge (2000) it is assumed that $\Delta c_i(r, \varphi)$ and $\rho_i(r, \varphi)$ are not correlated and that the population density is independent of the angle φ if the emission sites considered in each class are spread over a large area. In this way, there is no preferred direction for the spatial variation of the population density. Moreover, the simplification of expression (6.7) can be introduced (Nigge, 2000):

$$\Delta c_i(r) \equiv \frac{1}{2}\pi \int_0^{2\pi} \Delta c_i(r,\varphi)\, d\varphi \tag{6.7}$$

According to Nigge (2000), expression (6.5) then reads like expression (6.8).

$$I_{C,near} = \frac{1}{Q}\int_0^{100km} r \int_0^{2\pi} \Delta c_C(r)\, \rho_C(r)\, 2\,\pi dr \tag{6.8}$$

In expression (6.8), 100 km is a value for orientation; it is proposed as the limit between the short- and long-range contribution to I, the incremental receptor exposure per mass of one pollutant emitted. The index C indicates that the expression (6.8) does not refer to a single emission situation i, but to a generic class of emission situations that are statistically correlated with respect to dispersion conditions and population density and which have the same source characteristics. A further mathematical analysis is given in (Nigge, 2000). Remembering that $I_{i,far}$ is calculated as a country or regional average, the overall impact indicator for each class then reads like expression (6.9).

$$I_C \equiv I_{C,near} + I_{far} \tag{6.9}$$

where

I_C incremental population exposure per mass of one pollutant emitted [persons * (µg/ m^3) * a/ kg] at the generic class of emission situations that are statistically correlated with respect to dispersion conditions and receptor density and which have the same source characteristics

In order to compute the impact indicator for each class the following elements of expression (6.8) have to be calculated:

$\rho_C(r)$ radial receptor density [receptors/ m^2] for each class, in our case population density [persons/ m^2]

$\Delta c_C(r)$ radial concentration increment profile for each class [µg/ m^3]

The definition of classes of meteorological conditions and the derivation of generic meteorological data files to calculate the radial concentration increment profile $\Delta c_C(r)$ are questions of fate analysis. The definition of classes of population densities and the calculation of the radial population density $\rho_C(r)$ belong to the exposure analysis.

6.4 Generic classes of population exposures for human health effects due to airborne emissions in Catalonia

In this study the fate and exposure analysis is carried out for the Mediterranean region Catalonia next to the sea, what influences significantly the way on how to apply the presented method.

6.4.1 Fate analysis to characterise the dispersion conditions

This section discusses the transport of the airborne pollutants from the emission source to the receptors. For the purpose of short-range dispersion modelling (dispersion up to the radius of 100 kilometres around the stack) the program ISCST-3 (US-EPA, 1995) incorporated in the software BEEST (Beeline, 1998) was used. The calculations for the long-range transport were carried out using the program EcoSense (IER, 1998).

For the modelling of the short-range exposure only primary pollutants are considered due to the long formation time of secondary pollutants. In order to calculate the radial concentration $\Delta c_C(r)$ to derive I_{near} a statistical set of meteorological data has to be used. Since emissions occurring in a life cycle usually cannot be assigned to the calendar time when they happen, only mean average pollutant concentrations on an annual basis are calculated. The BEEST programme requires input data as presented in Table 6.1, where also the data used in this study are indicated. Test runs have shown that the concentration increment results are not sensitive to the ambient air temperature. The mixing height is calculated as a function of the stability class (VDI, 1992). An equal distribution of wind directions is assumed. Therefore, as derived by Nigge (2000), the combined frequency distribution of wind speed and stability class is independent of the wind direction.

Table 6.1: Values required by BEEST and respective data used in this study

Values required by BEEST	Data used in this study
Hourly values	
Wind speed	Weibull parameters, average annual wind speed to generate hourly wind speed values
Wind direction	Random values in intervals of 15º
Ambient air temperature	Annual average: 287 K
Stability class	Derived from a combined frequency distribution of wind speed and stability class, using hourly wind speed data
Rural and Urban mixing height	Set to be equal for rural and urban areas
Friction velocity	Function of wind speed
Monin-Obukhov length	Function of the stability class
Surface roughness	Rural character of Catalonia: 0.3 m
Precipitation	Not taken into account: 0 mm
Fixed values	
Elevation of modelling area	Entirely flat
Release heights	5 m, 100 m and 200 m
Exit temperature of stack	423.15 K
Stack diameter	5 m
Volume flow	$\dot{V} = 26862 \cdot 10^{0.0196 \cdot h_{stack}}$ [Nm^3/h] (6.10)
Exit velocity	Function of volume flow and stack diameter
Emission mass flow	$\dot{M}_p = \dot{V} * C_{threshold,p}$ [kg/ h] (6.11) where $C_{threshold,\ p}$ legal threshold concentration of pollutant p

The remaining task is to determine a statistical distribution of wind speed u and stability class s for each class of meteorological conditions in Catalonia. If the distribution parameters of the Weibull distribution are known, an hourly wind speed file can be generated from the average annual wind speed. Manier (1972) states that the distribution of stability class and the distribution of wind speed classes are correlated. As a consequence of this, the only parameter required as additional input for using the impact indicator is the mean annual wind speed of the considered district. Harthan (2002) shows the derivation of the Weibull parameters for Catalonia required to generate hourly wind speed data from a mean annual wind speed value and describes the analysis of the combined frequency distribution of wind speed and stability class. The variance coefficient of the average annual wind speed data for Catalonia (Cunillera, 2000), with an average for the years from 1996 to 1999 for all 14 stations considered, is high (47,4%). Thus, taking into account only the mean annual average value to represent all average annual wind speed data considered for Catalonia does not suffice to describe the Catalonia situation as a whole. The formation of wind speed classes on the district level is required, (see overview of districts in Catalonia in Figure 6.3). The districts are assigned the following codes for the classes of wind speed: A: 0 - 2 m/s, B: 2 - 3 m/s and C: 3 - 4 m/s. As only wind speed data from 14 stations were provided in a detailed manner, average annual wind speed data for 48 stations for the years 1997 and 1998 are taken from http://www.gencat.es/servmet. The overall distribution of wind speed classes among the districts in Catalonia can be seen in Figure 6.3 and the portions of districts belonging to each class of meteorological conditions are shown in Table 6.2.

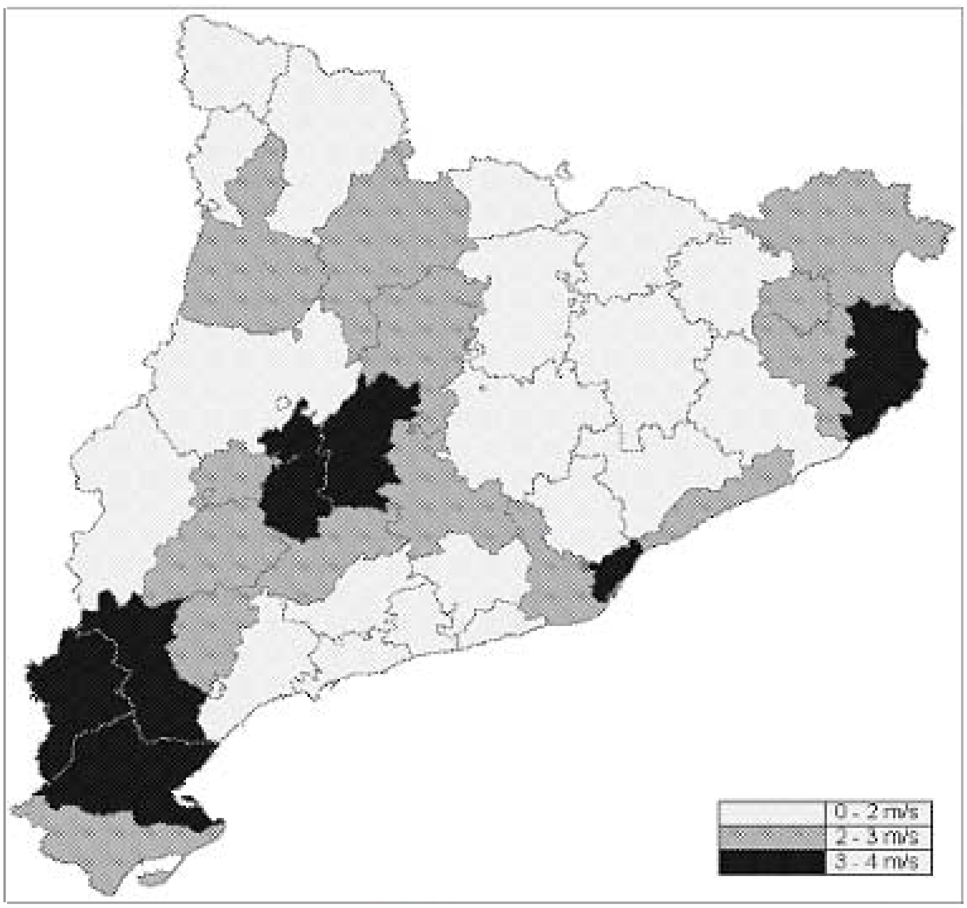


Figure 6.3: Classes of average wind speed for all districts in Catalonia (Data of 1997 and 1998)

Table 6.2: Number and share of districts belonging to each class of wind speed

Average wind speed class	Districts	Share
0 - 2 m/s	20	48.8%
2 - 3 m/s	14	34.1%
3 - 4 m/s	7	17.1%
0 - 4 m/s	31	100%

Mass flow, volume flow and exit velocity determine the outcome of the concentration calculations of ISCST-3. However, unlike in EcoSense, the concentration increment calculated by BEEST does not change linearly with changing certain parameters of source characteristics. Therefore, statistical values for these parameters (volume flow and mass flow) have to be defined.

A set of nine different industrial processes with stack heights ranging from 10 m to 250 m are evaluated with respect to their volume flows. The correlation between volume flow and stack height has a trend line that can be described with the potential approach in expression (6.10), where $\dot{V}$ is the volume flow in [Nm^3/h] and h_{stack} the stack height in [m], the regression coefficient r^2 equals 0.799. Expression (6.10) is only a rough approximation to calculate the volume flow. Nine processes is not at all a representative statistical number that allows to make general conclusions.

The mass flow of each pollutant is obtained from the volume flow and the respective threshold of each pollutant according to expression (6.11), where $\dot{M}_p$ is the mass flow of pollutant p, $c_{threshold,p}$ the legal threshold concentration of pollutant p in flue gas. The threshold values for municipal waste incinerators are taken from the Catalan Decret 323/1994 that includes the European Guideline 89/369/EEC. In order to apply the correct threshold for the organic substances considered the share of total organic carbon (TOC) of every pollutant is calculated and like this the threshold of TOC is adapted to each single organic substance considered.

The use of a threshold at this stage is probably not the best solution; in further works it should be tried to base the mass flow also on statistical reasoning according to industry types. An alternative would be for example to use the mean average emission value of the respective industry.

As a matter of fact, for dispersion the decisive parameter relating to the release height is not the stack height itself, but the effective stack height h_{eff} which also takes into account the momentum rise and the buoyancy rise of the plume and is automatically calculated by BEEST. In order to make the results of this work comparable to other studies that relate the impact indicators to the h_{eff} (such as Nigge (2000)), the effective stack height is calculated for the indicators derived in this study (Table 6.3). The calculation of h_{eff} is carried out according to Israel (1994). The comparison of the results from different studies has to be done with care, thoroughly checking the congruence of the applied algorithms for $h_{eff.}$

Table 6.3: Effective stack heights depending on wind speed and actual stack height

h_{stack} [m]	h_{eff} [m]		
	0 - 2 m/s	2 - 3 m/s	3 - 4 m/s
5	133	60	43
100	297	195	167
200	676	435	379

Figure 6.4 shows an example of a concentration curve for PM10 and the wind speed class 2-3 m/s, 100 km around the stack of 5 m height. The resolution of the grid is higher close to the stack than farther away in order to represent the sharp decrease of concentration close to the stack.

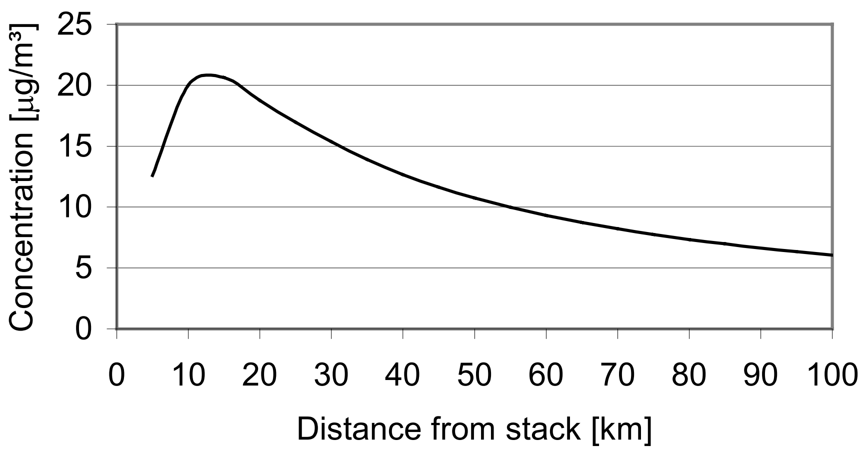


Figure 6.4: Concentration curve for PM10, 100 km around the stack of 5 m height (average annual wind speed of 2.5 m/s)

6.4.2 Exposure Analysis to determine the population density around the emission source

The basic classification of regions and districts according to population density is taken from Nigge (2000) in order to make results comparable. As region in this context the province level in Catalonia is considered. Furthermore, the districts are considered. The combination of region and district classes is called settlement structure classes. For Catalonia four classes have been identified to represent statistically sufficiently the Catalonian situation (Figure 6.5), which is characterised by the rapid decrease of the population from the city boundaries towards the countryside. For the purpose of this study, population data for Catalonia are taken from the geographic information system (GIS) MiraMon (Pons & Maso, 2000).

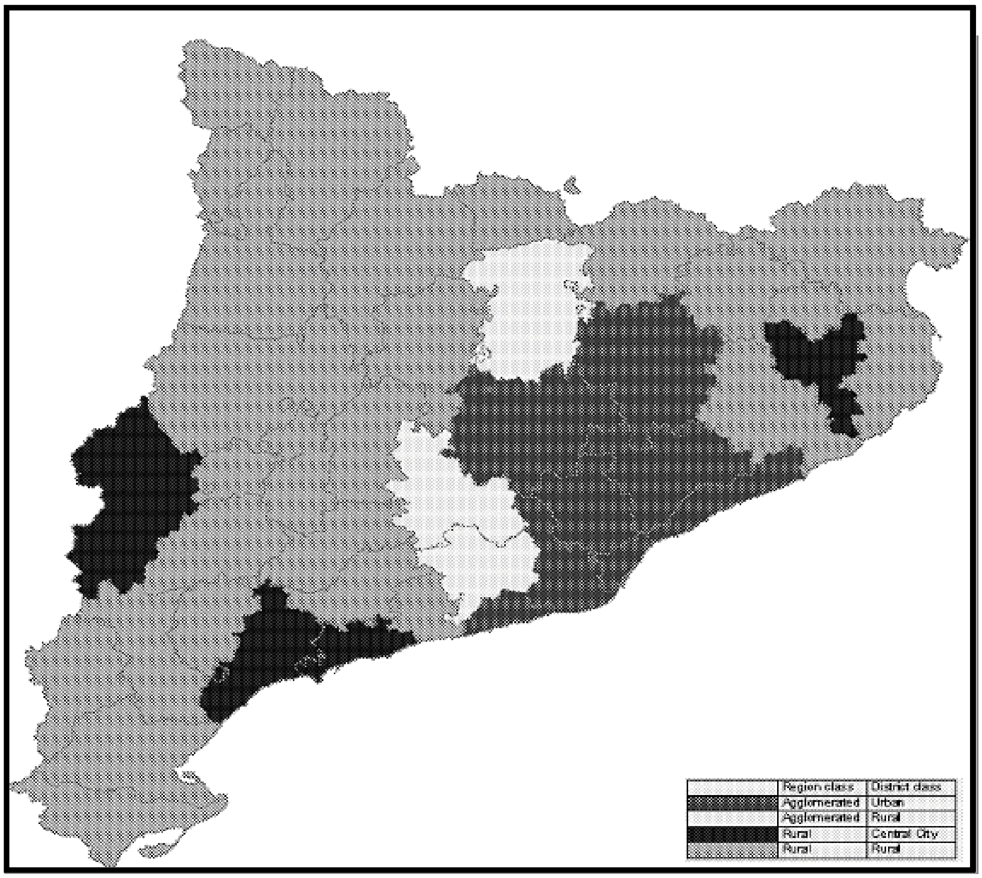


Figure 6.5: Settlement classes in Catalonia

For the calculation of the radial population density the radius of 100 km around each municipality in Catalonia is considered which corresponds to the modelling area of the short-range transport covered by the Gaussian dispersion model used. Annuli are formed in intervals of 10 km that leads to ten annuli around each municipality. For each municipality, every municipality lying around will be counted to the respective annulus if its centre lies within the considered annulus. The interval of 10 km is chosen assuming that the linear extension of the municipalities in Catalonia is in the range of up to 10 km to both sides of the centre of the municipality. Assuming that every municipality in Catalonia has the shape of a circle, the maximum area allowed for one municipality lying in the centre of all annuli (origin) is 314 km^2. As the biggest municipality in Catalonia (Tremp) comprises about 270 km^2, this assumption is valid.

The calculation of the radial population density bears two problems: On the one hand the radial population density in districts close to the sea is not independent of the direction one looks at (the population density at the coast falls down abruptly to ρ=0 persons/km^2). On the other hand, no data were available for the municipalities lying within 100 km outside of the regional borders of Catalonia.

The first problem implies that the total population of all municipalities lying in the considered annulus is divided by the area of the annulus. Like this, the fact that there is a considerable area around the municipality with ρ=0 persons/ km^2 (municipalities close to the sea) is respected. Due to the absence of population in the sea, the overall population density of the annulus is reduced.

In order to solve the second problem for the calculation of the population density in the municipalities of the area adjacent to the area of study for (France, Andorra, Aragón and the Comunidad Valenciana) population data on a district level is used rather than the population data for the municipalities themselves taken from http://www.turisme.ad, http://www.asam-aragon.org, http://www.annuairemairie.com and http://195.53.26.41/valencia/mapa.htm.

As most of the districts are bigger than municipalities uncertainties are introduced. The average area of the Catalonian districts is 778 km^2. The size of the districts in Aragón and Valencia are supposed to be in the same range, since the administrational structure is similar. This corresponds to a circle of less than 16 kilometres of radius. As discussed above, the area of every municipality is assumed not to exceed a circle with a radius of 10 kilometres if the municipality lies in the centre of the circle. The average area of districts definitely exceeds this value. This means that the population density of the annuli often is determined by the whole population of one district although this district extends over more than one annulus and therefore should “assign" its population to more than one annulus. It is assumed, however, that the uncertainties introduced are not too big, since the average exceeding of the centre circle is within a tolerable range. Moreover, this procedure is assumed to be valid because it is only chosen to include the area adjacent to Catalonia while Catalonia itself is dealt using a higher resolution, so that the overall uncertainties related to the radial population density are considered to be quite low.

To calculate the radial population density for every municipality the distance between each municipality has to be calculated to assign the municipalities to the respective annuli. These data are taken from the mentioned GIS MiraMon. MiraMon provides the coordinates in UTM (Universal Transverse Mercator Grid System) units expressed in meters as well as the respective population of each municipality. The coordinates describe the outer limits of each municipality. Assuming circular areas, the coordinates of the centre of the municipalities are calculated forming the average of all coordinates describing the outer limits. The latitudes and longitudes of the centres for the district outside of Catalonia are converted into UTM coordinates using one of the converters available in the internet.

After calculating the distances between the municipalities it is known which municipality lies in which annulus around a considered municipality. Using data about the population of each municipality, the population of each annulus around a certain municipality and the population density for each annulus and each municipality can be calculated.

The radial population density is then calculated for every generic class subsuming the population density for each annulus of each municipality belonging to the respective class and dividing the sum by the number of municipalities considered in this class. Figure 6.6 shows the radial population density for each class graphically. Each generic class is assigned a number from I to IV: I Agglomerated – urban, II Agglomerated – rural, III Rural - Central city and IV Rural – rural.

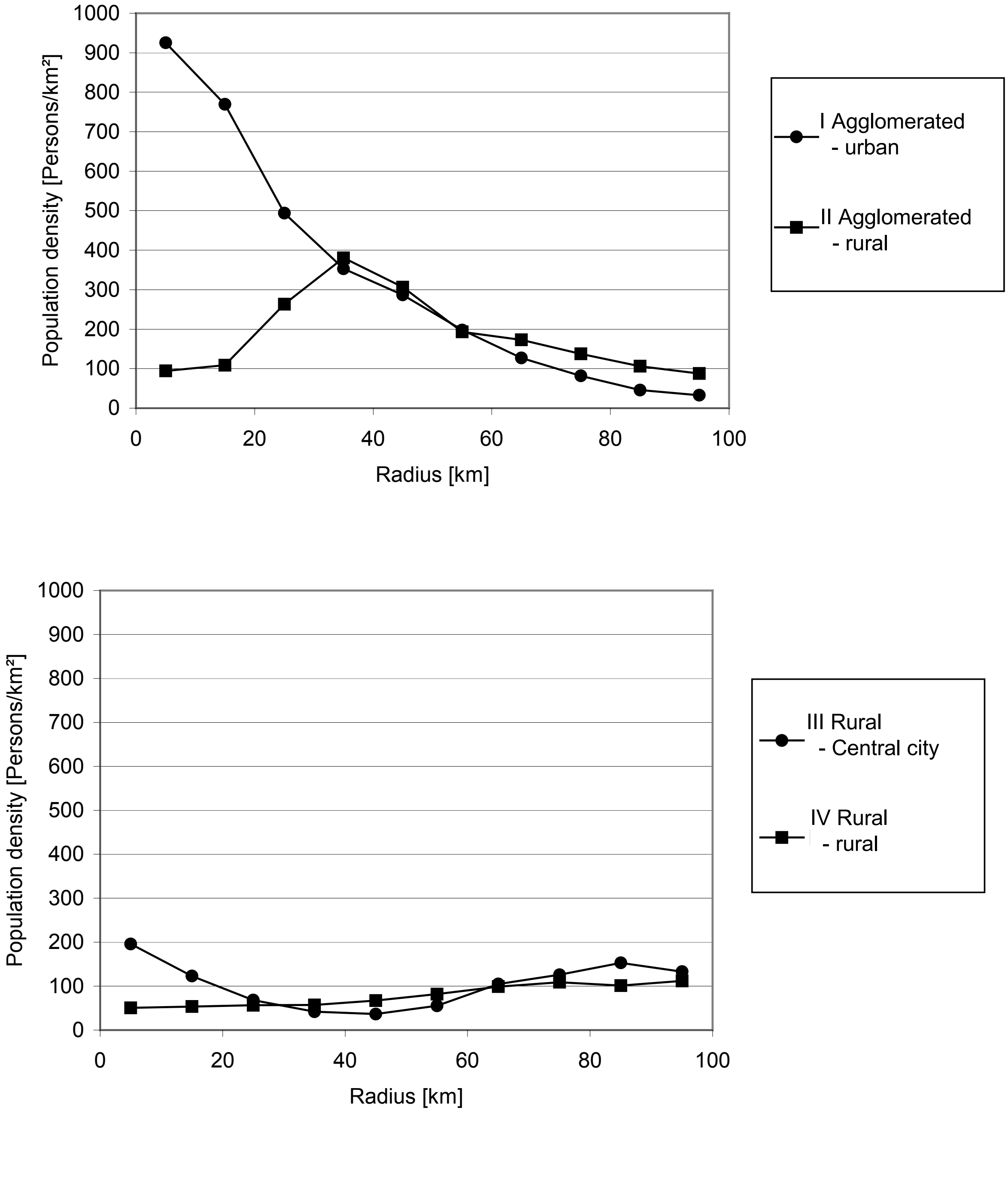


Figure 6.6: Radial population density distributions for the settlement structure classes in Catalonia

The radial population density for the agglomerated region was calculated for the province of Barcelona. For urban districts of the province, to which the city centre of Barcelona belongs, the radial population density is decreasing rapidly from the centre towards the outer radius of 100 kilometres. The main reason for this behaviour is the concentration of population within greater Barcelona whereas outside of the province the districts are scarcely populated.

The curve of the rural districts within the province of Barcelona shows the typical behaviour of a rural region close to a big city. During the first kilometres the population density is rather low. Between 30 and 40 kilometres away from a fictive stack in this district there is a peak of population density. This can be explained with the proximity of Barcelona city, which influences also the radial population density for rural districts in the province of Barcelona.

Within the rural regions Girona, Lleida and Tarragona there are districts classified "central city" and "rural". For central cities to which belong the districts of Tarragonès, Baix Camp, Segrià and Gironès the population density also decreases from the centre towards the countryside. However, the population density is not as high in the centre as for respective districts in the agglomerated region. In the particular situation of Catalonia, after a minimum between 40 and 50 kilometres the population density reaches a local maximum between 80 and 90 kilometres. A look on Figure 6.5 explains this behaviour: all rural regions are located around the agglomerated region. Especially the central cities in the rural regions are within a distance of 100 kilometres of the agglomerated region.

For the rural districts of the rural regions, the population density is much below the average of Catalonia (ρ=190 persons/ km^2 at the radius r=0), which is self-explaining in rural districts. Towards r=100 kilometres the population density increases. This explains the settlement structure of Catalonia. Statistically, there is a big city (central city) within a distance of 100 kilometres from each rural district in a rural area. A look again on Figure 6.5 underlines this reasoning.

From the curve for each class it can be seen that the population density reaches a similar value for all classes at a radius of 100 kilometres. Therefore, it is statistically justified to treat the long-range transport in an averaged manner (regional average) for all considered districts and regions and to set the limit between short-range and long-range modelling at this distance.

6.5 Effect Analysis to transform the incremental exposures into damage estimations: the DALY concept and External Costs

The effect analysis links the results of the fate and exposure analysis to the damage due to the emitted pollutant. The effect analysis is independent of the fate and exposure analysis and bases on epidemiological and toxicological studies as well as socio-economic evaluation.

The effect factor represents the number of health incidences (like asthma cases, cancer cases or restricted activity days) per person·˙ exposure time·˙ concentration. The dose-response and exposure-response functions used in this study are taken from IER (1998) and Hofstetter

(1998). In this study only carcinogenesis and respiratory health effects are taken into account. These are considered to be the main contributors to the overall human health effects due to environmental pollution (Krewitt et al., 1998).

In order to aggregate different health effects into a single indicator, the Disability Adjusted Life Years (DALY) concept developed by Murray & Lopez (1996) is implemented; see section 2.2.3 for further explanation. However, also an economic valuation by external costs could be applied easily, for example the scheme used by the European Commission (1995). The DALY value does not only depend on the pollutant and the type of disease, but also on the socio-economic perspective of each person. Thompson et al. (1990) introduce the concept of the cultural theory for different perspectives. Hofstetter (1998) distinguishes individualist, egalitarian and hierarchist cultural perspectives to represent the archetypes of socio-economic behaviour and relate it to the DALY concept. Corresponding to age-weighting for the different cultural perspectives, in the economic evaluation discount rates of 0 % and 3 % can be chosen. An overview of the different factors for the conversion of population exposure values into damage estimates is given in Table 6.4.

Table 6.4: Conversion factors between impact indicators and DALY and external costs

Pollutant	DALY I	DALY E	DALY H	Costs[a]	Costs[b]
Acetaldehyde	2.7E-7	4.1E-7	4.1E-7	0.05	0.05
As	2.2E-4	3.5E-4	3.5E-4	43.1	31.5
BaP	1.3E-2	2.0E-2	2.0E-2	2497	1825
1,3-butadiene	3.4E-5	5.2E-5	5.2E-5	6.6	6.6
Cd	2.7E-4	4.1E-4	4.1E-4	51.7	37.8
Formaldehyde	1.5E-6	2.3E-6	2.3E-6	0.33	0.23
Ni	5.0E-5	7.8E-5	7.8E-5	10.3	7.3
NO_x	-	2.5E-6	-	0.27	0.43
SO_2	-	5.4E-6	5.4E-6	0.57	0.89
PM2.5	2.8E-4	2.9E-4	2.9E-4	136.5	135.6
PM10	1.7E-4	1.8E-4	1.8E-4	79.7	79.1
Nitrate	9.0E-5	1.8E-4	1.8E-4	79.7	79.1
Sulfate	2.8E-4	2.9E-4	2.9E-4	132.7	131.8

DALY: [a/(persons·µg/m^3)], external costs: [€/(persons·µg /m^3)]

I: Individualist (age-weighting (0,1)); E: Egalitarian (no age-weighting (0,0));

H: hierarchist (no age-weighting (0,0)); a: discount rate 0%; b: discount rate 3%

6.6 Calculation of site-dependent human health impact assessment factors for Catalonia

The classes derived in the fate and exposure analysis are given combined class codes in order to enable an easy finding of the desired impact indicator for the combined classes. For example, the impact indicator for class III B describes the impact for a central city in a rural region with annual wind speed conditions between 2 and 3 m/s. Moreover, the stack height is mentioned as well as the effect analysis method chosen.

6.6.1 Short-range exposure

Calculations are carried out for every combination of pollutant, stack height and generic meteorological file. Concentration curves like in Figure 6.4 are overlapped with the curves of radial population density (Figure 6.6) for each generic class of population density and integrated over the modelling area, see expression (6.8). The integration over the modelling area is done by summing up the products of concentration increment, population density and area for all values lying in the chosen grid.

6.6.2 Long range exposure

Using the EcoSense software (IER, 1998) the concentration increment is calculated for eight grid cells of fictitious emission sites covering Catalonia. The runs are carried and for the stack heights 5 m, 100 m and 200 m. As a first step the background population exposure is calculated (zero emission run). For this purpose a no emission run is carried out for each of the eight grid cells. The concentrations obtained are the background values for the respective grid cells. The multiplication of the background concentration and the population of each grid cell leads to the background population exposure per grid cell. The results are then summed up over all grid cells what leads to the overall background population exposure for each emission cell considered.

The next step is to carry out the calculations for all considered emission cells and for all stack heights. The population exposure is calculated in the same way as for the background population exposure. In order to obtain the long-range contribution due to an incremental emission of a pollutant, first the background population exposure has to be subtracted from the calculated population exposure. Since the short-range population exposure is calculated using the BEEST software, the population exposure within the radius of 100 kilometres from the stack has to be subtracted as well. For this purpose, the population exposure of the emission cell as well as of part of the adjacent cells is subtracted.

After summing up the population exposure of each grid cell, subtracting the background and the short-range exposure, the sum is divided by the emitted mass. The result is the population exposure per mass of emitted pollutant for the long-range transport I_{far} for each pollutant, each emission cell and each considered stack height. As the modelling area is comparatively small (which explains the existence of only eight emission cells), the long-range exposure per mass of emitted pollutant does not differ very much for the considered emission cells. The variation coefficient for the different emission cells lies between 7.75% and 23.49%. Therefore an average Catalonian value is calculated for every pollutant and for every stack height and applied as I_{far} for the whole of Catalonia.

I_{far} for pollutants such as acetaldehyde and 1,3-butadiene that are not included in the EcoSense software are approximated by their atmospheric residence time. For that purpose I_{far} is calculated for a set of particles with different diameters and atmospheric residence times (Table 6.5). A linear regression is carried out and an approximation formula is calculated in expression (6.12).

$$I_{far} = \exp\left(-0.2043 \cdot \ln \tau)^2 + 2.3916 \cdot \ln - 7.3174\right) \tag{6.12}$$

where

I_{far} long-range contribution to the population exposure per mass of emitted pollutant [Persons $*$ µg/ m^3 $*$ a/ kg]

τ atmospheric residence time [h].

Table 6.5: Diameter and atmospheric residence time τ of several particles (Nigge, 2000)

Pollutant (diameter)	τ [h]
PM 2,5	135
PM 10	22
PM (d=4-10mm)	15
PM (d=10-20mm)	11
PM (d>20mm)	3

6.6.3 OVERALL IMPACT INDICATORS

The overall impact indicator I_{total} is the sum of I_{near} and I_{far}. Table 6.6 shows the results for several pollutants, including the values for I_{far} in order to show the long-range contribution to I_{total}. The impact indicators for the secondary pollutants nitrate and sulphate refer to the mass of primary pollutant emitted, i.e. to NO_x and SO_2, respectively, and are represented by the results for the long-range exposure that show only very slight variations for different stack heights. The average values for all stack heights are 0.25 persons $*$ µg/m^3 $*$ a/kg for nitrate and 0.13 persons $*$ µg/m^3 $*$ a/kg for sulphate, respectively.

Table 6.6: Site-dependent human health impact indicators for several pollutants and stack heights [Persons * µg/m³ * a/kg]

Pollutant		Acetaldehyde			1,3-butadiene			NO_x			SO_2			PM2.5			PM10		
Stack height		5 m	100 m	200 m	5 m	100 m	200 m	5 m	100 m	200 m	5 m	100 m	200 m	5 m	100 m	200 m	5 m	100 m	200 m
Class																			
District	Wind																		
I_{far} =	Long range contribution to the incremental receptor exposure per mass of pollutant																		
Cat	A,B,C	0.05	0.05	0.05	0.06	0.06	0.06	0.06	0.06	0.06	0.17	0.18	0.18	0.60	0.60	0.60	0.06	0.06	0.06
I_{total} =	Total incremental receptor exposure per mass of pollutant																		
I	A	3.29	0.18	0.06	3.67	0.22	0.07	4.72	0.26	0.08	5.93	0.43	0.19	6.52	0.91	0.61	5.61	0.43	0.09
I	B	2.47	0.29	0.09	2.69	0.32	0.10	3.27	0.36	0.11	3.97	0.51	0.23	4.44	0.95	0.65	3.64	0.42	0.11
I	C	2.14	0.30	0.09	2.31	0.33	0.11	2.76	0.36	0.12	3.30	0.51	0.24	3.72	0.93	0.65	2.97	0.40	0.11
II	A	1.24	0.16	0.05	1.50	0.19	0.07	2.32	0.24	0.07	3.42	0.42	0.19	3.87	0.92	0.61	3.06	0.41	0.09
II	B	0.96	0.20	0.08	1.13	0.23	0.09	1.59	0.27	0.10	2.22	0.43	0.22	2.64	0.87	0.64	1.93	0.34	0.10
II	C	0.84	0.20	0.08	0.97	0.23	0.10	1.32	0.26	0.11	1.82	0.41	0.23	2.22	0.84	0.64	1.54	0.31	0.10
III	A	0.71	0.10	0.05	0.83	0.12	0.07	1.23	0.15	0.07	1.88	0.31	0.19	2.29	0.79	0.61	1.67	0.24	0.08
III	B	0.57	0.12	0.06	0.65	0.15	0.08	0.88	0.17	0.08	1.28	0.31	0.20	1.69	0.75	0.62	1.08	0.21	0.08
III	C	0.50	0.12	0.06	0.57	0.15	0.08	0.75	0.17	0.08	1.07	0.30	0.21	1.48	0.73	0.62	0.88	0.19	0.08
IV	A	0.45	0.09	0.05	0.56	0.12	0.07	0.90	0.15	0.07	1.49	0.30	0.19	1.90	0.76	0.60	1.27	0.22	0.08
IV	B	0.37	0.11	0.06	0.44	0.13	0.08	0.64	0.15	0.08	1.01	0.29	0.20	1.42	0.73	0.62	0.82	0.18	0.08
IV	C	0.33	0.11	0.06	0.39	0.13	0.08	0.55	0.15	0.08	0.84	0.28	0.20	1.25	0.71	0.62	0.66	0.17	0.08
Cat	A	1.22	0.12	0.05	1.41	0.15	0.07	1.97	0.18	0.07	2.76	0.34	0.19	3.22	0.82	0.61	2.51	0.29	0.08
Cat	B	0.94	0.16	0.07	1.06	0.19	0.08	1.38	0.21	0.09	1.85	0.36	0.21	2.28	0.80	0.63	1.62	0.26	0.09
Cat	C	0.82	0.17	0.07	0.92	0.19	0.09	1.16	0.21	0.09	1.54	0.35	0.21	1.95	0.78	0.63	1.31	0.24	0.09

District population densities: I: Agglomerated - urban; II: Agglomerated - rural; III: Rural – Central city; IV: Rural – rural; Cat: Catalonian average

Wind speeds: A: 0-2 m/s; B: 2-3 m/s; C: 3-4 m/s

Generally speaking, it can be said that the population exposure is smaller the lower the population density is. Another general correlation is the fact that an increasing stack height leads to a decreasing impact indicator. Since Catalonia is quite populated in comparison with other regions in Spain, since every pollutant being deposited on the Mediterranean does not lead to any human health effect via inhalation and since the modelling area of EcoSense is limited to Europe (which therefore neglects harmful effects of Spanish emissions to North Africa, for instance), every molecule or particle going into the long-range transport favours the decrease of the overall population exposure.

Another obvious effect is the fact that I_{total} decreases with higher wind speeds. Moreover, the influence of atmospheric residence time and decay can be derived. If one compares the impact indicators for PM10, PM2.5, NO_x and SO_2, it can be seen that the values decrease from PM2.5 over SO_2, PM10 to NO_x according to their atmospheric residence time and decay rate. The span between the highest and the lowest value of I_{total} for each pollutant ranges from factor 10 for PM2.5 to factor 70 for PM10. This can be explained with the fact that PM2.5 accounts for a much greater long-range contribution to the population exposure as PM10 due to its long atmospheric residence time. Therefore, the lowest value of PM2.5 is determined by the comparatively high long-range contribution, which leads to the comparatively small span between highest and lowest value.

Using the dose-response and exposure-response functions, physical impacts (e.g. cases of chronic bronchitis) per mass of pollutant can be calculated. Applying these functions and the respective unit values, it is possible to convert the impact indicators into DALY and external costs per kilogram pollutant using the conversion factors in Table 6.4.

6.6.4 ESTIMATES FOR OTHER REGIONS AND OTHER STACK HEIGHTS

As process chains often comprise processes outside of Catalonia, an approximation formula for other regions in Spain is presented in the following. It is supposed that the long-range exposure for other regions in Spain does not vary significantly from the values in Catalonia. Therefore, holds the expression (6.13).

$$I_{far,other\ regions} = I_{Catalonia}. \tag{6.13}$$

With respect to the short-range exposure, it is supposed that there are significant variations due to the population density. Nigge (2000) presents an approach to approximate indicators according to meteorological conditions as well. Due to a lack of data and time, only the population density is considered here. For this purpose, the population density for all other provinces (the administrational level between region and district) is calculated. The impact indicators are calculated using the average impact indicator for Catalonia according to expression (6.14). As no distinction is made with respect to meteorological conditions, the respective values of $I_{near,Catalonia}$ for the different classes of meteorological conditions are weighted with the share of occurrence in the districts of Catalonia (Table 6.2).

$$I_{near,otherprovince} = I_{near,Catalonia} \cdot (\rho_{otherprovince} / \rho_{Catalonia}) \tag{6.14}$$

Finally, it shall be mentioned also that for emission heights which are different than the given ones an interpolation can be carried out.

6.7 Site-dependent impact assessment in the methodology of Environmental Damage Estimations in Industrial Process Chains

On the basis of the calculation of site-dependent impact assessment factors described beforehand, in this section the presented site-dependent method is introduced as a further element into the methodology of Environmental Damage Estimations for Industrial Process Chains outlined in chapter 5. Site-dependent impact factors can be perfectly used in the mathematical framework developed and are especially recommended for transport processes. The main task remaining is to establish a flowchart of the site-dependent impact assessment by statistically determined generic classes that can be included in the overall methodology.

Hence, in Figure 6.7 such an overview of the different working steps for this site-dependent method is given:

For the calculation of the impact factors first the considered region has to be divided into classes. In this chapter this has been done for the receptor human population. However, in principle, this can be done also for other receptors, as proposed in the general framework described in section 6.2.

In the fate & exposure analysis each class is divided into the near (<= 100 km) and the far contribution (> 100 km). For the near contribution on the one hand the radial concentration increment for each pollutant and on the other hand the radial receptor density is calculated. As transport model on the local scale the Gaussian dispersion model ISCST-3 as incorporated in BEEST has been applied in the application to Catalonia. The multiplication of both results obtained yields the receptor exposure per class and pollutant $I_{\text{near, pollutant, class}}$. In a similar way for the far contribution first the average concentration increment on a continental grid is calculated for each pollutant. Then this result is multiplied with the average receptor density in each grid. Finally the receptor exposure per class and pollutant $I_{\text{far, pollutant, region}}$ is obtained for the region under study.

In a next step I_{far} and I_{near} are added and so finally the overall receptor exposure per pollutant for each class and a specific region $I_{\text{total, pollutant, class\&region}}$ is determined.

By carrying out the consequence analysis factors of physical impact parameters are obtained. When additionally applying the selected weighting and aggregation scheme the factors can also directly be expressed in the form of environmental costs or damage endpoint indicators. These both steps have been summed up in section 6.5 as effect analysis.

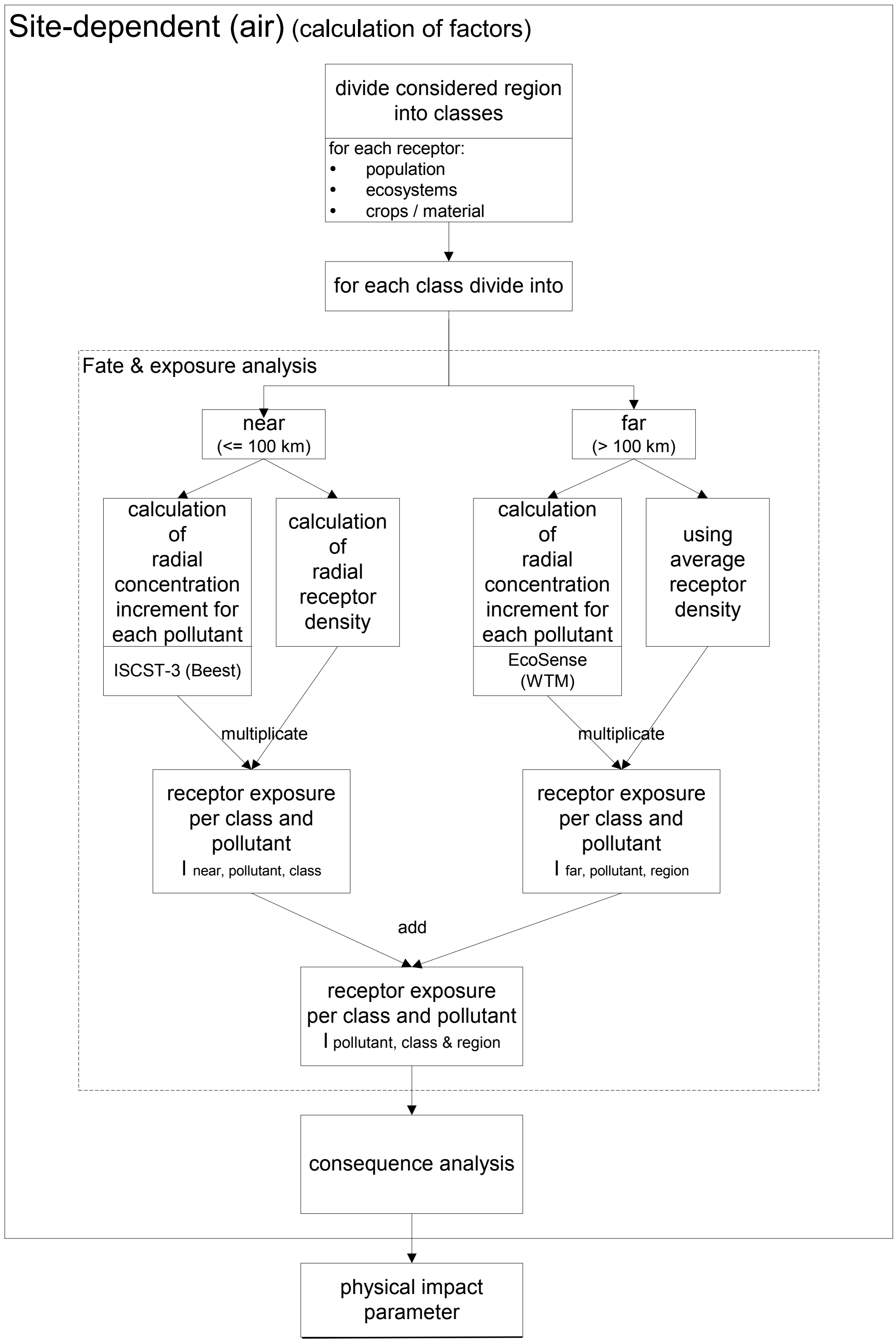


Figure 6.7: Site-dependent impact assessment for air emission

6.8 APPLICATION OF THE METHODOLOGY BASED ON THE CALCULATED SITE-DEPENDENT IMPACT ASSESSMENT INDICATORS TO THE WASTE INCINERATION PROCESS CHAIN

In this section the impact indicators derived in section 6.6.3 are applied to the industrial process chain of the case study, the municipal waste incineration plant of Tarragona and the related transport processes. Chapter 3 describes the data of the municipal waste incinerator as well as the related transport processes. A comparison between the results for the applied impact indicators and the results for a mid-point indicator is provided and discussed.

1) Goal and scope definition

Two situations and one scenario of the MSWI process chain are compared to estimate avoided environmental damages on human health.

Situation 1 is the basis for all comparisons. It describes the SIRUSA incinerator at its state in 1996. It includes the treatment of scrap in Madrid, ash treatment at the company TRISA in Constantí and the disposal of the treated ash and the slag in Pierola and Reus, respectively. In the ash treatment plant cement is mixed with the ash. The cement is transported from S^{ta} Margarida i els Monjos.

Situation 2 is the situation after the installation of an advanced acid gas removal system in 1997. Besides the features in situation 1 there is an additional transport of CaO for the acid gas treatment. CaO is transported from S^{ta} Margarida i els Monjos to the waste incinerator in Tarragona. The data for the situations 1 and 2 are taken from the existing LCA study (STQ, 1998).

As a third alternative a scenario has been created based on the modular model for the waste incineration process described briefly in chapter 3 (Hagelüken, 2000). Besides the features in situation 2, a fictitious DeNox system is installed to eliminate NOx emissions. Ammonia for this purpose is transported from the industrial area of Tarragona to the waste incinerator. It has to be stressed that this is a scenario with a not validated MSWI model. However, it is introduced into this case study in order to discuss possible results of a further improvement of the flue gas treatment. This alternative is called scenario 3.

The function of the MSWI is its service of treating the municipal solid waste of the Tarragona region. In this context, the functional units presented here are 1 year of treated waste (which equals 153 467 t). However, also 1t of treated waste as well as 1 TJ and 1 kWh of produced electricity could be chosen.

According to the lessons learned in the application of the methodology of environmental damage estimations for industrial process chains in the previous section the focus in this study was restricted on the transport and the incineration process. Hence, it is not necessary to carry out a dominance analysis and no cut-off criteria have to be defined. However, for the comparison of the results with midpoint indicators two LCIA methods with the human toxicity potential (HTP) are used: CML (Heijungs et al., 1992) and EDIP (Hausschild & Wenzel, 1998).

For the weighting of the human health impacts the physical impact parameter “population exposure” and the DALY concept with the Egalitarian perspective are applied. As only the AoP human health is taken into account here no further aggregation can be carried out.

The options uncertainty analysis, environmental risk assessment, accidents and eco-efficiency are not considered.

2) LCI analysis

In order to get the LCI data of this abbreviated waste incinerator process chain the volume flow [Nm^3/a] of the waste incineration plant is multiplied with the unit emission files for the waste incinerator in the different cases, and the overall transport [tkm] is multiplied with the unit emissions files for the transport. These unit emission files provide the emission of different air pollutants per Nm^3 and per tkm, respectively. The above-mentioned data and the unit emission files for SIRUSA can be found in Table 3.2; the unit emission files for the transport processes are presented in Table 6.7. The table only lists the pollutants for which impact indicators are available and that are considered crucial for environmental problems related to transport. The unit emission data for the waste incinerator for the situations 1 and 2 stems from the existing LCA study (STQ, 1998} and for scenario 3 from Hagelueken (2001).

Table 6.7: Unit emissions for the transport processes (Kern, 2000)

Pollutant	Emission [kg/tkm]
Benzo[a]pyrene	4,00E-10
Cd	2.10E-09
NMVOC	4.40E-04
NOx	3.12E-03
PM10	2.00E-04
SO_2	2,08E-04

The most obvious result is that the overall tkm increases significantly from alternative 1 over 2 to 3. By this, the additional transport processes in somehow compensate the decrease of the emissions due to the improvement of flue gas treatment. In how far the transport compensates the damages avoided through more powerful gas treatment has to be shown by the environmental damage estimations.

3) LCIA method

The human toxicity potential (HTP) according to the CML method (Heijungs et al., 1992) and the EDIP methods (Hauschild & Wenzel, 1998) is calculated (the nomenclature of the latter is changed). Further explanations of the HTP can be found in chapter 2. In Table 6.8 the HTP results for the three alternatives are presented.

Table 6.8: HTP results for the three alternatives

Process	CML [-/a]	[%]	EDIP [m3/a]	[%]
Situation 1				
MSWI	912885	97.9	$2.85 \cdot 10^{12}$	98.3
Transport	19365	2.1	$4.88 \cdot 10^{10}$	1.7
Total	932251	100.0	$2.90 \cdot 10^{12}$	100.0
Situation 2				
MSWI	160492	88. 0	$8.95 \cdot 10^{11}$	94.2
Transport	21885	12. 0	$5.51 \cdot 10^{10}$	5.8
Total	182378	100.0	$9.50 \cdot 10^{11}$	100.0
Scenario 3				
MSWI	43361	64.5	$2.31 \cdot 10^{11}$	79.3
Transport	23912	35.5	$6.02 \cdot 10^{10}$	20.7
Total	67272	100.0	$2.91 \cdot 10^{11}$	100.0

4) Dominance analysis and spatial differentiation

No dominance analysis is necessary for this application. The medium is air since the impact factors have been calculated for that medium. The pollutants are those for which site-dependent impact factors have been determined in section 6.6.3. They correspond largely to the pollutants that have been identified as predominant in chapter 4. The processes that are studied are those that have been resulted most important in the previous section: the incineration process and the transport.

Additionally to the assignation of the corresponding region to each process, in this application the transport process has to be differentiated with respect to the districts crossed. That is the example for what is meant with expression (5.8). The size of the region, here the district level, determines the number of regions, by which a mobile process, i.e. transport, is represented in the eco-technology matrix.

Table 6.9 describes the transport routes for every process and the districts crossed. Table 6.10 summarises the transport related to each process taking place in each class of population density and meteorology class as well as the distance in the provinces outside of Catalonia. For the abbreviation of classes see section 6.6. The emission height for the municipal waste incinerator is 50 m and for the transport processes 5 m. That means, the product of mass and distance of transport [tkm] is differentiated with respect to the districts crossed. Using the transport distances in each class from Table 6.10 and the tkm presented in chapter 3, it can be calculated how many tkm refer to which class of population density and meteorological conditions. The multiplication with the unit emission file of transport leads to the spatially differentiated LCI, the eco-technology matrix.

Table 6.9 Transport routes and districts crossed for the considered processes

Purpose	Route	Districts crossed (class)
Municipal waste	Tarragonès - Constantí	Tarragonès (III A)
Slag	Constantí - Reus	Tarragonès (III A), Baix Camp (III A)
CaO	Els Monjos - Constantí	Alt Penedès (II A), Baix Penedès (IV A), Tarragonès (IIIA)
Ammonia	Tarragona - Constantí	Tarragonès (III A)
Scrap treatment	Constantí - Madrid	Tarragonès (III A), Alt Camp (IV A), Conca de Barberà (IV B), Garrigues (IV B), Segrià (III A), Huesca (HU), Zaragoza (ZZ), Soria (SO), Guadalajara (GU), Madrid (M)
Ash treatment	Constantí	Tarragonès (III A)
Ash disposal	Constantí - Pierola	Tarragonès (III A), Baix Penedès (IV A), Alt Penedès (II A), Baix Llobregat (I B), Anoia (II B)
Cement	Els Monjos - Constantí	Alt Penedès (II A), Baix Penedès (IV A), Tarragonès (III A)

Table 6.10 Transport distances in each class

	Distance in combined wind speed and population density class [km]											
Purpose	IB	IIA	IIB	IIIA	IVA	IVB	HU	ZZ	SO	GU	M	Sum
Municipal waste				9								9
Slag				16								16
CaO		9		24	17							50
Ammonia				5								5
Scrap treatment				41	12	53	49	206	36	97	40	534
Ash treatment				4								4
Ash disposal	21	32	11	24	17							105
Cement		9		24	17							50

An example of such a spatially differentiated eco-technology matrix is presented in Table 6.11 for the former situation 1. As it can be seen the transport is divided into 11 classes while the MSWI is presented by one class since its site is well known.

Table 6.11: Eco-technology matrix for the former situation 1 [kg/a] (h_{source}=5 m)

	MSWI	**Transport**										
Pollutant	IIIA[e)]	IB	IIA	IIB	IIIA	IVA	IVB	HU	ZZ	SO	GU	M
As	15	0	0	0	0	0	0	0	0	0	0	0
BaP	0	1E-5	2E-5	7E-6	2E-3	4E-5	1E-4	1E-4	5E-4	8E-5	2E-4	9E-5
Cd	15	7E-5	1E-4	4E-5	9E-3	2E-4	6E-4	6E-4	2E-3	4E-4	1E-3	5E-4
PM10	20,411	6	10	3	877	19	58	54	226	39	106	44
Ni	22	0	0	0	0	0	0	0	0	0	0	0
NOx	142,333	101	158	53	13,677	296	906	838	3522	616	1658	684
SO_2	602,643	7	11	4	912	20	60	56	235	41	111	46
O_3 a)	0	14	22	7	1,929	42	128	118	497	87	234	96
O_3 b)	142,333	101	158	53	13,677	296	906	838	3522	616	1658	684
Nitrate[c)]	142,333	101	158	53	13,677	296	906	838	3522	616	1658	684
Sulphate[d)]	602,643	7	11	4	912	20	60	56	235	41	111	46

a) as NMVOC, b) as Nox, c) as NOx, d) as SO_2, e) h_{stack} = 50 m

5) Fate & exposure and consequence analysis

Table 6.12 shows the damage-assigning matrix for DALY (Egalitarian). It has to be noted that this matrix only represent an example of the many possibilities according to the decision table described in section 5.3.2. The matrix is based on the results for the site-dependent impact indicator, expressed as population exposure per mass of emitted pollutant, calculated in section 6.6. Another damage-assigning matrix has also been established with the values of the impact indicator itself expressed with the unit [Persons $\mu g/m^3 * a/kg$]. It has to mentioned that the impact indicators for the heavy metals As, Cd and Ni as well as for benzo[a]pyrene (BaP) are supposed to be the same as for PM2.5. It is assumed that these substances are adsorbed on particles PM2.5 and therefore behave in the same terms of fate and exposure. However, the DALY value is specific for each substance since the dose-response and exposure-response functions are substance-specific as well. The ozone value is taken as country average from Krewitt et al. (2001). The values for nitrate and sulphate do not differ either. This is owing to the fact that these are secondary pollutants for which country averages have been calculated. Only the value for the sulphate in the first row differs slightly. This is due to the stack height of the waste incinerator (50 m stack height), which differs from the one of the transport processes (5 m stack height). For the primary pollutants it can be said that the highest values appear in Madrid and the lowest in Huesca. These values are calculated according to the expression described in section 6.6.4 for the transfer of impact factors to other regions. As Madrid is densely populated and Huesca scarcely populated in comparison to Catalonia, these results are obvious.

Table 6.12: Damage-assigning matrix for DALY (Egalitarian) [a∗a/kg] (h_{source}=5 m)

Class	Pollutant										
	As	BaP	Cd	PM10	Ni	NOx	SO_2	O_3 a)	O3 b)	Nit. c)	Sut. d)
***III A* e)**	5.4E-4	3.2E-2	6.5E-4	1.7E-4	1.2E-4	1.8E-6	6.1E-6	1.0E-6	4.1E-7	4.5E-5	3.7E-5
I B	1.5E-3	8.9E-2	1.8E-3	6.4E-4	3.4E-4	8.3E-6	2.1E-5	1.0E-6	4.1E-7	4.5E-5	3.6E-5
II A	1.3E-3	7.7E-2	1.6E-3	5.4E-4	3.0E-4	5.9E-6	1.8E-5	1.0E-6	4.1E-7	4.5E-5	3.6E-5
II B	9.1E-4	5.3E-2	1.1E-3	3.4E-4	2.1E-4	4.0E-6	1.2E-5	1.0E-6	4.1E-7	4.5E-5	3.6E-5
III A	7.9E-4	4.6E-2	9.5E-4	2.9E-4	1.8E-4	3.1E-6	1.0E-5	1.0E-6	4.1E-7	4.5E-5	3.6E-5
IV A	6.6E-4	3.8E-2	7.9E-4	2.2E-4	1.5E-4	2.3E-6	8.0E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
IV B	4.9E-4	2.8E-2	5.9E-4	1.4E-4	1.1E-4	1.6E-6	5.4E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
HU	2.6E-4	1.5E-2	3.1E-4	3.4E-5	5.8E-5	4.4E-7	1.7E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
ZZ	3.9E-4	2.2E-2	4.6E-4	9.6E-5	8.7E-5	1.2E-6	3.7E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
SO	2.4E-4	1.4E-2	2.9E-4	2.7E-5	5.5E-5	3.6E-7	1.5E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
GU	2.6E-4	1.5E-2	3.1E-4	3.4E-5	5.8E-5	4.4E-7	1.7E-6	1.0E-6	4.1E-7	4.5E-5	3.6E-5
M	2.6E-3	1.5E-1	3.1E-3	1.1E-3	5.8E-4	1.3E-5	3.8E-5	1.0E-6	4.1E-7	4.5E-5	3.6E-5

a) as NMVOC, b) as Nox, c) as NOx, d) as SO_2, e) h_{stack} = 50 m

6) Damage profile

Table 6.13 shows the damage matrix for the former situation 1 resulting from the multiplication of the eco-technology matrix (Table 6.11) and the damage-assigning matrix (Table 6.12). If the first column is considered, the first box (the first element of the diagonal) represents the damage for the waste incinerator. The second element in the second column represents the damage for the transport in district class I B and so on. If one leaves the diagonal, the impact of the processes at (fictitious) other locations is shown. For example, the last element of the first column shows the damage of the waste incinerator if it were located in Madrid. Of course, it has to be admitted, however, that the impact indicator applied to this cell refers to a stack height of 5 m (transport) rather than to 50 m (waste incinerator). The damage would be twice as high as it is currently. Nevertheless, this shows impressively the importance of spatial differentiation.

Table 6.13: Damage matrix with DALY (Egalitarian) for the former situation 1 [a∗a/kg] (h_{source}=5 m)

	MSWI	Transport										
III A[a)]	36.12	0.01	0.01	0.00	0.83	0.02	0.06	0.05	0.21	0.04	0.10	0.04
I B	55.17	0.01	0.02	0.01	1.34	0.03	0.09	0.08	0.35	0.06	0.16	0.07
II A	50.97	0.01	0.01	0.00	1.22	0.03	0.08	0.07	0.31	0.05	0.15	0.06
II B	42.73	0.01	0.01	0.00	1.01	0.02	0.07	0.06	0.26	0.05	0.12	0.05
III A	40.54	0.01	0.01	0.00	0.96	0.02	0.06	0.06	0.25	0.04	0.12	0.05
IV A	37.77	0.01	0.01	0.00	0.88	0.02	0.06	0.05	0.23	0.04	0.11	0.04
IV B	34.46	0.01	0.01	0.00	0.80	0.02	0.05	0.05	0.21	0.04	0.10	0.04
HU	29.81	0.01	0.01	0.00	0.69	0.01	0.05	0.04	0.18	0.03	0.08	0.03
ZZ	32.38	0.01	0.01	0.00	0.75	0.02	0.05	0.05	0.19	0.03	0.09	0.04
SO	29.53	0.00	0.01	0.00	0.68	0.01	0.05	0.04	0.18	0.03	0.08	0.03
GU	29.81	0.01	0.01	0.00	0.69	0.01	0.05	0.04	0.18	0.03	0.08	0.03
M	76.05	0.01	0.02	0.01	1.87	0.04	0.12	0.11	0.48	0.08	0.23	0.09

a) this row refers to h_{stack}= 50 m

Next, the diagonal elements of damage matrix are added to obtain the damage profile, then the part corresponding to transport is compared with the value of the waste incinerator. Like this the damages due to the waste incinerator and the transport can be compared.

The most obvious result of this environmental damage estimation study for industrial process chains is that the contribution of transport to the overall damage increases significantly from alternative 1 over 2 to 3. On the one hand, this is due to the sharp decrease of the overall damage. On the other hand, the decrease of damage due to the improvement of flue gas treatment is partly compensated by the additional transport processes.

From the results of all situations it can be seen that the ratio between the damage due to transport and the overall damage is in the same magnitude for all chosen indicators in this study (PE and DALY). Whether the contribution of transport to the overall result is more significant for the population exposure or DALY in the case of each pollutant depends on the relationship between the toxicity and the mass of pollutants emitted. The more toxic a substance is, the higher is the increase in DALY.

From the results using the end-point indicators derived in this study on the one hand and the HTP on the other hand, it can be said that it seems that the HTP concept underestimates the environmental importance of transport. While the share of transport in the first case is 2.1%/ 1.7% for HTP (CML/ EDIP), it is between 3.2% and 4.2% for the end-point indicators derived in this study. While in case 2 the share of transport for the end-point indicators reaches between 17.1% and 19.6%, the share for HTP is still quite small (12.0%/ 5.8%). However, the CML approach is closer to the results obtained by the endpoint approach than the EDIP approach. The gap widens even more in case 3 (35.5%/ 20.7%) for HTP and 44.1% to 49.2% for the end-point indicators, respectively. The reason for the differences between the results for the HTP and the end-point indicators are clear. The HTP concept considers the fate of the substances, but does not include exposure information. As the environmental impact of transport is very much dependent on the location where it takes place, the deviation from the

HTP results is obvious. The results using the impact indicators of this study, however, show the limits of the ecological benefits of a further technical improvement of the flue gas cleaning. Scenario 3 indicates still a clear overall reduction. However, it shows also that nearly half of the overall environmental impact is due to transport. Therefore, further technical improvement at the waste incinerator shall only be carried out if there is no significant increase in transport, which would worsen the overall environmental efficiency of the process chain.

It has been found that the HTP concept underestimates transport in this case study and, hence, does not very well identify the differences in the environmental impact for the two different processes considered. Figure 6.8 shows the results for all three cases, differentiated in waste incinerator and transport, for the population exposure, DALY (Egalitarian) and HTP (EDIP).

It has to be stated that the HTP indicators may be misleading in the comparison of the absolute environmental burden as well. For instance, if significant reductions only happen in regions with a low population density and high wind speeds (the factors which account for a low population exposure), the reduction for the end-point indicators derived in this study will be rather small while the HTP indicators will identify significant reductions. It can therefore be concluded that the use of end-point indicators as derived in this study is beneficial with respect to a gain of information for both purposes, the comparison of different situations and the comparison of different processes within one situation.

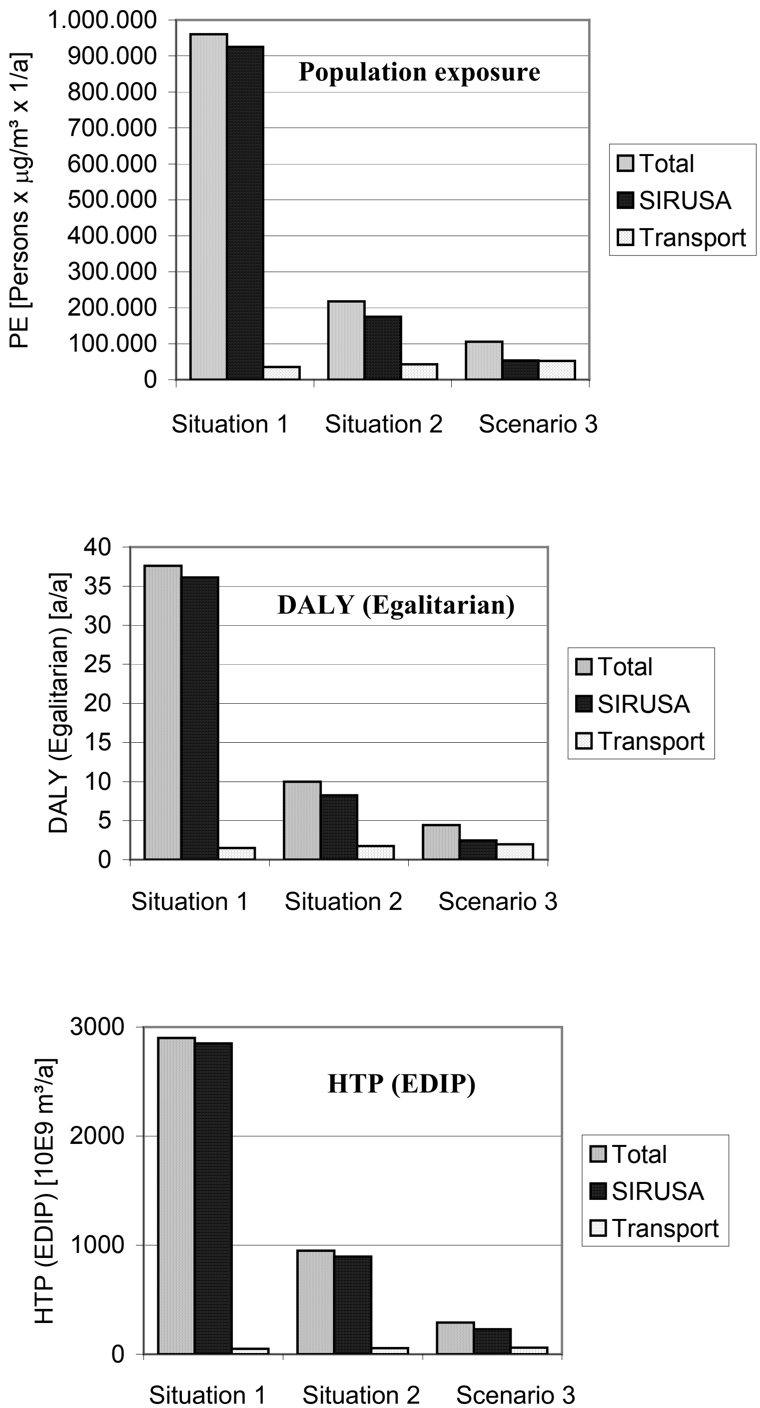


Figure 6.8: Results of the case study for all three situations/ scenarios for the population exposure PE, DALY (Egalitarian) and HTP (EDIP), where SIRUSA stands for the MSWI process

6.9 Uncertainties in the methodology including site-dependent impact assessment

Essential for the validity of the presented application of site-dependent impact assessment is the question whether the uncertainties introduced are justified by the gain of information in comparison to the traditional impact potential used in LCIA. In this context, it must be highlighted that the methodology described in this chapter has to be seen as a balance between the uncertainties introduced on the one hand and the handiness and feasibility of the applied method. In general it can be said that the larger the amount of data available the lower the uncertainty behind the damage estimates.

6.9.1 Goal and scope definition and inventory analysis

In the goal and scope definition subjective elements exit in the form of the selection of the functional unit, what implies if or if not benefits are given for avoided environmental loads. Moreover, the selection of the cut-off criteria for the system boundaries and the dominance analysis can influence the outcome. It is only possible to address these uncertainties and influences by the sensitivity analysis and the use of scenarios for the different options.

In the inventory analysis, the main types of uncertainties and variability that have been analysed by Monte Carlo simulation in chapter 4 are those referring to parameters, including frequency of sampling, method of measurements (continuous on-line, or from time to time by more or less sophisticated analytical methods) and homogeneity of fuels. For the exactness of the modular process model, as all process simulations, the adequate estimation of not well-established physical properties is crucial. The estimations are taken from the publication of Kremer et al. (1998). So the quality depends on the values proposed by these authors.

6.9.2 Dominance analysis and spatial differentiation

The dominance analysis adds a part of uncertainty by the reduction of the media, environmental loads and processes considered. However, it improves the relevance of the remaining information for further assessment.

With respect to the spatial differentiation it is has to be said that the area under study Catalonia is rather small (in comparison to other administration units like Spain or Europe). Therefore, a statistical reasoning is per se limited due to the restricted area. Therefore, the determination of class limits with respect to the administrational units, to the settlement structure and according to meteorological conditions has to be done especially carefully.

The problem with choosing the outer boundaries of the classes (in this case provinces and districts) is the fact that information on dispersion conditions (wind speed) as well as on the settlement structure (population density) is not always bound to administrational units. Especially meteorological conditions are strongly subject to the geographic situation such as topography or latitude, while administrational units are generally linked to the settlement structures. For instance, the district Barcelonès comprises the city of Barcelona with few other municipalities and represents an urban district in an agglomerated region.

However, not all district boundaries delimit that clearly the settlement structure. For example, the Tarragonès district (central city in rural region) comprises the city of Tarragona as well as rural municipalities, which is contradicting in terms of the definition of each class. Administrational units also depend on history and political decisions, because the determination of the limits of the municipalities usually is not a recent decision but ranging in the past. From this reasoning it can be seen that the choice of administrational units is problematic in terms of the representativity of settlement and dispersion conditions. However, these limits are chosen because data are usually available for these administrational units; population data are available for municipalities, districts and provinces.

6.9.3 FATE & EXPOSURE ASSESSMENT

The comparison of the parameter uncertainty and variability involved in the life cycle inventory and the impact pathway analysis (see chapter 4) shows that the uncertainties in the fate & exposure assessment as wells as in the consequence/ effect analysis, and hence also for ERA, are more important due to the fate models, the dose-response and exposure-response functions and the weighting schemes used.

A major source of uncertainties are the fate models used in this study that include dispersion software, multi-media models and long-range transport programmes (as included in integrated impact assessment models).

The program BEEST as well as EcoSense were primarily developed for the calculation of concentration increments due to power plant emissions. This work, however, applies these programmes also for lower stack heights and volume flows (in the case of BEEST), in order to make the method applicable to a wider range of industrial process chains, in this case especially to include transport processes, several assumptions are needed. Unfortunately, the uncertainties cannot be quantified.

In general, the uncertainties related to Gaussian dispersion models are important. They are rather simple descriptions of quite complicated natural processes. Substance data influence the dispersion of pollutants. Therefore, uncertainties in these data are directly introduced into the results of the damages estimations. Another major source of uncertainties related with the fate models used is the choice of the modelling area and the grid size of the dispersion programmes. In the long-range transport programme WTM the overall modelling area is confined to Europe (Eurogrid as implemented in EcoSense), a contribution to the population exposure coming from outside is neglected. An outside contribution is related to North Africa, which lies close to Spain, but which is not included in the Eurogrid. The outside contribution cannot be quantified, however. As the long-range exposure does not depend very much on local variations, the resolution of 100 $*$ 100 kilometres seems to be appropriate for the use of the Gaussian dispersion models, and subsequently UNIRISK. The grid for BEEST calculations is chosen in a way so that the very proximity of the stack (where the concentration increment is very sensitive to the distance from the stack) is described quite well without increasing the uncertainties for greater distances from the stack up to 100 km.

In the case of the site-dependent impact assessment study, with respect to the BEEST programme it has been seen that the outcome for I_{near} is highly sensitive to the volume flow and the mass flow chosen. While EcoSense allows the introduction of any mass or volume flow and still leads to the same results for I_{far}, for BEEST calculations the volume flow and the mass flow have to be determined carefully. The volume flow is derived as a function of the stack height from the evaluation of several industrial processes. The mass flow is derived from the volume flow and the thresholds for emissions of waste incinerators that apply in Catalonia. It has to be stated, though, that the assumption for the volume flow only bases on a small number of industrial processes and that the used threshold is a political value.

As done by Nigge (2000), the calculation of the effective emission height $h_{eff,}$ which includes the physical stack height as well as information about stack temperature and volume flow, may resolve these problems of uncertainties. For this reason, h_{eff} is provided in this study as well. If the impact indicators are given for h_{eff}, the results may be applicable to all kinds of industries, under the condition that the stack height, the volume flow and the temperature are known. Moreover, it has to be stated that there is large variety of algorithms to calculate the effective emission height for different purposes. Generally speaking, it seems to be difficult to find one procedure of resolving these problems of defining source characteristics. This requires further research.

One other important source of uncertainties for the numerical results calculated of the site-dependent human health impact factor for Catalonia is the fact that the models ISCST-3 in BEEST (short-range) and the WTM in EcoSense (long-range) may be in poor accordance in terms of dispersion. As stated above, the dispersion results for BEEST are highly sensitive to the source characteristics. Moreover, the source characteristics have to be chosen in a way, so that the results at the outer boundary comply with the results calculated by EcoSense, the modelling area of which begins there. This is especially difficult because the EcoSense program calculates concentration increments for the grid cells as a whole, i.e. the actual concentration at the modelling boundary with the BEEST programme cannot be determined. This means, if the source characteristics for BEEST are chosen "wrong", the overall outcome of I_{total} would be erroneous because the EcoSense model is not adapted to the source characteristics introduced in BEEST.

A related problem is the fact that higher wind speeds usually lead to lower values of I_{near}. One may suppose that at least parts of the substances are therefore going into long-range transport covered by the Eurogrid of EcoSense. Thus, I_{far} should increases, but EcoSense is not able to take this problem into account, because it provides a coherent set of meteorological data for every grid cell in Europe and is usually not applied in combination with other models. This leads to lower values of I_{total} for higher wind speeds. The only reason to accept this, is the fact that one may suppose that the population density outside of Catalonia is smaller, for which reason an enhanced long-range transport due to higher wind speed leads to lower values of I_{total} because the increase in long-range exposure is smaller than the decrease in the short-range exposure. However, this is just a theoretical reasoning, because no change in I_{far} can be observed.

The same problem applies to the stack height. While the stack height has a great influence on the outcome for the short-range modelling (because it is assumed that the higher the stack, the more pollutants go into the long-range transport), the results for the long-range modelling are quite insensitive to the stack height. It can be said that these problems related to the wind speed and the stack height lead to greater uncertainties the longer the atmospheric residence time of the pollutant is. A pollutant with a short atmospheric residence time is deposited and decays mostly in the short-range modelling area while a pollutant with a longer atmospheric residence also accounts for a significant long-range exposure and is therefore subject to greater uncertainties with respect to wind speed and stack height.

Moreover, the local population exposure subtracted from the population exposure in the EcoSense area cannot be compared to the results of I_{near}, because I_{near} is much more differentiated with respect to meteorological conditions and population density than the EcoSense results for each grid cell. Therefore, compliance between BEEST and EcoSense is not achievable by saying that the population exposure subtracted by EcoSense must be equal to the I_{near} results.

In multimedia models as UNIRISK for ERA additional sources of uncertainty have to be considered. These are especially site characteristics and transfer factors for intermedia transport. The site characteristics include human life styles as different diets and media composition what shows important variations from one site to another.

The evaluation of the dispersion conditions of pollutants in air has been carried out using meteorological data of a limited number of years (site-specific assessment) and measurement stations (site-dependent assessment). The meteorological conditions are strongly determined locally and therefore a reduction of data always increases the uncertainties. And a smaller number of year and stations makes the results less representative for the site or Catalonia as a whole. However, the derivation of the meteorological data files in this study has been carried out on this basis because more data were not available. Of course, the statistical evidence would increase if more data were available. However, for pragmatic reasons, the limited number of data was accepted.

If one considers the derivation of the statistical meteorological data files by Harthan (2001) mentioned in section 6.4.1, it has to be said that the formation of classes of wind speed (0 - 2 m/s, 2 - 3 m/s and 3 - 4 m/s) has not been undertaken according to a statistical reasoning. The limits of these classes are chosen according to the limits as defined by the Deutscher Wetterdienst (DWD). These class limits constitute a good compromise between the concept of classes itself and the meaningfulness of the classes. If the class limits were chosen broader, this would increase the handiness of the results because the overall number of classes would be reduced. However, this would lead to a large variation of actually occurring wind speeds within each class, i.e. the statistical determination of the classes, describing the wind speed of all locations lying in this class with a reasonable standard deviation, would no longer be well-founded. If the class limits were chosen narrower, this would lead to a smaller standard deviation within each class and would therefore decrease the uncertainties. Nevertheless, the number of classes would increase and the number of districts lying in each class would decrease. A statistical reasoning combining several districts in one class would no longer be possible. The class limits chosen here therefore seem to be appropriate, because they allow a minimum differentiation of wind speed (into three classes), but still with a reasonable number of districts per class.

The neglect of terrain elevations and precipitation is necessary in the site-dependent impact assessment due to the absence of a statistical reasoning for this parameter. However, this leads to uncertainties. Especially the concentration increment of particles calculated is overestimated, since wet deposition is not considered. An evaluation of different temperatures has led to the conclusion that the results for the concentration increment are not sensitive to the temperature. Therefore, it is valid to choose the country average for all data sets and throughout the year. The neglect of wind direction leads to uncertainties especially on a local level, because the wind direction strongly influences the dispersion on that level. However, in order to form class averages accounting for several wind directions, the neglect of wind directions is assumed to be valid. The height of the mixing layer is calculated according to VDI 3782/1, which is a German guideline on dispersion modelling. The uncertainties introduced are therefore considered to be small. The surface roughness is chosen as one value for the whole of Catalonia as a rural value, which is assumed to be a good estimate for the general settlement structure of Catalonia.

Classes according to the population density are formed. It is argued that the statistical basis is good enough to calculate the radial population density and that it is valid to use an interval of 10 km for the annuli considered, since the biggest municipality in Catalonia has a smaller surface. However, it is stated that reducing the resolution outside of Catalonia to the district level increases the uncertainties. It is assumed, though, that this is a reasonable procedure, as this only concerns a limited number of adjacent districts to Catalonia and because, with respect to working load, this is a feasible way.

6.9.4 CONSEQUENCE/ EFFECT ANALYSIS

One of the most important sources of uncertainties relates to the dose-/ exposure-response functions of pollutants further described in Chapter 2. These functions determine the consequence or effect analysis. Uncertainties due to these functions therefore directly apply to the end-point-related indicators or damages estimates (physical impacts like cancer cases as well as YLD, YOLL, DALY and external environmental costs). If someone wants to avoid these uncertainties, the impact indicators themselves can be applied as "pressure on human health". However, in order to take the differences in the toxicity of the pollutants and its sensitivity to human health into account dose-/ exposure-response have to be considered. For instance Table 2.2 offers a variety of dose-response functions, which can be chosen according to the value preferences of the user and which show huge relative differences. More functions can be obtained from other public health or environmental authorities.

YLD, YOLL, DALY and external environmental costs are determined by subjective judgement that directly influences the outcome. In order to reduce these uncertainties and the subjective influence, this work offers several options. For uncertainties about YLD, YOLL and DALY values for several pollutants see Hofstetter (1998). For uncertainties in the evaluation of external environmental costs see EC (2000).

Part D:

Overall discussion and conclusions

7 OVERALL DISCUSSION

7.1 INTRODUCTION

Finding an adequate trade-off between process-chain-orientated and site-orientated environmental impact assessment and expressing environmental damages estimates in meaningful results like environmental costs has been the aim of this thesis. The subsequent topics are related to the interfaces and potentials for integration of existing environmental management tools, the uncertainties behind the impact assessment methods and the creation of a common framework for midpoint and endpoint approaches. Corresponding to this aim and the related topics, the discussion is divided into 1) the evaluation of the interfaces and potentials for integration in the commonly used methods, 2) the assessment of uncertainties, 3) the methodology of environmental damage estimations for industrial processes and 4) the site-dependent impact assessment method. Trade-offs between process chain-orientated and site-orientated environmental impact assessments are offered that allow to express environmental damages estimates in meaningful results like environmental costs. The methodology of environmental damage estimations for industrial processes establishes a common framework for midpoint and endpoint approaches. Moreover, the results of the case study on waste incineration with a special focus on human health are highlighted.

In the previous chapters a methodological framework for environmental damage estimations in industrial process chains has been developed. This framework is a theoretical construct, a combination of existing and new methods, somehow a hyper-model, that has been applied in this work only in two cases. The methodology development and its applications are based on the current literature; some empirical knowledge has been added. Hence, it is sensible to ask which elements have been added to the knowledge in the respective field of science through this work. In order to make this clear we indicate in the discussions what is considered to be new findings or lessons learned.

7.2 Evaluation of interfaces and potentials for integration in the commonly used methods

The question of integration and interfaces between different methods of the environmental management toolbox can be considered as an issue of research. The EU funded OMNIITOX project (Pant et al., 2001) focuses on the relation of ERA to LCIA and the LCM program of the UNEP/ SETAC Life Cycle Initiative is addressing this issue for a life-cycle economy (Sonnemann et al., 2001). The evaluation of the environmental toolbox shows that the tool with which an environmental problem is studied has to be carefully selected. The outcome of such study is in somehow predominated by the analytical environmental management tool used. A guide prepared for this work indicates which tool or combinations of tools should be applied to which type of application that requires a life cycle perspective.

The parallel application of chain- and site-orientated impact assessment methods has shown that the bulk of data needed is the same. Hence, it should be explored in how far it is possible to enhance the integration of methods in even more powerful integrated assessment models.

Looking at the presented picture of chain- and site-orientated environmental impact assessment methods with regard to fate and exposure, the engineering part of the assessment procedure, it is clear that both fate and exposure are not sufficiently reflected in current LCIA methods. To overcome the inconsistency between impact potentials and actual impacts it is proposed to differentiate the life cycles also according to their number of processes considered. That means to use different levels of sophistication for different applications that here are defined by their chain length. Further work is needed on the differentiation of life cycle methods for different types of life cycles according to their length, i.e. number of processes involved.

New findings in the evaluation and by the application of the commonly used methods employed in this study are:

- The application-dependency of LCA (Wenzel, 1998) was extended to an application dependency of the environmental management toolbox as presented in CHAINET (1998).
- LCIA has been applied so far in a uniform way for all life cycles, however there is a significant difference in the type of a life cycle under study:
  - Complex product life cycles need other type of environmental impact assessment than industrial process chains, here defined as life cycles with less than 100 processes.
  - While complex product life cycles are best assessed by generic methods (if available with best practice as proposed by the Life Cycle Initiative of UNEP/ SETAC (2001)), industrial process chains allow the use of different levels of detail in the environmental impact assessment.

7.3 Uncertainty assessment

By the assessment of uncertainties in Life Cycle Inventories and the Impact Pathway Analysis using the stochastic model Monte Carlo simulation, after filtering the data according to their availability, results are transformed from a concrete values into probability distributions around mean values. A full quantitative uncertainty assessment is so far time consuming to be applied to environmental damage estimations in industrial process chains. In particular, the determination of the probability distribution needs a lot of efforts. For the future we hope that more studies will be carried out in order to amplify the experience among the LCA and IPA practitioners on uncertainty assessment by Monte Carlo simulation. In this way a basis of practical hints and probability distributions for different parameters and technologies could be created that would facilitate the use of this technical element and increase its consistency from one study to another. The realisation of the Monte Carlo simulations is not demanding so much time due to the use of a special software package.

The uncertainty and variability calculated using Monte Carlo Simulation is less than obtained in a study with analytical methods carried out by Rabl & Spadaro (1999). The presented results give a geometric standard deviation of less than 3, whereas by analytical methods the geometric standard deviation is bigger than 4.

The IPA study was carried out on a local scale for which dispersion models seem to be more reliable. Now the developed framework should also be applied to analyse the uncertainties in IPA studies carried out on a regional or continental scale. Moreover, in future research also the uncertainties of the damage estimations for other sensitive receptors, as crops, materials and ecosystems, should be assessed in this way.

The following point can be considered to be a lesson learned:

- By the parallel assessment of the uncertainties in the IPA and the LCI with the same approach it could be shown that the uncertainties are higher in the IPA (with an average geometric standard deviation σ_g of 2.7) than in the LCI (with a σ_g between 1.1 and 2.3 depending on the pollutant). In particular this is due to the fact that IPA introduces the effect analysis that contains the parts with the highest uncertainties (σ_g of 3 for dose-response functions for cancer, σ_g of 4 for YOLL and σ_g of 2 for monetary valuation), what corresponds to the parts that belong to the valueshere according to Hofstetter (1998).

7.4 Methodology of environmental damage estimations for industrial process chains

The established methodology of the environmental damage estimations for an industrial process chain integrates different concepts, methods and models of the environmental management toolbox with the life cycle approach. The methodology is the central part of the thesis where most others chapters are related to. The methodology can be characterised by its two working directions and the accompanying steps:

1. Working directions:
   a) Distribution of the environmental load along the different life cycle stages.
   b) Assignation of the damages in the corresponding region of emission.
2. Dominance analysis to identify relevant components (processes, pollutants and media).
3. Different levels in fate and exposure analysis for a compromise between practicability and accuracy.
4. Choice by the decision-maker of weighting and aggregation scheme.
5. Optional calculation of the eco-efficiency and consideration of accidents.

The algorithm developed in order to consider the damage function permits to manage in an elegant way both directions. Based on the distribution of the environmental loads among the processes, eco-technology matrixes are constructed. Taking into account results for the environmental damage cost and/ or other damage indicators of each process in its region, damage-assigning matrixes are determined. The multiplication of both matrixes yields a result with information on the environmental performance of the functional unit considered in the industrial process chain.

By the utilisation of the methodology for the case study its applicability is shown and the possibility to improve decision-making in certain applications is demonstrated. The framework is established but the first application is quite restricted. Work has to go on for a more sophisticated version of the case study that really includes multi-media fate modelling and more site-specific data, in particular for the ecological damage parameter. Moreover, the overall results should be analysed concerning their uncertainties and sensitivity by Monte Carlo simulation. In general, a detailed numeric assessment of the uncertainties in the different steps of the methodology compared to midpoint indicators is an obvious topic for further work that has to be accompanied by several case studies. Such applications are also necessary to allow a further evaluation of the presented methodological framework in practice. The integration of the algorithm into commonly used software tools could quickly enhance the number of case studies.

The main limits of the proposed methodology of environmental damage estimations for industrial process chains and the related site-dependent impact assessment approach are related to the fact that, at least currently, the method seem only to be applicable to so-called industrial process chains, here defined as process chains with 100 or less industrial processes, but not to complete life cycles with a much higher number of processes involved. In the case that the process chain is not very long, the framework works well, as demonstrated in this study. This application is especially facilitated if lots of processes are situated in the same region, which is for instance often the case in an end of life chain, a so-called gate-to-cradle LCA. The presented methodological framework therefore has to be considered as a complementary tool among others that is able to bridge the gap between tools (such as between LCA and ERA).

As main reasons for the mentioned limitation the insufficient availability of data and the high workload to carry out a study for more complex life cycles are considered to be crucial. At the moment site-specific data such as release location (or type of generic class), stack height, volume flow and emission temperature are insufficiently available. This is due to the fact that first the life cycle inventory is generated without looking at the requirements of an adequate impact assessment. In general, the current LCA software supports this fact by not allowing any type of spatial differentiated LCIA (Potting, 2001).

Another limitation consists in the limited number of existing published data in the literature on site-specific and site-dependent damage evaluations at different levels of detail for several endpoints and regions or generic classes. Site-dependent impact factors have been published for a short period of time. Main examples are: Krewitt et al. (2001), Huijbregts & Seppälä (2000), Moriguchi & Terazono (2000), Nigge (2000), Potting (2000), Spadaro & Rabl (1999). Of course it is necessary to approximate these data for the industrial process chain under study. Future research is required to produce a variety of such damage estimates. Since different stakeholders work with diverse weighting schemes, it has to be accepted that, moreover, a lot of data will be available in incompatible damage estimation formats, unless an international agreement is reached with regard to this, what not seems to be very probable at the moment.

The damage estimation phase needs much more efforts than the LCIA with midpoint potential. This is especially true if the data for spatial differentiation, at different levels of detail, are not available. Then the collection and/ or production of this data can easily generate a workload that is much more than the life cycle inventory itself. Since LCA has been criticised by being too expensive and data-intensive (UNEP, 1999) it is not very probably that decision-makers are willed to invest even more resources in the damage estimations for complex life cycles, until a practical and feasible software tool can be developed in the direction of this approach.

As a final point, it can be stated that the realisation of an environmental evaluation for a functional unit according to the site-specific strategy, outlined in chapter 5, is certainly the best way to obtain accurate results about the environmental damage caused by products and services, but it does not seem a very practical approach. However, it can be expected that on a long perspective the impracticability cannot be an argument against it, as decision-makers prefer exact results if they can obtain them. When the first LCA studies were carried out people would not have thought that nowadays it is so much easier to do it due to the high amount of available LCA data. Moreover, it should be stressed that the existence of a theoretical structure often stimulates the collection of data in practice (Heijungs, 1997). For instance, damage estimations for each process in its corresponding surrounding could be made available on the web, if necessary in a confidential form only for the clients, and then the fabricant of an intermediate or final product could sum up the damage estimates of the different processes involved in its production and make them available to its customer, also on the web or in the form of an added information paper or even a barcode. This would be quite similar to the idea behind the environmental product declarations according to ISO/TR 14025 (2000).

Trade-offs between practicability and accuracy are also imaginable nowadays that might allow to enhance the application to process chains with more than 100 industrial processes. In the case of a more complex industrial process chain, the environmentally most relevant processes in the life cycle could be identified by screening in a dominance analysis (Heijungs et al., 1992) after a LCA. The few selected processes would have to be analysed more in detail. In the same way it would be worth examining how to reduce the number of environmental laods for any further assessment. In such an application the main task would be to carry out very carefully the dominance analysis since then most of the environmental relevance is determined by this step.

Furthermore, it has to be stated that the special focus of the present study is on effects to human health. Hence, it would be necessary to see in how far the methodology works for other types of environmental effects. However, in principle, no methodological problems should occur with regard to other impacts caused by emissions to the environments. In opposite, the questions of input-related impact categories, especially the evidently site-orientated land use aspects, are not well reflected in this report and hence their relation to environmental damages estimations continue to be a subject for further research, while resource consumption can be considered as global damage indicators, similar to GWP if a decision-maker wants to take it into account.

Besides, the established framework permits several other types of further development. The developed matrix algorithm can be applied in a similar way to identify the principal emission points in a region. A first illustration on how to do so has still to be elaborated. Another way of enhancing the presented methodology is by considering production and internal environmental costs in a more comprehensive manner, similar to the Total Cost Assessment approach by Rogers (2001) that is in somehow a response to the full-cost accounting suggested by Popoff & Buzzelli (1993).

With regard to the methodology the following findings are considered as crucial:

- The mathematical framework for site-orientated assessments in a life cycle perspective allows environmental damage estimations for industrial process chains. It can be characterised by the distribution of the environmental load along the different life cycle stages in the eco-technology matrix and the assignation of the damages in the corresponding region of emission in the damage-assigning matrix. The flowchart for the different steps of the methodology can be characterised by the responsibility of the decision-maker for the weighting, the dominance analysis for filtering the most relevant elements and its transparency with regard to different levels in fate and exposure analysis for a compromise between practicability and accuracy.
- The methodology is of interest for the public administration, financial entities, companies and consumers as well as the general community for a variety of applications proposed in this study. The applications can be characterised in general by their combination of site- and chain-orientated environmental problems.

- The methodology puts midpoint and endpoint approaches in a common framework. That allows to get an idea of the differences in the results of both approaches and to make the uncertainties behind them more transparent.

Strategy for generic applications

Since it is evident that a lot of environmental problems related to a process chain-perspective cannot be studied by the developed framework, at least currently, due to insufficient data availability and infeasible workload, a strategy for process chain applications of a generic type is briefly discussed:

Products and services with more than 100 processes involved, in general, refer to economic activities that form part of the so-called globalised economy. Products are traded worldwide and therefore problems related to life cycles of product systems have to be resolved on a global level. International cooperation is necessary to establish a consistent and encompassing framework and information system (Sonnemann, 2001; UNEP/ SETAC, 2001).

The focus in the area of Life Cycle Impact Assessment should be on best practice with regard to the characterisation of emissions, resources extractions and land use, that means on the aggregation by adequate factors of different types of substances in a selected number of environmental issues, or impact categories such as resource depletion, climate change, acidification or human toxicity. It includes the following tasks, which should be developed in order of increasing sophistication and with the option for generic application dependency:

1. Consistent and encompassing framework of environmental processes and Areas of Protections enabling the choice of category indicators at different midpoint levels (like GWP for climate change) and at endpoint level (like forecast indicators for impacts on human health in term of Years Of Life Lost).
2. Identification of a default list of impact categories and correspondent category indicators, first at global and then also at regional level, possibly consisting of two sets, one at midpoint and one at endpoint level.
3. Modelling, determination and verification of adequate factors for characterisation within each category, to provide the best available approach with regular updating.
4. Analysis of the relationship between characterisation, normalisation and weighting methods and drafting of guidelines in order to make the LCIA outcomes more decision-friendly.

According to Jolliet & Pennington (2001), establishing best practice for LCIA means an interesting challenge: bringing together scientific pragmatism, to obtain characterisation factors and data sets that are scientifically defendable, in a practical form that is relevant to the decision endpoints. Starting with the transfer from diverse projects from different parts of the world (e.g. the Japanese National LCA project or Eco-indicator 99) and from the results of the Second SETAC-Europe Working Group on Life Cycle Impact Assessment (Udo de Haes et al., 2002), it should be defined to what extent existing approaches and databases can be adopted to meet the requirements of these deliverables and where there are gaps requiring further research. This procedure should be carried out in a very open process involving worldwide stakeholders from industry, academia, governments and NGOs. Locating, collecting and sharing of life cycle inventory data should be accompanying measures to the work on LCIA.

7.5 SITE-DEPENDENT IMPACT ASSESSMENT METHOD

The site-dependent impact assessment method developed by Nigge (2000) has been adapted to fit perfectly into the methodology of environmental damage estimations for industrial process chains. Spatial differentiation in LCIA has shown to be feasible with a reasonable effort and little additional information required in the LCI (district name, stack height, information about particle size distribution). The particle size distribution has been identified to be a very important parameter for the fate and exposure modelling as well as the effect analysis on human health. Therefore, apart from spatial differentiation, a differentiation of particle sizes in LCI is desirable. The site-dependent impact assessment method uses components from ERA as well as LCA. Hence, it can be said that models usually used for environmental risk assessment (e.g. dispersion models) can be used for LCIA purposes as well, if the procedure is based on a statistical reasoning.

The site-dependent impact indicators derived for airborne pollutants causing carcinogenic and respiratory health effects in Catalonia express the "pressure on human health" due to the emission of 1 kg of the considered pollutants in each class and are ready for use in other applications. More site-dependent human health impact factors for Catalonia should be calculated in the future, based on the same meteorological and population data prepared, for more pollutants emitted from industrial processes than those considered; this study has been confined only to a handful representative pollutants responsible for respiratory effects and carcinogenesis.

The main restriction for the future use of the calculated site-dependent impact factors is their inherent uncertainty and therefore the question of their reliability. For all the different sources of uncertainties introduced as described in section 6.9 it has to be said that they are especially important for the absolute outcome of environmental damages estimations in industrial process chains, because it depends on the fate, exposure and effect analysis. As absolute results never can comply with reality due to the many assumptions made, the more important question is whether the uncertainties have an effect on the relative results for comparison purposes.

Uncertainties in the fate analysis can affect the relative results, for example when the considered process chains differ in source characteristics (stack height, volume flow, etc.) and these characteristics are subject to significant uncertainties. Uncertainties in the exposure analysis can be significant, if for example the process chains differ in short- and long-range transport (due to different stack heights, for instance). Uncertainties in the effect analysis can affect the relative results, for example if the relative impact of pollutants (if one compares two process chains with different pollutants) depends very much on the uncertainties in substance data, dose-/ exposure-response function etc.

From these considerations, it can be seen that all steps of the development of endpoint indicators can be subject to uncertainties and influence the relative outcome of LCIA. It has to be said, however, that other approaches with no spatial differentiation are subject to uncertainties as well, usually in the effect analysis (weighting of the environmental impact potential of different pollutants). However, the accordance between predicted impact and actual impact for human health effects is often very poor for midpoints. In this context, the uncertainties introduced by deriving the impact indicators in this study are overcompensated by the gain of information by spatial differentiation which has been achieved with reasonable effort and which can be applied to case studies with limited additional information.

To improve the reliability of the site-dependent impact indicators further research is especially required on how to statistically define the source characteristics for several industrial processes. The correlation functions used in this study are only based on few samples and obviously the use of the threshold does not seem to be the best solution to determine the mass flow. An evaluation of the correlation of stack heights, volume flows and mass flows per industry type and country, as provided by Potting (2000) for stack heights in the Dutch industry, would decrease significantly the uncertainties related to the legal threshold and other approximated values applied in this study for that purpose, which may not be valid for other industry branches and countries. Moreover, the temperature of the gas accounts for the buoyancy plume rise. As the temperature was chosen as one value for all runs carried out, there might be great differences to other industries. Therefore, research about stack temperatures in the different industry branches seems to be necessary, too.

The further enhancement of GIS and its integration into ERA and LCIA modelling may further simplify the use of data on receptors and dispersion conditions for the consideration of spatial aspects. In particular, that might support the application with regard to other receptors than population and the inclusion of their statistical distribution. Quite a lot of research, however, will be necessary to explore in how far the method can be applied with reasonable efforts to the fate in other media than air. Schulze (2001) has presented a first work in this field for detergents that reveals the principle feasibility of that approach. The use of GIS in the interface between MFA and LCA for resources has been further explored by Bauer & Zapp (2000). The intention of the Danish EPA to include spatial differentiation in the new version of EDIP (Hauschild & Potting, 2000) is a good sign to promote further work in this direction.

Finally, it should be noted that in line with the uncertainty considerations in section 6.9 the accordance of different pollutant dispersion and transport models at their overlapping area is also definitely a topic for future research. The same holds true for the temporal differentiation of the impact assessment into calendar years (or other type of intervals) by following a similar approach than for spatial differentiation.

With regard to site-dependent impact assessment new achievements are:

- The comprehensive framework for site-dependent impact assessment proposed by Potting (2000) and the fully functional method for human health impacts based on statistically verified classes by Nigge (2000) were integrated in the methodology of environmental damage estimations in industrial process chains and further adapted in this study to the use of a more complicated Gaussain dispersion model and the support by a Geographic Information System.

- The basic idea of the method of Nigge (2000) was generalised for the application with other receptors than population like crops, maintenance surfaces and ecosystems.
- A set of site-dependent impact indicators of Catalonia on a district level was calculated for airborne pollutants responsible for human health effects.

7.6 Case study: Waste incineration with special focus on human health

The chosen case study of the process chain of a municipal solid waste incineration with the special focus on human health effects has proven to be an excellent example for environmental damage estimations in industrial process chains. The study of externalities carried out confirms that the most important contribution to the damages estimated by monetary means stems from the human health effects (99 %); the damages caused to crops and materials are by far less important (1 %).

The results of the LCA show that neither all environmental loads in the inventory analysis nor all impact potentials improve after the installation of the advanced gas treatment system. However, the chosen single index Eco-indicator'95 indicates an overall improvement of 60 %. The comparison with the Spanish electricity mix gives an idea about the positive environmental effects of the waste incineration, which were the result of the consideration of avoided environmental loads if the waste treated as service to society would be chosen as functional unit.

The multi-media model applied in this work (UNIRISK) is not well known internationally. Therefore, it would be good to compare the current results for PCDD/Fs with those obtained by the application of a more recognised one for organic pollutants, which is supposed to be more powerful, but that may be less adapted to the zone under study, e.g. EUSES or CalTox. Then also a correction factor, to include multimedia fate modelling in the damage factor of the IPA, could be calculated based on the results.

The application of the IPA in a generic way allows to calculate the external environmental costs. Considering an average cost of the electricity of 0.06 €/ kWh, it can be seen that approximately 30 % of the benefit obtained by the selling the electricity are external costs, i.e. 0.02 €/ kWh. Such evaluations can create the basis for the determination of ecotaxes.

The analysis of the uncertainties and variations in the Life Cycle Inventory of the waste incineration LCA study by a Monte Carlo Simulation has shown that the variations within the concentrations of the incinerator emissions are in most cases more important than the uncertainties derived from literature for site-specific data and especially for the information taken out of life cycle databases. Hence, it must be questioned if the estimated uncertainties published in the considered literature are sufficient in comparison to the variations in the measurements of environmental loads. Consequently, for good practises in uncertainty assessment it would be recommendable that information on the probability distribution type and the corresponding standard deviation of the considered parameters were added to the databases.

The sensitivity analysis within the Monte Carlo simulation applied to the damage estimations of the MSWI emissions by IPA on a local scale indicates that, using toxicological dose-response and epidemiological exposure-response functions, the main impact on human health does not stem from the PCDD/Fs and the heavy metals, but particles and NOx (together more than 99 %). This can be considered as an important contribution to the public discussion on the human health risks of waste incineration (Greenpeace, 2001). However, it has to be taken into account that no general agreement exists about the reliability of epidemiological studies (Crettaz et al., 2002; Hofstetter, 1998).

The application of the developed methodology of environmental damage estimations for industrial process chains reveals that the environmental external cost estimations give much more weight to the transport processes than the Eco-indicator 95 method. The transport contributes with approximately 25 % to the external environmental costs before the installation of the advanced gas treatment system and with 30 % afterwards, while according to the Eco-indicator 95 the transport adds less than 10 % in the current situation and less than 5 % in the former situation. The comparison of the results derived by the site-dependent endpoint impact indicators for Catalonia with those obtained by the midpoint indicator Human Toxicity Potential (HTP) indicates that the HTP underestimates the environmental impact of the transport processes, too. This is especially relevant for the most advanced gas treatment technologies, i.e. additional with DeNOx, where the improved cleaning capacity is compensated more and more by the additional efforts in raw material supply, including supplementary transport.

The case study has been carried out with a reduced number of environmental loads and a special focus on human health. The reduced number of environmental loads was considered as an acceptable restriction and sufficient to show the principal feasibility of the methodology development, in accordance with the general experience that advances made for one element out of a homogeneous group work in generall for all the others. It would be interesting to increase the number of environmental loads and related impact categories in a continuation of that work. Also the application of the methodology to another MSWI, but especially to other types of waste treatment plants would give the possibility to compare different industrial process chains. That might imply also the consideration of other media than air, the time-differentiation and the simulation of accidents, which for instance in the case of landfills are of crucial importance for the final outcome. At the end, a new generation of integrated waste management tools as Wizard (Ecobilan, 1999), or the more recently presented inventory by McDougall et al. (2001), seems to be feasible that takes into account the setting of the waste treatment installations and the site affected by the transport routes, allowing in this way an overall environmental optimisation. However, to reach this point still a lot of further development is necessary since this work established only the foundation.

The main insights that the case study has facilitated can be summed up in the following way:

- The application of the site-dependent endpoint impact indicators of human health in the case study to compare the MSWI with the related transport processes has shown that the human health impact due to transport is more and more important the stronger the efforts are to reduce the MSWI emissions, and therefore they should be considered in further flue gas cleaning installation to assure an overall impact reduction. To focus only on the MSWI installation as typically done by ERA and IPA seems not to be justified, if a highly advanced gas cleaning facility is installed.
- In a comparison of the results of endpoint-orientated indicators (external costs and DALY in the derived site-dependent impact assessment indicators for Catalonia) with LCIA midpoint indicators (Eco-indicator 95 and the Human Toxicity Potential), it was found that for the situation of the case study apparently the midpoint indicators underestimate the environmental impact of the transport processes. This seems to be reasonable because the midpoint indicator does not consider the clear differences in the impact due to emissions at 5 m stack height in comparison to those at 50 or more meters height of an incinerator.

8 CONCLUSIONS AND OUTLOOK

8.1 CONCLUSIONS

Considering the structure used for the overall discussion in chapter 7 the conclusions may be summarised as follows:

Evaluation of interfaces and potentials for integration in the commonly used methods

- The evaluation of the environmental toolbox shows that the tool with which an environmental problem is studied has to be carefully selected. Guidance is needed since the outcome of such study is in somehow predominated by the analytical environmental management tool used.
- To overcome the inconsistency between impact potentials and actual impacts it is proposed to differentiate the life cycles according to their number of processes included. That means to use different levels of sophistication for different applications that here are defined by their chain length.
  - Complex product life cycles need another type of environmental impact assessment as industrial process chains; here defined as life cycles with less than 100 processes.
  - While complex product life cycles are best assessed by generic methods (if available with best practice as proposed by the Life Cycle Initiative of UNEP/ SETAC), industrial process chains allow the use of different levels of detail in the environmental impact assessment.

Uncertainty assessment

- By the assessment of uncertainties in Life Cycle Inventories and the Impact Pathway Analysis using the stochastic model Monte Carlo simulation, after filtering the data according to their availability, results are transformed from a concrete values into probability distributions around mean values.
- By the parallel assessment of the uncertainty and variability in the IPA and the LCI with the same approach it could be shown that the uncertainties are higher in the IPA (with an average geometric standard deviation σg of 2.7 than in the LCI (with a σg between 1.1

and 2.3 depending on the pollutant). In particular, this is due to the effect analysis that contains the parts with the highest uncertainties (σg of 3 for dose-response functions for cancer, σg of 4 for YOLL and σg of 2 for monetary valuation).

- The uncertainty and variability calculated using Monte Carlo Simulation is lower than that obtained in a study with analytical methods. The presented results give a geometric standard deviation of less than 3, whereas by analytical methods the geometric standard deviation is bigger than 4.

Methodology of environmental damage estimations for industrial process chains

- The developed mathematical framework for site-orientated assessments in a life cycle perspective with the distribution of the environmental load along the different life cycle stages in the eco-technology matrix and the assignation of the damages in the corresponding region of emission in the damage-assigning matrix is a powerful instrument for environmental damage estimations for industrial process chains.
- The flowchart makes the different steps of the presented methodology very transparent for those who want to estimate environmental damages in industrial process chains. It can be characterised by the responsibility of the decision-maker for the weighting, the dominance analysis for filtering the most relevant elements and the different levels in fate and exposure analysis for a compromise between practicability and accuracy.
- The methodology puts midpoint and endpoint approaches in a common framework, allowing to get an idea of the differences in the results of both approaches and to make the uncertainties behind them more transparent.

Site-dependent impact assessment method

- The adapted site-dependent impact assessment method fits perfectly into the established methodology of environmental damage estimations for industrial process chains. It is not only applicable to human health effects, but also to other receptors than population.
- Spatial differentiation in LCIA has shown to be feasible with a reasonable effort and little additional information required in the LCI (district name, stack height, information about particle size distribution).
- Models usually used for environmental risk assessment, e.g. dispersion models, can be used for LCIA purposes as well if the procedure is based on a statistical reasoning.

Case study: Waste incineration with special focus on human health

- The sensitivity analysis within the Monte Carlo simulation applied to the damage estimations of the MSWI emissions by IPA on a local scale indicates that using toxicological dose-response and epidemiological exposure-response functions the main impact on human health does not stem from the PCDD/Fs and the heavy metals, but particles and NOx (together more than 99 %).
- The application of the site-dependent endpoint impact indicators of human heath in the case study has shown that the stronger the efforts are to reduce the MSWI emissions more

important are the human health impacts due to transport, and therefore they should be considered in further flue gas cleaning installation to assure an overall impact reduction. To focus only on the MSWI installation as typically done by ERA and IPA seems not to be justified, if a highly advanced gas cleaning facility is installed.

- In a comparison of the results of endpoint-orientated indicators (external costs and DALY in the derived site-dependent impact assessment indicators for Catalonia) with LCIA midpoint indicators (Eco-indicator 95 and the Human Toxicity Potential) it was found that for the situation of the case study apparently the midpoint indicators underestimate the environmental impact of the transport processes.

8.2 Outlook

The research in the field of environmental science and engineering between environmental risk assessment and life cycle assessment is a young discipline. Currently the huge EU funded OMNIITOX project focuses on the relation of ERA to LCIA. The question of integration and interfaces between different methods of the environmental management toolbox is studied within the LCM program of the UNEP/ SETAC Life Cycle Initiative. Hence, evidently there are still a large number of items left that have to be worked out in future research. This thesis has to be seen as one contribution to the general methodology development going on. In order to structure this outlook the same item groups are used than in the previous section.

Evaluation of interfaces and potentials for integration in the commonly used methods

- Further work in terms of the differentiation of life cycle methods for different types of life cycles according to their length, i.e. the number of processes involved, is necessary.
- Since the parallel application of chain- and site-orientated impact methods has shown that the bulk of data needed is the same, it should be explored in how far it is possible to enhance the integration of methods in even more powerful integrated assessment models.

Uncertainty assessment

- A basis of practical hints and probability distributions for different parameters (in IPA) and technologies (in LCI) should be created in order to facilitate the use of Monte Carlo simulation and to increase its consistency from one study to another.
- The developed framework for stochastic uncertainty assessment in IPA should also be applied to analyse the uncertainties in IPA studies carried out on regional or continental level and for other sensitive receptors, as crops, materials and ecosystems.

Methodology of environmental damage estimations for industrial process chains

- A detailed numeric assessment of the uncertainties in the different steps of the methodology compared to midpoint indicators is an obvious topic for further work; this proceeding should be based on several case studies.

- The developed matrix algorithm can be applied in a similar way to identify the principal emission points in a region. A first illustration on how to do so has still to be elaborated.
- Another way of enhancing the presented methodology is by considering production and internal environmental costs in a more comprehensive manner.
- Future research seems to be fruitful on the question on how to establish an information system, i.e. by internet and/ or using barcodes, to enable the transmission of site-orientated damages estimates from one player in the process chain to another.

Site-dependent impact assessment method

- Research is required on how to statistically define the source characteristics for several industrial processes.
- The accordance of different pollutant dispersion and transport models at their overlapping area is also definitely a topic for future research.
- Further enhancement of geographic information systems and its integration into ERA and LCIA modelling may further simplify the use of data on receptors and dispersion conditions for the consideration of spatial aspects, especially with respect to the statistical distribution of other receptors than population.
- Quite a lot of research will be necessary to explore in how far the method can be applied with reasonable efforts to the fate in other media than air.

Case study: Waste incineration with special focus on human health

- The application of the methodology to other types of waste treatment plants than MSWI would give the possibility to compare different industrial process chains. That might imply also the consideration of other media than air, temporal differentiation and the simulation of accidents that, for instance, in the case of landfills are all of crucial importance for the final outcome.
- At the end, a new generation of integrated waste management tools seems to be feasible that takes into account the setting of the waste treatment installations and the site affected by the transport routes, allowing in this way an overall environmental optimisation. However, to reach this point still a lot of further development is necessary since this work established only the foundation.

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Indexes

I.1 General abbreviations and symbols

Abbreviations

a	year
ACV	Análisi del Cicle de Vida
AETP	Aquatic Eco-Toxicity Potential
AoP	Area of Protection
AP	Acidification Potential
b_m	baseline mortality
BMD10	Benchmark dose
BOD	Biological Oxygen Demand
CBA	Cost Benefit Analysis
CEA	Cost Effectiveness Analysis
CML	Centre of Environmental Science at Leiden University
COD	Chemical Oxygen Demand
COI	Costs of Illness
CV	Coefficient of Variation
CVM	Contingent Valuation Method
DALY	Disability Adjusted Life Years
DEAM	Data for Environmental Analysis and Management
DeNOx	Control device to reduce NOx emissions
DfE	Design for Environment
DM	Dematerialisation
D-R	Dose-Response
DT	Detoxification
DW	Disability Weight
DWD	Deutscher Wetterdienst
Eco-ind.	Eco indicator
ED_{10}	Effect Dose inclucing a 10% response over background
EDIP	Environmental Design of Industrial Products
EEC	External enviromental cost
EIA	Environmental Impact Assessment
EL	Environmental Load
ELG	Eco-labelling
EMA	Environmental Management and Audit
EPS	Environmental Prioritary Strategies
Equiv.	Equivalent
E-R	Exposure-Response
ERA	Environmental Risk Assessment
ERV	Emergency room visit
ETH	Eidgenössische Technische Hochschule
GenCat	Generalitat de Catalunya

GIS	Geographic Information System
GWP	Global Warning Potential
HTF	Human Toxicity Factor
HTP	Human Toxicity Potential
HU	Hazard Units
IOA	Input-Output Analysis
IP	Intermediate Produkt
IPA	Impact Pathway Analysis
IRIS	Integrated Risk Information System
ISC	Industrial Source Complex
ISO	Internationa Standard Organisation
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCM	Life Cycle Management
LOEL	Lowest Observable Effect Level
MC	Monte Carlo
MCE	Multi-Criteria Evaluation
MFA	Material Flow Accounting
MIPS	Material Intensity Per Service-unit
MSWI	Municipal Solid Waste Incinerator
NOEL	No Observable Effect Level
NOEC	No Effect Concentration
NP	Nutrification Potential
ODP	Ozone Depleton Potential
PAF	Potentially Affected Fraction
PAH	Poly-Aromatic Hydrocarbons
PCDD/Fs	Dioxins and Furans
PE	Population Exposure
PM	Particulate Matter (PM10: Particle with an aerodynamic diameter of ≤ 10)
POCP	Photochemical Ozone Creation Potential
PR	Process
RE	Receptor exposure
REW	Relative Excedance Weighted
RfD	Reference dose
RM	Raw material
SETAC	Society of Environmental Toxicology and Chemistry
SF	Slope Factor
SFA	Substance Flow Accounting
SIRUSA	Societat d'Incineració de Residus Urbans, S.A.
SMC	Servei Meteorologic de Catalunya
SPI	Sustainable Process Index
ST	Short Term
TD	Tumour dose
TEQ	Toxicity Equivalent
TEAM	Tool for Environmental Analysis and Management
TOC	Total Organic Carbon

TQM	Total Quality Management
UF	Uncertainty factor
UNEP	United Nations Environment Programme
US-EPA	United States Environment Protection Agency
UWM	Uniform World Model
VLYL	Value of Year Lost
VOC	Volatile Organic Carbon
VSL	Value of Statistical Life
WHO	World Health Organisation
WTA	Willingness-to-Accept
WTM	Windrose Trajectory Model
WTP	Willingness-To-Pay
YLD	Years of Life Disabled
YOLL	Years of Life Lost

Symbols

ρ	Receptor density
$\tilde{e}_v$	Weighted eco-vector
φ	Angle
η	Efficiency
ξ	Standard Deviation (of the Gaussian variable)
Δ	Increment
μ	Ordinary Mean
λ	Specific weighting factor
θ	Standard Deviation
σ	Standard Deviation (of lognormal distribution)
μ_g	Geometric mean
σ_g	Geometric Standard deviation
A	Area
τ	Residance time
c	concentration
C	Cost
D	Damage
E	Effect (factor)
E	Energy flow
$\mathbf{E_M}$	Eco-Technology matrix
$\mathbf{e_v}$	Eco-vector
F	Fate and exposure (factor)
H	Variation insensivity in the human population
h	Height
I	Incremental receptor exposure per mass of pollutant emitted
k	removal velocity
M	Mass
P	Quantity of pollutants
Q	Emission rate
r	radius or discount rate

R	Outer boundary of the modelling area
T	Duration
u	Mean wind speed
v	vector
V	Volume
W	Waste
$\mathbf{WE_M}$	Weighted eco-matrix or damage matrix
$\mathbf{W_M}$	Weighting or damage-assigning Matrix
X_o	Damage today
X_t	Damage through the years
z_s	Height above mean sea level

Indices

1	Primary pollutant
2	Scondary pollutant
CR	Concentration-Respone
e	energy or ecological
e, eco	ecological
EE	Ecosystem Exceeded
eff	effective
Env	Environment
far	long-range contribution
g	geometric
i	emission situation
m	mass or target medium
M	Matrix
n	initial medium
near	short-range contribution
p	Pollutant
p	pollutant
P	Product
Prod	Produktion
r	receptor
RE	Relative Exceedance
s	stack
uni	uniform
y	lateral
z	vertical

I.2 Tables

I.3 Figures

Personal data

P.1 Curriculum Vitae

Guido Willi Sonnemann was born in 1966 in Germany. He finished the grammar school in 1986 in Plön. From 1986 to 1988 he made his civil service in a living group for mentally handicapped people in Stetten. Before this duty he went three months for work to Israel and afterwards to the US in order to gain different experiences. In 1991, he got his Bachelor of Science degree in environmental engineering at the TU Berlin. The next year he studied Chemical Engineering at the TU Compiègne (France). Then back at the TU Berlin he specialised in soil and waste treatment. As tutor for courses in waste minimisation he had the first contact with Life Cycle Assessment (LCA) as an integrated way to reduce waste. During his studies he has been working part-time in the environmental consultancy GUT and the TU Berlin as well as in the form of an internship in several other companies, e.g. VEW and Dow Chemical. In 1995, he finished his master studies and obtained the 'Diplom Ingenieur' from the TU Berlin. Then, he got the scholarship from the French government and the German DAAD to make the D.E.A. ('Diplôme d'Etudes Approfondies') in Water Chemistry and Microbiology from the University of Potiers (France) with a project on coagulation-flocculation at the University of Limoges (France), enhancing in this way his knowledge in the area of water treatment. He was offered in 1996, when working for the international consultancy 'Consulting Engineers Salzgitter' in Germany, the position as a research fellow at the Department of Chemical Engineering of the Universitat Rovira i Virgili (URV) in Tarragona (Spain). This was the result of a contact made during the work on LCA at the TU Berlin. The first year in Tarragona he continued working in the field of water treatment by setting up a waste water pilot plant, but since 1997 he has carried out research on LCA and related environmental management tools. He was very actively involved in the creation of the Environmental Management and Analysis Group (AGA) of the URV and a Socrates exchange programme between the URV and the TU Berlin. In 1998 he passed the qualifiers for the Ph.D. studies in Chemical Engineering. His research activities are focused predominantly on Life Cycle Impact Assessment (LCIA) and the uncertainty analysis, interfaces and integration of environmental management tools, especially LCA, environmental risk assessment, impact pathway analysis and environmental costs. At present, he is member of the Society of Environmental Toxicology and Chemistry (SETAC), the Society for Industrial Ecology and the Catalan network for LCA (Xarxa ACV). He actively participated in several international groups (CHAINET and SETAC-Europe working groups on 'Scenario Development in LCA' and 'LCIA'). Since 2000 he has been working for the UNEP office in Paris promoting the UNEP/ SETAC Life Cycle Initiative and elaborating a report on LCIA. In his appointment at the URV, Guido Sonnemann has been actively involved in educational matters. Moreover, he taught several specialised courses in the field of environmental management.

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S.1 Resum

Les estimacions de danys ambientals en cadenes de processos necessiten l'avaluació d'impactes ambientals en dues perspectives: orientades cap a cadenes de processos i orientades localment. Per a ambdues perspectives s'han desenvolupat eines específiques d'avaluació.

L'Avaluació del Cicle de Vida (ACV o LCA) és una eina, bastant nova, orientada cap a cadenes de processos, per avaluar el perfil ambiental dels productes, enfocada cap al cicle de vida complet d'aquests: des de l'extracció de recursos, fabricació i ús fins a la disposició final del producte. A través de totes aquestes fases, el consum de recursos i l'emissió de contaminants a l'aire, a l'aigua i a la terra s'identifiquen amb l'anàlisi de l'Inventari del Cicle de Vida (LCI). Posteriorment, segueix la fase de l'Avaluació de l'Impacte del Cicle de Vida (LCIA), l'objectiu del qual és avaluar els resultats d'un Inventari del Cicle de Vida del sistema d'un producte, per entendre millor el seu significat ambiental.

L'Avaluació del Risc Ambiental (ERA) de substàncies químiques és una eina per avaluar el risc de substàncies químiques específiques, que són perjudicials per a l'home o per al medi ambient, sota certes circumstàncies. Comparant amb el mètode orientat cap a cadenes de processos, la LCIA, que en principi considera els llocs d'una manera genèrica, el mètode ERA, comunament usat per avaluacions de substàncies químiques, és específic del lloc, ja que generalment és aplicat per l'avaluació del perill d'un procés en un lloc concret. L'Anàlisi de Vies d'Impacte (IPA) és un mètode semblant que s'ha desenvolupat per a l'avaluació dels danys ambientals en termes de paràmetres d'impactes físics, com per exemple els casos de càncer o els dies d'activitat limitada, basat en experiències fetes amb l'aplicació de l'ERA. A l'IPA normalment els paràmetres d'impacte físic estan convertits en costs externs.

Per manca d'una referència comuna, la comparació entre diferents danys ambientals suposa, inevitablement, un judici de valor. Un camí per convertir el dany ambiental en resultats significatius per a la societat és mitjançant costs externs. Els danys externs poden ser introduïts en les expressions de balanços econòmics, donant valor monetari als danys ambientals, permetent una comparació amb altres tipus de costs. Seguint aquest procediment, poden ser internalitzats. No obstant això, existeixen diversos mètodes de monetització i, depenent de la perspectiva socioeconòmica, els individuals, podrien preferir altres esquemes de valoració. En general, es pot dir que els esquemes de valoració aprovats per l'organitzacions internacionales tenen un alt nivell d'acceptació.

Per tal de superar els desaventatges dels mètodes citats, específics i genèrics en relació amb el lloc, recentment s'han desenvolupat mètodes d'avaluació d'impacte que s'anomenen depenents del lloc. Per ser operatius, la consideració de la diferenciació espacial en el procés d'avaluació hauria de requerir un mínim de dades addicionals, és a dir, només alguns paràmetres generals de lloc. Aquests paràmetres han de tenir en compte el transport i l'exposició de contaminant i la informació d'impacte en forma d'indicadors, que seran aplicables per classes de llocs d'emissions en lloc d'indrets específics.

Desenvolupaments recents estan portant avenços a la pràctica de la LCIA, però no es pot esperar que les majors limitacions i incerteses desapareixin en un futur proper. Només es poden fer òbvies mitjançant l'anàlisi sistemàtica d'incertesa i comunicant els resultats d'una manera adequada. Seria un pas important, un pas més enllà, si es pogués desenvolupar una metodologia que abarqués tant els punts mitjans com els punts finals, incloent els paràmetres més importants d'ambdós tipus de mètodes, i així facilitar la modelització al llarg dels dos mètodes i comparar els resultats entre ells. Això pot implicar la integració d'altres eines de gestió ambiental. El nivell de sofisticació pot dependre del tipus d'aplicació i la disponibilitat de dades. Es consideren punts mitjans potencials d'impacte o quasevol punt en la cadena de causa-efecte abans del punt final. Els punts finals son els impactes reals o danys que causa la càrrega ambiental analitzat en el inventari.

Per a permetre un alt nivell de localitat i d'exactitud, aquesta tesi fa la diferència entre dos tipus de cicle de vida respecte al seu número de processos: cadences de processos industrials amb 100 o menys processos o un sistema de producte complet amb un gran número de processos, com per exemple un ordinador. S'han realitzat pocs esforços, fins ara, per explorar sistemàticament les incerteses inherents, interfases i tipus de possibilitats d'integració i comunicació dels mètodes d'avaluació ambientals citats, orientats per un costat a cadenes de processos i per l'altre els orientats localment, en el cas de cadenes de processos industrials. Això és el que la tesi té en ment. La qüestió de recerca d'aquesta tesi és:

- De quina manera es poden fer estimacions de danys ambientals en cadenes de processos industrials d'una manera, al més acuradament possible, però encara amb un esforç acceptable i d'una manera comunicable?

Això vol dir que l'objectiu d'aquesta tesi, tal com s'ha descrit al capítol 1, és trobar un compromís adequat entre els mètodes d'avaluació d'impacte ambiental, orientats a cadenes de processos i els orientats localment i, convertir les estimacions de danys ambientals en resultats significatius, com per exemple els costs ambientals.

A partir d'aquest objectiu clau, es poden derivar les següents qüestions:

- Quines són les incerteses dels mètodes d'avaluació de danys ambientals?
- Quines possibilitats existeixen per a les interfases i per a la integració de les eines de gestió ambiental, especialment l'ACV, l' ERA, l'IPA i els costs mediambientals?
- Com es poden posar mètodes de punts mitjans i de punts finals a la LCIA, en una metodologia comuna?

Evidentment, totes aquestes qüestions formen part del que s'anomena sofisticació d'eines de gestió ambiental i requereixen models complexos per a obtenir una resposta.

El capítol 2 familiaritza al lector amb els conceptes bàsics, eines i elements tècnics, darrere d'estimacions de danys ambientals en cadenes de processos industrials. Els mètodes descrits són revisats críticament. També s'explica l'anomenada caixa d'eines per a la gestió ambiental i es mostra la seva dependència d'aplicació en una perspectiva de cadena de processos, i s'ha desenvolupat també una guia. Es recomana tenir cura en la selecció d'una eina per a un determinat problema ambiental, ja que el resultat d'un estudi és d'alguna manera condicionat per l'eina analítica de gestió ambiental emprada.

En general, tots els mètodes analítics d'avaluació d'impacte ambiental inclouen una anàlisi d'efecte que indica la severitat de les conseqüències degudes a la contaminació. Els elements crucials són funcions de dosi-resposta i exposició-resposta. A més a més, cada anàlisi pot incloure una valoració de la relació entre els diversos tipus d'efectes. Una opció de valoració consisteix en els costs que són un aspecte important de la gestió ambiental. Es dóna un enfocament especial als costs ambientals externs que s'obtenen per monetització. Es demostra la relació entre la monetització i altres esquemes de valoració.

Després, es descriu la metodologia de l'ACV. Pel que fa al subjecte de LCIA, per començar és presenta la metodologia i es discuteixen diversos mètodes, especialment aquells amb un índex singular de perfil ambiental. Els nous avançaments inclouen els mètodes de la LCIA orientats cap a danys o punts finals. Finalment, els darrers desenvolupaments a la LCIA es caracteritzen per la sofisticació, que es correspon amb l'habilitat de proveir informes molt acurats i comprensius que reflecteixen l'impacte potencial de les càrregues ambientals, per ajudar en la presa de decisions en cada cas particular.

En la secció següent, es descriuen l'Avaluació del Risc Ambiental i l'Avaluació de les Vies d'Impacte. Mentre el mètode ERA convencional està enfocat cap a l'avaluació de la probabilitat de l'ocurrència de perill, el mètode IPA, més recentment desenvolupat, ofereix una metodologia per a les estimacions. En aquest estudi, l'aplicació dels mètodes ERA requereix elements tècnics complexos, tals com la dispersió Gaussiana i models de transport i exposició de contaminants multimèdia, descrits espressament aquí, a propòsit. Es es dóna a més una introducció al model de software integrat d'avaluació d'impacte anomenat EcoSense, que és emprat a l'IPA.

Finalment, l'última secció del capítol 2, analitza el caràcter dels cicles de vida, respecte al nombre dels seus processos i estableix una diferenciació entre les cadenes de processos industrials, menys de 100, i els sistemes de productes complets. S'ha demostrat la rellevància d'aquesta diferenciació en la manera d'avaluar els impactes ambientals.

El capítol 3 introdueix el lector al cas pràctic de la incineració de residus amb un enfocament especial en els impactes a la salut humana. S'han obtingut resultats, basats en un conjunt comú de dades fonamentals, per a aplicacions dels mètodes emprats comunament ACV, ERA i IPA a la Incineradora de Residus Sòlids Urbans (MSWI) de Tarragona i la seva cadena de processos en dues situacions: amb i sense un sistema avançat de tractament de gas. A més, es descriu, breument, un model modular que permet generar un escenari d'incineradora MSWI amb un sistema DeNOx d'eliminació de NOx. S'ha verificat, en una part d'aquest capítol, que aquest cicle de vida pot ser definit com una cadena de processos industrials.

Amb el capítol 4 comença el desenvolupament de la metodologia. L'existència de diversos tipus d'incerteses, s'han citat sovint, com una limitació crucial per a una clara interpretació d'estimacions de danys ambientals. Per això, en aquest capítol 4, s'ha presentat una estratègia per a l'avaluació estocàstica d'incerteses per a la simulació de Monte Carlo. S'ha presentat una metodologia, per aquest tipus d'avaluació, per l'ICV i per l'AVI. El procediment és aplicat a l'ICV del cas pràctic i les estimacions de danys de les emissions de l'IRSU per l'AVI a l'escala local. L'aplicació del mètode a l'AVI demostra que la incertesa i la variabilitat calculades són menys importants que les obtingudes en un estudi amb mètodes anlítics. L'anàlisi de sensibilitat indica que l'impacte principal sobre la salut humana no prové dels PCDD/Fs i els metalls pesats, sinó de les partícules i del NOx, usant funcions de dosi-resposta toxicològiques i exposició-resposta epidemiològiques.

Aquesta aplicació paral·lela permet mostrar que les incerteses de l'anàlisi d'inventari són menys importants que les de l'avaluació de danys; especialment l'ús de models de dispersió, de les funcions de dosi-resposta i d'exposició-resposta, així com l'aplicació dels esquemes de valoració, impliquen enormes incerteses.

El capítol 5 ofereix un algorisme matemàtic i un esquema que permet la diferenciació espacial a diferents nivells de detall i proposa una integració d'ACV, IPA i costs ambientals. Això és especialment aplicable a les anomenades cadenes de processos industrials, no a cicles de vida complets de sistemes de productes. Per això, la metodologia establerta s'anomena Estimacions de Danys Ambientals per a Cadenes de Processos Industrials. Una part important d'aquesta metodologia és el seu algorisme matemàtic que permet de gestionar la distribució de la càrrega ambiental al llarg de les etapes del cicle de vida i l'assignació dels danys a la regió corresponent d'emissió. Es tracta esencialment d'un diagrama de flux. S'han donat diverses opcions per trobar un compromís adequat entre l'avaluació d'impactes ambientals de tipus genèric i específics de lloc i per convertir les estimacions de danys ambientals en resultats significatius tals com costs ambientals.

La metodologia posa mètodes de punts mitjos i de punts finals a la LCIA en un marc comú. D'aquesta manera, s'ha facilitat la comparació directa de resultats d'indicadors de punts mitjos i punts finals. El diagrama de flux està dividit en diferentes parts, que d'alguna manera són similars a les fases del ACV, però que tenen sovint un contingut diferent. La primera fase és la de Definició de l'Objectiu i l'Abast amb la selecció de l'esquema d'avaluació i agregació que vol usar la persona que pren decisions, la segona fase és l'IPA i la tercera, la LCIA. Aquí s'atura la similitud amb la metodologia ACV; en lloc de l'interpretació, els següents passos són l'Anàlisi del Domini i la Diferenciació Espacial, una Avaluació del Transport i de l'Exposició de contaminants i l'Anàlisi de Conseqüència així com el Perfil de Dany i l'Ecoeficiència. En la metodologia proposada, s'empren un nombre d'eines analítiques, com l'ERA i elements tècnics com l'avaluació d'incertesa per a la simulació de Monte Carlo. Es preveu, opcionalment, la consideració d'accidents. Finalment, la metodologia s'ha aplicat als resultats de l'estudi de l'ACV de la cadena de processos de la incineradora de residus amb la finalitat de mostrar la seva aplicabilitat.

El capítol 6 aplica i segueix desenvolupant una metodologia per a l'avaluació d'impactes depenents del lloc com una manera per aconseguir un compromís entre les avaluacions del dany específic de lloc (a nivell de punts finals) i els indicadors potencials del cicle de vida (a nivell de punts mitjans). Les raons principals pel desacord entre els potencials d'impacte calculats i els impactes reals esperats en la LCIA són, d'una banda, la no consideració de la distribució espaial dels receptors i, d'altra banda, la manca d'informació sobre el punt d'emissió i les condicions de dispersió corresponents en el medi respectiu. Per relacionar aquests factors principals, el mètode presentat utilitza classes d'impacte genèric representatives corresponent a diverses distribucions de receptors i condicions de dispersió basades en un raonament estadístic. El mètode existent s'ha adaptat per incloure models més sofisticats: un programa detallat de dispersió i un sistema d'informació geogràfic. No obstant això, ha quedat clar que és necessària més recerca sobre com definir estadísticament les característiques del focus emissor. El mètode bàsic existent s'ha generalitzat per cobrir més receptors. S'aplica en detall pels efectes a la salut humana a cause de les emissions a l'aire a Catalunya; s'han calculat factors a l'àmbit comarcal.

El mètode d'avaluació d'impacte depenent del lloc encaixa perfectament amb la metodologia desenvolupada al capítol 5 i s'aplica per comparar la importància dels transports, en relació al procés de la incineració de residus, que s'ha identificat per ésser de particular interès dintre l'aplicació del capítol 5. S'ha demostrat que la diferenciació espacial en la LCIA és factible amb un esforç raonable i poca informació addicional necessària dintre del LCI (nom de la comarca, alçada de la xemeneia, informació sobre la distribució del tamany de les partícules). La comparació dels resultats derivats mitjançant indicadors d'impacte de punt final amb els obtinguts per a l'indicador de punt mitjà, Potencial de Toxicitat Humana (HTP), indica que el HTP desestima l'impacte ambiental dels processos de transport. Aquest és especialment rellevant per a les tecnologies de tractament de gas més avançades, és a dir, addicionalment amb DeNOx, on la capacitat de neteja s'ha compensat més i més pels esforços addicionals en la provisió de primeres matèries, incloent-hi el transport suplementari.

El capítol 7 presenta la discusió general. El marc metodològic presentat està limitat, almenys pel que fa a dos aspectes: les incerteses inherents i la seva aplicabilitat principal a cadenes de processos industrials, amb 100 o menys processos. Les aplicacions més adequades del sistema d'avaluació de danys ambientals presentats per a cadenes de processos industrials son la gestió del final de vida i l'optimització ambiental de la recerca de llocs per a noves plantes. És evident que hi ha molts problemes ambientals relacionats amb una perspectiva de cicle de vida que no poden ser estudiats mitjançant aquesta metodologia, almenys, en aquest moment, a causa de la insuficiència de disponibilitat de dades i la irrealitzable càrrega de feina. Basada en aquesta i altres reflexions, es proposa una estratègia d'aplicacions genèriques. Aquesta estratègia fa referència al desenvolupament d'una metodologia acceptada internacionalment per mètodes AICV de punts finals i punts mitjans.

En les conclusions, capítol 8, s'emfatitza com a nova troballa principal el desenvolupament de metodologia. Finalment, s'ofereixen perspectives interessants per a la futura recerca. S'encoratja a continuar amb el treball sobre la diferenciació de mètodes de cicle de vida respecte al seu nombre de processos. Es conclou que s'ha establert la base per a la creació d'una nova generació de models de gestió integrada de residus que inclou l'optimització de la ubicació de plantes de tractament de residus i de de la difució de les rutes de transports corresponents. No obstant això, encara queda molt camí de fer per tal que aquest mètode sigui completament operacional.

S.2 Resumen

Las estimaciones de daños medioambientales en cadenas de procesos requieren la evaluación de impactos medioambientales en dos perspectivas: orientado hacia las cadenas de procesos y orientado hacia sitios concretos. Para ambas perspectivas se han desarrollado herramientas específicas de evaluación.

El Análisis del Ciclo de Vida (ACV o LCA) es una herramienta, bastante nueva, orientada hacia cadenas de procesos, para evaluar el perfil medioambiental de los productos, enfocada hacia su ciclo de vida completo: desde la extracción de recursos, vía la fabricación y el uso hasta el final de vida del producto. A través de todas estas fases, el consumo de recursos y la emisión de contaminantes al aire, el agua y la tierra se identifican en el análisis del Inventario del Ciclo de Vida (LCI). Posteriormente, continua la fase de la Evaluación del Impacto del Ciclo de Vida (LCIA), el objetivo del cual es evaluar los resultados de un Inventario del Ciclo de Vida del sistema de un producto, para conocer mejor su relevancia para el medioambiente.

La Evaluación del Riesgo Ambiental (ERA) de substancias químicas es una herramienta para evaluar el riesgo de substancias químicas específicas, que son perjudiciales para el hombre o el medioambiente, bajo ciertas circunstancias. Comparando con el método orientado hacia cadenas de procesos, la LCIA, que en principio considera los lugares de una manera genérica, el método ERA, comúnmente usado para evaluaciones de substancias químicas, es específico del lugar, ya que generalmente está aplicado para la evaluación del riesgo de un proceso en un sitio concreto. El Análisis de Vías de Impacto (IPA) es un método semejante que se ha desarrollado para la evaluación de los daños medioambientales en términos de parámetros de impactos físicos como por ejemplo los casos de cáncer o los días de actividad limitada, basadas en experiencias hechas en aplicaciones de ERA. En el IPA normalmente los parámetros de impacto físico están convertidos en costes externos.

A causa de la falta de una referencia común, la comparación entre diferentes daños ambientales supone, inevitablemente, un juicio de valor. Una manera para convertir el daño medioambiental en resultados significativos para la sociedad es mediante costes externos. Dando valor monetario a los daños medioambientales, los daños externos pueden ser introducidos en las expresiones de balances económicos, permitiendo una comparación con otros tipos de costes. Según este procedimiento, pueden ser internalizados. No obstante, existen diversos métodos de monetización y, dependiendo de la perspectiva socio-económica, los individuos, podrían preferir otros esquemas de valoración. En general, se puede decir que esquemas de valoración aprobadas por organizaciones internacionales tienen un alto nivel de aceptación.

A fin de superar las desventajas de los métodos citados, específicos y genéricos en relación con el lugar, recientemente se han desarrollado métodos de evaluación de impacto que se llaman dependientes del sitio. Para ser operacional, la consideración de la diferenciación espacial en el proceso de la evaluación requiere un mínimo de datos adicionales, es decir sería preciso disponer de algunos parámetros generales del lugar. Estos parámetros tienen que considerar el transporte i la exposición del contaminante y la información sobre el impacto en forma de indicadores, que serán aplicables por clases de sitios de emisiones en lugar de lugares específicos.

Desarrollos recientes llevan avances a la práctica de la LCIA, pero no se puede esperar que las mayores limitaciones e incertidumbres desaparezcan en un futuro próximo. Sólo se pueden hacer obvios mediante el análisis sistemático de incertidumbres y comunicando los resultados de una manera adecuada. Sería un paso importante adelante si se pudiera desarrollar una metodología que abarcara tanto los puntos medios como los puntos finales, incluyendo los parámetros más importantes de ambos tipos de métodos, y así facilitar la modelización a lo largo de los dos métodos y comparar los resultados entre ellos. Esto puede implicar la integración de otras herramientas de gestión ambiental. El nivel de sofisticación puede depender del tipo de aplicación y de la disponibilidad de datos. Se consideran puntos medios potenciales de impacto o cualquier punto en la cadena de causa-efecto antes del punto final. Los puntos finales son los impactes reales o daños que causan las cargas ambientales analizados en el inventario.

Para permitir un alto nivel de localización y de exactitud, esta tesis hace la diferencia entre dos tipos de ciclo de vida respecto a su numero de procesos: cadenas de procesos industrials con 100 o menos procesos o un sistema de producto completo con un gran numero de procesos, como por ejemplo un ordinador. Se han realizado pocos esfuerzos, hasta la fecha, para explorar sistemáticamente las incertidumbres inherentes, interfases y tipos de posibilidades de integración y comunicación de los métodos de evaluación medioambientales citados, orientados de un lado hacia cadenas de procesos y de otro lado hacia los lugares específicos, en el caso de cadenas de procesos industriales. Esto es lo que la tesis tiene en mente. La pregunta de investigación de esta tesis es:

- ¿De qué manera se pueden hacer estimaciones de daños medioambientales en cadenas de procesos industriales de una manera, lo más exactamente posible, pero todavía con un esfuerzo aceptable y de una manera comunicable?

Esto quiere decir que el objetivo de esta tesis, tal como se ha escrito en el capítulo 1, es encontrar un compromiso adecuado entre los métodos de evaluación de impacto medioambiental, orientados hacia cadenas de procesos y los que incluyen una especificación de lugar, y convertir las estimaciones de daños medioambientales en resultados significativos, como por ejemplo los costes medioambientales.

A partir de este objetivo clave, se pueden derivar las siguientes preguntas:

- ¿Cuáles son las incertidumbres de los métodos de evaluación de daños medioambientales?
- ¿Qué posibilidades existen para las interfases y para la integración de las herramientas de gestión medioambiental, especialmente el ACV, la ERA, el IPA y los costes medioambientales?
- ¿Cómo se pueden poner métodos de puntos medios y de puntos finales en la LCIA en una metodología común?

Evidentemente, todas estas preguntas forman parte de lo que se llama sofisticación de herramientas de gestión medioambiental y requieren modelos complejos para obtener una respuesta.

El capitulo 2 familiariza al lector con los conceptos básicos, herramientas y elementos técnicos desarrallados para las estimaciones de los daños medioambientales en cadenas de procesos industriales. Se ha revisado críticamente a los métodos descritos. También se explica la llamada caja de la gestión medioambiental y se muestra su dependencia de la aplicación en una perspectiva de cadenas de procesos. Como etapa final, se ha desarrollado una guía para facilitar la elección de herramientas. Se recomienda tener cuidado en la selección de una herramienta para un determinado problema medioambiental, ya que el resultado de un estudio está, de alguna manera, predeterminado por la herramienta analítica de gestión medioambiental utilizada.

En general, todos los métodos analíticos de evaluación de impacto medioambiental incluyen un análisis de efecto que indica la severidad de las consecuencias debidas al contacto con el contaminante. Los elementos cruciales son funciones de dosis-respuesta y exposición-respuesta. Además, cada análisis puede incluir una valoración de la relación entre los diversos tipos de efectos. Una opción de valoración consiste en los costes que son un aspecto importante de la gestión medioambiental. Se da un enfoque especial a los costes medioambientales externos que se obtienen por monetización. Se demuestra la relación entre la monetización y otros esquemas de valoración.

Después, se describe la metodología del ACV. Por lo que a la LCIA se refiere, al principio se presenta la metodología y se discuten diversos métodos, especialmente estos con un índice singular de perfil ambiental. Los nuevos avances incluyen los métodos de la LCIA orientados hacia los daños o puntos finales. Finalmente, los últimos desarrollos en la LCIA se caracterizan por la sofisticación, que se corresponde con la habilidad de producir informes, muy exactos y comprensivos, que reflejan el impacto potencial de las cargas medioambientales, para ayudar en la toma de decisiones en cada caso particular.

En la sección siguiente, se describen: la Evaluación del Riesgo Ambiental y el Análisis de las Vías de Impacto. Mientras el método ERA convencional está enfocado hacia la evaluación de la probabilidad de la ocurrencia de peligro, el método IPA, más recientemente desarrollado, ofrece una metodología para las estimaciones. En este estudio, la aplicación de los métodos ERA requiere elementos técnicos complejos, tales como los modelos de dispersión Gaussiana y de transporte y exposición de contaminantes multimedia, descritos aquí. Además, se da una introducción al modelo de software integrado de evaluación de impacto llamado EcoSense, que se utiliza en el IPA.

Finalmente, la última sección del capítulo 2, analiza el carácter de los ciclos de vida, respecto al número de sus procesos, y establece una diferenciación entre las cadenas de procesos industriales, menos de 100, y los sistemas de productos completos. Se ha demostrado la relevancia de esta diferenciación, en la manera de evaluar los impactos medioambientales.

El capítulo 3 introduce al lector al caso práctico de la incineración de residuos con un enfoque especial en los impactos a la salud humana. Se han obtenido resultados, basados en un conjunto común de datos fundamentales, para aplicaciones de los métodos utilizados comúnmente ACV, ERA y IPA a la Incineradora de Residuos Sólidos Urbanos (MSWI) de Tarragona y su cadena de procesos en dos situaciones: con y sin un sistema avanzado de tratamiento de gas. Además, se describe, brevemente, un modelo modular que permite generar un escenario de MSWI con un sistema de eliminación de NOx denominado DeNOx. Se ha verificado, en una parte de este capítulo, que este ciclo de vida puede ser definido como una cadena de procesos industriales con, aproximadamente, unos 20 procesos significativos, y no como un sistema de producto completo, como por ejemplo un ordenador.

Con el capítulo 4 empieza el desarrollo de la metodología. La existencia de diversos tipos de incertidumbres, se ha citado frecuentemente, como una limitación crucial para una clara interpretación de estimaciones de daños medioambientales. Por eso, en este capítulo 4, se presenta una estrategia para la evaluación estocástica de incertidumbres por la simulación de Monte Carlo. Se ha elaborado una metodología, para este tipo de evaluación, para el LCI y para el IPA. El procedimiento está aplicado al LCI del caso práctico y las estimaciones de daños de las emisiones de la MSWI para el IPA a escala local. La aplicación del método en el IPA demuestra que la incertidumbre y la variabilidad calculada son menos importantes que las publicadas sobre un estudio con métodos analíticos de evaluación de incertidumbres. El análisis de sensibilidad indica que el impacto principal, sobre la salud humana no proviene de los PCDD/Fs y los metales pesados, sino de las partículas y del NOx, usando funciones de dosis-respuesta de toxicología y de exposición-respuesta de epidemiología.

Esta aplicación paralela permite mostrar que las incertidumbres en el análisis de inventario son menos importantes que aquellas en la evaluación de daños; especialmente el uso de modelos de dispersión, de las funciones de dosis- y de exposición-respuesta de la misma manera como de los esquemas de valoración que implican enormes incertidumbres.

El capítulo 5 ofrece un algoritmo matemático y un esquema que permite la diferenciación espacial a diferentes niveles de detalle y propone una integración de ACV, IPA y costes medioambientales. Esto es especialmente aplicable para las mencionadas cadenas de procesos industriales, es decir, no a ciclos de vida completos de sistemas de productos. Por eso, la metodología establecida se llama Estimaciones de Daños Medioambientales para Cadenas de Procesos Industriales. Una parte importante de esta metodología es su algoritmo matemático que permite gestionar la distribución de la carga medioambiental a lo largo de las etapas del ciclo de vida y la asignación de los daños a la región correspondiente de emisión. El cuerpo consiste en un esquema metodológico. Se han elaborado diversas opciones para encontrar un compromiso adecuado entre la evaluación de impactos medioambientales de tipo genérico y específicos del sitio y para convertir las estimaciones de daños medioambientales en resultados significativos tales como costes medioambientales.

La metodología presentada en este trabajo pone métodos de puntos medios y de puntos finales a la LCIA en un marco común. De esta manera, se ha facilitado la comparación directa de resultados de indicadores de puntos medios y puntos finales. El esquema está dividido en diferentes partes, que de alguna manera son similares a las fases del ACV, pero que tienen a menudo un contenido diferente. La primera fase es la de Definición del Objetivo y del Alcance con la selección del esquema de evaluación y agregación que quiere usar la persona que toma decisiones, la segunda fase es el LCI y la tercera, la LCIA. Aquí se acaba la similitud con la metodología ACV; en lugar de interpretación, los siguientes pasos son el Análisis del Dominio y la Diferenciación Espacial, una Evaluación de Transporte y de Exposición de contaminantes y Análisis de Consecuencias de la misma manera que el Perfil de Daño y la Ecoeficiencia. En la metodología propuesta, se utilizan un número de herramientas analíticas, como la ERA y elementos técnicos como la evaluación de incertidumbres para la simulación de Monte Carlo. Se prevé, opcionalmente, la consideración de accidentes. Finalmente, la metodología se ha aplicado a los resultados del estudio de ACV de la cadena de procesos de la incineradora de residuos con la finalidad de mostrar su aplicabilidad.

El capítulo 6 aplica y continua desarrollando un método existente para la evaluación de impactos dependientes del sitio como una manera para conseguir un compromiso entre las evaluaciones del daño específico de sitio (a nivel de puntos finales) y los indicadores potenciales del ciclo de vida (a nivel de puntos medios). Las razones principales por el desacuerdo entre potenciales de impacto calculados y impactos reales esperados en el LCIA son, de un lado, no considerar la distribución espacial de los receptores y, de otro lado, la falta de información sobre el punto de emisión y las condiciones de dispersión correspondientes en el medio respectivo. Para relacionar estos factores principales, el método presentado utiliza clases de impacto genéricas, representatives, correspondientes a las distribuciones de receptores y las condiciones de dispersión, todo basado en un razonamiento estadístico. El método existente se ha adaptado para incluir modelos más sofisticados: un programa detallado de dispersión y un sistema geográfico de información. No obstante, ha quedado claro que se necesita más investigación sobre cómo definir estadísticamente las características del foco emisor. El método básico existente se ha generalizado para cubrir más receptores. Se ha aplicado en detalle para los efectos sobre la salud humana debido a las emisiones en el aire en Cataluña; se han calculado factores en el ámbito comarcal.

El método de evaluación de impacto dependiente del sitio encaja perfectamente en la metodología desarrollada en el capítulo 5 y se ha aplicado para comparar la importancia del transporte, con relación al proceso de la incineración de residuos, que se ha identificado de ser de particular interés. Se ha demostrado que la diferenciación espacial en la LCIA es factible con un esfuerzo razonable y poca información adicional necesaria dentro del LCI (nombre de la comarca, altura de la chimenea, información sobre la distribución del tamaño de las partículas). La comparación de los resultados derivados mediante indicadores de impacto de punto final con los obtenidos por el indicador de punto medio, Potencial de Toxicidad Humana (HTP), indica que el HTP desestima el impacto medioambiental de los procesos de transporte. Esto es especialmente relevante para las tecnologías de tratamiento de gas más avanzadas, es decir, adicionalmente con DeNOx, donde la capacidad de limpieza se está compensando más y más por los esfuerzos adicionales que implica el suministro de materias primas, incluyendo el transporte suplementario.

En el capítulo 7 se presenta la discusión general. El marco metodológico presentado está limitado a dos aspectos: las incertidumbres inherentes y su aplicabilidad principal a cadenas de procesos industriales, que se han definido aquí como cadenas con 100 procesos o menos.

Las aplicaciones mas adecuadas del sistema de evaluación de daños medioambientales presentado para las cadenas de procesos industriales son la gestión del final de vida y la optimización medioambiental de la búsqueda de lugares para nuevas plantas. Es evidente que hay muchos problemas medioambientales relacionados con una perspectiva de ciclo de vida que no se pueden estudiar mediante esta metodología, al menos, en este momento, debido a la insuficiencia de disponibilidad de datos y la irrealizable carga de trabajo que se requiere. Basada en ésta y en otras reflexiones, se propone una estrategia de aplicaciones genéricas. Esta estrategia hace referencia al desarrollo de una metodología aceptada internacionalmente para métodos de LCIA de puntos finales y puntos medios.

En las conclusiones, capítulo 8, el desarrollo de la metodología destaca como la aportación principal de la tesis. Finalmente, se ofrecen perspectivas interesantes para la investigación futura. Se anima a continuar con el trabajo sobre la diferenciación de métodos de ciclo de vida con respecto a su número de procesos. Se concluye que se ha establecido la base para la creación de una nueva generación de modelos de gestión de residuos integrales que incluyen la optimización de los lugares de instalación de plantas de tratamiento de residuos y de rutas de transportes relacionadas. No obstante, todavía queda mucho camino por hacer para que este método sea completamente operacional.

S.3 Résumé

Les estimations des dommages causés à l'environnement dans des chaînes de procédés ont besoin d'analyses d'impacts orientées dans deux directions: vers des chaînes de procédés et vers des sites concrets. Pour ces deux types d'études, des outils d'analyse spécifiques ont été développés.

L'Analyse du Cycle de Vie (ACV ou LCA) est un outil, assez récent, orienté vers les chaînes de procédés, pour analyser le profil environnemental des produits, en considérant tout leur cycle de vie: de l'extraction de ressources, via la fabrication et l'usage jusqu'à la fin de vie du produit. Lors de toutes ces phases, la consommation de ressources et les émissions de polluants dans l'air, l'eau et le sol sont identifiées au sein de l'analyse de l'inventaire (LCI). Dans une seconde phase, l'Analyse de l'Impact du Cycle de Vie (LCIA) est réalisée. L'objectif de cette phase est l'évaluation des résultats d'un inventaire du cycle de vie d'un produit pour mieux comprendre son impact sur l'environnement.

L'Analyse du Risque Environnemental (ERA) des substances chimiques est un outil développé pour évaluer le risque de substances chimiques spécifiques, qui sont dangereuses pour l'homme ou pour l'environnement, dans des conditions définies. En comparaison avec la méthode orientée vers les chaînes de procédés (LCIA), qui en principe considère les sites d'une manière générique, la méthode ERA, normalement utilisée pour des analyses des substances chimiques, est spécifique au site, puisque généralement elle est appliquée pour l'évaluation du danger d'un procédé dans un site concret. L'Analyse de Chemins d'Impact (IPA) est une méthode similaire qui a été développée pour l'analyse des dommages environnementaux, sous la forme de paramètres d'impact physique, comme par exemple les cas de cancer ou les jours d'activité limitée. Elle est basée sur les expériences obtenues avec l'application de l'ERA. Dans l'IPA, normalement les paramètres d'impact sont convertis en coûts externes.

À cause de l'absence de référence commune, la comparaison entre les différents dommages environnementaux implique, inévitablement, un jugement de valeur. Une solution pour convertir les dommages environnementaux en résultats interprétables par la société est par des coûts externes. En donnant une valeur monétaire aux dommages environnementaux, on peut introduire les dommages externes dans des expressions de balances économiques, ce qui permettrait une comparaison avec d'autres types de coûts. On dit que selon cette procédure, il est possible d'internaliser ces dommages. Mais, il faut considérer qu'il y a diverses méthodes de monétisation et que les individus pourraient favoriser certains schémas d'évaluation, en fonction de leurs perspectives socio-économique. En général, on peut dire que les schémas d'évaluation approuvés par des organisations internationales ont un niveau plus haut d'acceptation.

Afin de surmonter les désavantages des méthodes citées, spécifiques et génériques en relation avec l'endroit, récemment des méthodes d'évaluation d'impact se sont développées qui s'appellent «dépendant d'endroit». Ça veut dire que la considération de la différentiation spatiale dans l'analyse de l'impact est réduite à un minimum de données additionnelles pour

être opérationnelle. Ceci signifie qu'il serait nécessaire de disposer de quelques paramètres généraux liés au lieu. Il faut que ces paramètres prennent en compte des informations sur le transport, l'exposition et l'effet des polluants, sous la forme d'indicateurs qui sont applicables pour des classes qui représentent des lieux d'émission.

Des développements récents amènent des évolutions dans l'utilisation de la LCIA, mais on ne peut pas prévoir la disparition prochaine de ses limitations majeures et de ses incertitudes. On peut seulement les rendre apparentes par une analyse systématique des incertitudes et par la communication des résultats de manière appropriée. Un pas important en avant serait le développement d'une méthodologie qui embrasse aussi bien les points moyens, les potentiels d'impact environnemental comme les points finaux, les dommages environnementaux, incluant les paramètres les plus importants des deux types de méthodes, pour ainsi faciliter la modélisation dans les deux méthodes et comparer les résultats entre eux. Cela peut impliquer l'intégration des autres outils de la gestion de l'environnement. Le niveau de sophistication peut dépendre du type d'application et la disponibilité de données.

Pour permettre un haut niveau de considération du lieu d'émission et d'exactitude, cette thèse fait la différence entre deux types des cycles de vie en regard à son numéro de procédés: chaînes de procedés industrials (avec 100 ou mois procédés) ou produit complêt avec un grand numéro de procédés, comme par exemple un ordinateur. Jusqu'à maintenant, peu d'efforts ont été réalisés pour explorer systématiquement les incertitudes inhérentes, des interfaces et des types de possibilités d'intégration et de communication des méthodes de l'analyse de l'impact environnemental citées (orientées d'un coté vers des chaînes de procédés et de l'autre coté vers des sites concrets) dans le cas de chaînes de procédés industriels. Ces limites sont les fondements de ce travail de thèse. La question de recherche de la thèse est:

- De quelle manière est-il possible de faire des estimations de dommages environnementaux, dans des chaînes de procédés industriels, d'une manière la plus exactement possible, d'une manière communicable, et avec des efforts acceptables.

Ceci signifie que l'objectif de la thèse, tel qu'il est défini dans le chapitre 1, est de trouver un compromis approprié entre les méthodes d'analyse de l'impact environnemental orientées vers des chaînes de procédés et celles qui considèrent spécifiquement des sites, et de convertir les estimations de dommages environnementaux dans des résultats significatifs, comme par exemple les coûts environnementaux.

A partir de cet objectif crucial, peuvent dériver les questions suivantes:

- Quelles sont les incertitudes des méthodes d'analyse des dommages environnementaux ?
- Quelles possibilités existent pour les interfaces et pour l'intégration des outils de la gestion de l'environnement, spécialement l'ACV, la ERA, l'IPA et les coûts environnementaux ?
- Comment peut-on poser des méthodes de points moyens (potentiels d'impact environnemental) et de points finaux (dommages environnementaux) de la LCIA dans une méthodologie commune?

Evidement, toutes ces questions font partie de ce qui s'appelle sophistication des outils de la gestion de l'environnement et ont besoin de modèles complexes pour obtenir une réponse.

Le chapitre 2 familiarise le lecteur avec des concepts basics, des outils et des éléments techniques développés pour les estimations des dommages environnementaux dans les chaînes de procédés industriels. Les méthodes sont décrites d'une manière critique. La boite à outils de la gestion de l'environnement est expliquée et sa dépendance vis à vis de l'application dans une perspective de chaînes de procédés est démontrée. En finalité, un guide pour faciliter le choix des outils est développé. Il est recommandé de faire attention à la sélection d'un outil, pour l'étude des problèmes environnementaux, puisque le résultat d'une étude est, d'une certaine manière, prédéterminé par l'outil analytique de gestion de l'environnement utilisé.

En général, toutes les méthodes d'analyse de l'impact environnemental incluent une analyse d'effet qui indique la sévérité des conséquences dues à la pollution. Les éléments cruciaux sont des fonctions de dose-réponse et d'exposition–réponse. En plus, chaque analyse peut inclure une évaluation de la relation entre les divers types d'effets. Une option d'évaluation consiste à estimer les coûts qui sont un aspect important de la gestion de l'environnement. Une attention spéciale est portée sur les coûts environnementaux externes qui sont obtenus par monétisation. La relation entre la monétisation et les autres schémas d'évaluation est démontrée.

Ensuite, la méthodologie de l'ACV est décrite plus en détail Les nouvelles avancées incluent les méthodes de la LCIA orientées vers les dommages environnementaux ou points finaux. Finalement, les derniers développements de la LCIA peuvent se caractériser par la sophistication, qui correspond à l'habileté à produire des rapports très exacts et compréhensibles, qui démontre l'impact possible par la pollution, pour aider à prendre des décisions dans chaque cas particulier.

Dans la section suivante l'Analyse du Risque Environnemental et l'Analyse des Chemins d'Impact sont décrites. Alors que la méthode ERA conventionnelle permet l'évaluation de la probabilité de l'occurrence du danger, la méthode IPA, plus récemment développée, offre une méthodologie pour les estimations de dommages. L'Analyse du Risque Environnemental est réalisée par l'application des modèles de dispersion Gaussienne et des modèles de transport et d'exposition de polluants «multimédia». En plus, une introduction au modèle de logiciel intégré d'évaluation de l'impact utilisé dans l'IPA est décrite.

Finalement, la dernière section du chapitre 2, analyse le caractère des cycles de vie qui sont étudiés par l'ACV. Une différenciation entre une chaîne de procédés industriels et l'autre est faite en relation avec le nombre des procédés inclus dans chaque chaîne. Nous considérons les chaînes avec moins de 100 procédés comme des chaînes de procédés industriels qui permet une analyse d'impact environnemental à l'échelle locale et les systèmes de produits complets avec plusieurs milles de procédés qui ne permet qu'une évaluation par des potentiels d'impact. Il est démontré la pertinence de cette différenciation, dans la manière d'évaluer les impacts environnementaux.

Le chapitre 3 introduit le lecteur au cas pratique de l'incinération de déchets, en portant une attention spéciale aux impacts sur la santé humaine. Nous avons obtenu des résultats, basés sur une série de données fondamentales communes aux applications des méthodes utilisées normalement (ACV, ERA et IPA) à l'Incinérateur de Déchets Solides Urbains (MSWI) de Tarragone et sa chaîne de procédés dans deux situations: avec et sans un système avancé de traitement de gaz. En plus, sommairement, un modèle modulaire est présenté qui permet générer un scénario d'une MSWI avec un système additionnel de traitement de gaz pour éliminer des NOx (DeNOx). Il est vérifié, dans une partie de ce chapitre, que ce cycle de vie peut être définit comme une chaîne de procédés industriels avec, approximativement 20 procédés significatifs, et non pas comme un cycle de vie d'un produit complet, comme par exemple un ordinateur.

Dans le chapitre 4, commence le développement de la méthodologie. L'existence de types divers d'incertitudes est fréquemment citée comme une limitation cruciale d'une interprétation claire des estimations de dommages environnementaux. Pour cette raison, une stratégie est présentée pour l'analyse stochastique d'incertitudes par la simulation de Monte Carlo. Nous avons élaboré une méthodologie, pour ce type d'évaluation, dans le cas de la LCI et de l'IPA. La procédure est appliquée à la LCI du cas pratique et les estimations de dommages des émissions de la MSWI pour l'IPA à échelle local. L'application de la méthode dans le cas de l'IPA démontre que l'incertitude et la variabilité calculées sont moins importantes que celles publiées dans une étude avec des méthodes analytiques. L'analyse de sensibilité indique que l'impact principal sur la santé humaine ne provient ni des PCDD/Fs, ni des métaux lourds, mais plutôt des particules et des NOx, en utilisant des fonctions de dose - réponse de toxicologie et d'exposition–réponse d'épidémiologie.

L'application parallèle aux deux méthodes permet de montrer que les incertitudes dans l'analyse de l'inventaire sont moins importantes que celles dans l'évaluation de dommages. En particulier, l'utilisation des modèles de dispersion, des schémas d'évaluation et des fonctions de dose- réponse et d'exposition – réponse impliquent des incertitudes énormes.

Le chapitre 5 propose un algorithme mathématique et un schéma qui permet la différenciation spatiale à différents niveaux d'échelle et propose une intégration de l'ACV, de l'IPA et des coûts environnementaux. C'est spécialement applicable pour une chaîne de procédé industriel mentionné, mais pas pour un cycle de vie complet d'un produit. Pour cette raison, la méthodologie établie s'appelle «Estimations de Dommages à l'Environnement pour des Chaînes de Procédés Industriels». Une partie importante de cette méthodologie est son algorithme mathématique qui permet de gérer la distribution de la pollution pendant les différentes étapes du cycle de vie et l'assignation des dommages à la région correspondante d'émission. Le corps de la méthodologie est fondé sur le diagramme de flux. Nous avons élaboré diverses options pour trouver un compromis approprié entre l'évaluation d'impactes sur l'environnement, de type générique et de type spécifique du site et pour convertir les estimations de dommages environnementaux en résultats significatifs comme des coûts environnementaux.

La méthodologie pose des méthodes de points moyens (potentiels d'impact) y de points finals (dommages environnementaux) dans la LCIA dans une marque commune. De cette manière, la comparaison directe des résultats d'indicateurs de points moyens et de points finaux est rendue possible. Le schéma est divisé en parties différentes, qui d'une manière sont similaires aux phases de l'ACV, mais qui ont souvent un contenu différent. La première phase est la Définition de l'Objectif et de l'Échelle avec la sélection du schéma d'évaluation et d'agrégation que veut appliquer la personne qui prend les décisions, la seconde phase est la LCI et la troisième, la LCIA. Ici se termine la similitude avec la méthodologie de l'ACV; à la place de l'interprétation, les étapes suivantes sont l'Analyse de la Dominance et la Différenciation Spatiale, l'Analyse de Transport et d'Exposition des Polluants et l'Analyse de Conséquences ainsi que le Profil de Dommage et l'Eco-efficience. Dans la méthodologie proposée, un numéro d'outils analytiques est utilisé, comme l'ERA et des éléments techniques comme l'analyse d'incertitudes par la simulation de Monte Carlo. Optionnellement, la considération d'accidents peut être prévue. Finalement, la méthodologie est appliquée aux résultats de l'étude d'ACV de la chaîne de procédés de l'incinérateur de déchets avec pour finalité, démonter son applicabilité.

Le chapitre 6 applique et continue avec le développement d'une méthodologie pour l'évaluation d'impacts dépendant d'endroit, comme une solution pour arriver à un compromis entre les évaluations de dommage spécifique de site (au niveau des points finals) et les indicateurs potentiels du cycle de vie (au niveau de points moyens). Les raisons principales de la discordance entre les potentiels d'impact calculés et la prévision d'impacts réels dans la LCIA sont, d'une part, de ne pas considérer la distribution spatiale des récepteurs et, d'autre part, le manque d'information sur le lieu d'émission et les conditions de dispersion correspondantes dans le milieu respectif. Pour mettre en relation ces facteurs principaux, la méthode présentée utilise des classes d'impact génériques, représentatives correspondant aux distributions de récepteurs et aux conditions de dispersion. Toute cette méthode est basée sur un raisonnement statistique. La méthode existante est adaptée pour inclure les modèles les plus sophistiqués: un programme détaillé de dispersion et un système géographique d'information. Tout de même, il est clair qu'il faut plus d'efforts pour travailler dans la recherche de la définition statique des paramètres caractéristiques du point d'émission. La méthode de base existante a été généralisée à fin d'inclure plus de récepteurs. La méthode est appliquée en détail aux effets sur la santé humaine à cause des émissions dans l'air en Catalogne; nous avons calculé des facteurs au niveau départemental.

La méthode de l'analyse d'impacte dépendante d'endroit convient parfaitement avec la méthodologie développée dans le chapitre 5 et s'applique pour comparer l'importance du transport, en relation avec le procédé de l'incinération des déchets, qui était identifié comme étant d'un intérêt particulier dans l'application du chapitre 5. Nous avons pu démontrer que la différentiation spatiale est faisable dans la LCIA avec un effort raisonnable et peu d'information additionnelle nécessaire dans la LCI (nombre du département, hauteur de la cheminée, information sur la distribution de la taille des particules). La comparaison des résultats dérivés par des indicateurs d'impacte de point final avec ceux obtenus par l'indicateur de point moyen, Potentiel de Toxicité Humain (HTP) indique que le HTP sous-estime l'impact sur l'environnement des procédés de transport. C'est spécialement relevant pour les technologies de traitement de gaz les plus avancées, ceci signifie avec un système DeNOx d'élimination des NOx. Dans ce cas on peut observer que la capacité de nettoyage est fortement compensée par des efforts additionnels de nettoyage qui implique la provision de la matière primaire, incluant le transporte supplémentaire.

Dans le chapitre 7, la discussion générale est présentée. La méthodologie développée est limitée par deux aspects: les incertitudes inhérentes et son applicabilité principale aux chaînes de procédés industriels, que nous avons définies comme chaînes avec 100 procédés ou moins. Les applications les plus appropriées du système de l'évaluation des dommages environnementaux sont la gestion de fin de vie et l'optimisation de la recherche des sites de nouvelles usines industrielles par des critères environnementaux. Il est évident qu'il y a beaucoup de problèmes vis-à-vis de l'environnement dans le cas de perspectives du cycle de vie qu'on ne peut pas étudier par la méthodologie présentée, du moins pour le moment à cause de l'insuffisance de disponibilité de données et de la quantité de travail nécessaire. Basé sur cette conclusion et d'autres réflexions, une stratégie des applications génériques est proposée. Cette stratégie fait référence au développement d'une méthodologie acceptée internationalement pour établir des méthodes de LCIA embrassant points finaux et points moyens.

Dans les conclusions, chapitre 8, nous mettons l'accent sur le développement de la méthodologie comme contribution principale de la thèse. Finalement, des perspectives intéressantes sont proposées pour les recherches futures. Nous encourageons à continuer le travail sur la différentiation des méthodes de cycle de vie en respect avec son nombre de procédés. Nous concluons que la base a été établie pour la création d'une nouvelle génération de modèles de gestion de déchets intégrale qui inclue l'optimisation des lieux d'installation des usines de traitement des déchets et des routes de transports correspondantes. Cependant, il faut admettre qu'il reste encore du travail à faire pour que cette méthode soit complètement opérationnelle.

S.4 Zusammenfassung

Für die Abschätzung der Umweltschäden in industriellen Prozessketten werden Umweltwirkungsanalysen in zwei Perspektiven benötigt: prozesskettenorientiert und ortsorientiert. Für beide Perspektiven wurden spezielle Methoden entwickelt.

Ökobilanzen oder Lebensweganalyse (LCA) ist ein relativ neues analytisches Werkzeug, um das Umweltprofil eines Productes zu analysieren. Dabei wird der ganze Lebensweg des Produktes betrachtet: von der Entnahme der Rohstoffe aus dem Boden über die Weiterverarbeitung und den Gebrauch bis zur Endbehandlung des abgelegten Produktes. Über alle diese Etappen hinweg werden in der Sachbilanz (LCI) der Rohstoffverbrauch und die Emissionen in Luft, Wasser und Boden ermittelt. Der nachfolgende Schritt ist die Umweltwirkungsbilanz (LCIA), dessen Aufgabe es ist die Sachbilanz des Produktsystems zu bewerten, um dessen Umweltrelevanz besser zu verstehen.

Umweltrisiskoanalyse von Chemikalien ist ein analytisches Werkzeug, um das Risiko von gewissen chemischen Stoffen zu ermitteln, die unter bestimmten Umständen für den Menschen und die Umwelt schädlich sind. Im Vergleich mit der prozesskettenorientierten, ortsunspezifischen LCIA, ist die üblicherweise für chemische Stoffe genutzte Methode der Umweltrisikoanalyse (ERA) ortsspezifisch, denn sie wird im allgemeinen für die Gefahrensbewertung von einem Prozess an einem Ort angewandt. Die Wirkungspfadansatz (IPA) ist eine ähnliche Methode, die für die Umweltschadensabschätzung in der Form von physikalischen Wirkungsparametern entwickelt wurde. Physikalische Wirkungsparameter sind zum Beispiel Krebsfälle oder Tage mit verringerter Arbeitsleistung. IPA basiert auf den Erfahrungen, die mit der Anwendung der Umweltrisikoanalyse gewonnen wurden. Gewöhnlicherweise werden die physikalischen Wirkungsparameter in externe Kosten überführt.

Da eine gemeinsame Bemessungsgrundlage von verschiedenen Umweltschäden fehlt, ist dafür eine Wertentscheidung unausweichlich. Ein Weg um Umweltschäden in relevante Resultate für die Gesellschaft zu überführen sind externe Kosten. Durch Monetarisierung können Umweltschäden in Wirtschaftbilanzen integriert werden. Dadurch wird ein Vergleich mit anderen Kostenarten möglich, und sie könnten internalisiert werden. Hierbei ist jedoch zu beachten, dass verschiedene Monetarisierungsmethoden existieren und dass Individuen in Abhängigkeit von ihren sozio-ökonomischen Vorstellungen andere Bewertungsschemas bevorzugen mögen.

Um die Nachteile der angeführten ortsspezifischen und ortsunspezifischen Methoden zu überwinden, wurden in der letzten Zeit sogenannte ortsabhängige Methoden zur Wirkungsanalyse entwickelt. Diese Methoden brauchen ein Minimum an zusätzlichen Daten (allgemeine Ortsparameter) für die Beachtung der Ortsabhängigkeit der Wirkungen. Diese Ortsparameter sollten alle wichtigen Informationen zur Ausbreitung, Exposition und Wirkung des Schadstoffs in der Form von Indikatoren berücksichtigen. Die Indikatoren müssen für verschiedene Klassen von Emissionsorten anwendbar sein, und nicht für bestimmte Orte.

Die letzten Entwicklungen führen zu Fortschritten in der Anwendungspraxis von LCIA, aber es ist nicht anzunehmen, dass die grössten Beschränkungen und Unsicherheiten in naher Zukunft aufhören werden zu bestehen. Sie können nur durch Unsicherheitsanalysen und eine geeignete Kommunikation der Ergebnisse deutlich gemacht werden. Es wäre ein wichtiger Schritt vorwärts, wenn ein umfassender methodischer Rahmen für Endpunkte (Schadensvorhersagen) und Mittpunkte (Schadenspotentiale) geschaffen würde, der die wichtigsten Variabeln beider Ansätze einschliesst. Dadurch würde die Modellierung entprechend den Vorgaben beider Methoden und der Vergleich ihrer Ergebnisse ermöglicht. Diese Aufgabe ist wahrscheinlich nicht ohne die Integration mit anderen Umweltmanagementmethoden zu bewerkstelligen. Der Differenzierungsgrad und die Komplexheit kann von der Art der Anwendung und der Verfügbarkeit von Daten abhängig sein.

Um einen hohen Grad an lokaler Relevanz and Genauigkeit zu ermöglichen, wird in dieser Doktorarbeit zwischen zwei Arten von Lebenszyklen im Hinblick auf die Anzahl ihrer Prozesse unterschieden. Es wird eine Differenzierung zwischen industriellen Prozessketten mit bis zu 100 Prozessen und ganzen Produktsystemen mit einer viel grösseren Anzahl von Prozessen vorgenommen. Nur wenig Anstrengungen wurden bisher unternommen, um systematisch die Unsicherheiten, Überschneidungen und die Möglichkeiten für Integration und Kommunikation der genannten prozesskettenorientierten und ortsorientierten Umweltwirkungsanalysen im Fall der oben definierten industriellen Prozessketten zu untersuchen. Hierin liegt die Motivation dieser Dissertation. Die Forschungsfrage dieser Dissertation ist deshalb:

- Auf welche Art und Weise können Umweltschäden von industriellen Prozesskettern möglichst genau abgeschätzt werden, wenn gleichzeitig der Arbeitsumfang akzeptabel und die Ergebnisse kommunizierbar sein sollen?

Das heisst, das Ziel dieser Dissertation ist, wie im Kapitel 1 beschrieben, einen geeigneten Kompromiss zwischen prozesskettenorientierten und ortsorientierten Umweltbewertungsmethoden zu finden und die Umweltschadensabschätzungen in bedeutungsvolle Ergebnisse wie Umweltkosten zu überführen.

Das allgemeine Ziel führt zu den folgenden nachgeordneten Fragen:

- Was sind die Unsicherheiten in den Methoden zur Umweltschadensbestimmung?
- Was für Möglichkeiten gibt es für Überschneidungen und Integration bei den Umweltmanagementmethoden, besonders LCA, ERA, IPA und Umweltkosten?
- Wie können Mittpunkt- und Endpunkt-Methoden in einen gemeinsamen methodischen Rahmen überführt werden?

Es ist klar, dass alle diese Fragen zum dem Gebiet der Differenzierung und Detaillierung von Umweltmangementmethoden gehören. Komplexe Modelle sind notwendig um diese Fragen in der Praxis zu beantworten.

Kapitel 2 führt den Leser in die grundlegenden Konzepte und die technischen Elemente ein, die in Umweltschadensabschätzungen in industriellen Prozessketten eine Rolle spielen. Die beschriebenen Methoden werden kritisch beleuchtet. Die sogenannte Umweltmanagement-methodenkiste wird erläutert. Ihre Abhängigkeit von der jeweiligen Anwendung wird in einer Prozesskettenperspektive gezeigt; eine Anleitung wird dafür entwickelt. Es wird empfohlen, vorsichtig in der Auswahl der Methode für ein bestimmtes Problem zu sein, weil die Art des Ergebnisses der Studie in gewisser Weise durch die gewählte Umweltmanagmentmethode vorgeben wird.

Im allgemeinen schliessen alle Umweltschadensbewertungsmethoden eine Wirkungsanalyse ein, um die Schwere der Folgen der Umweltverschmutzung zu betrachten. Die wichtigsten Elemente sind die Dosis-Wirkungsbeziehungen und die Expositions-Wirkungsbeziehungen. Ausserdem kann jede Analyse auch Bewertungen der Beziehung zwischen den verschiedenen Wirkungsarten beinhalten. Eine Alternative für solche Bewertungen stellen Kosten dar, die ein wichtiger Bestandteil von Umweltmanagementmethoden sind. Ein Schwerpunkt sind dabei die externen Umweltkosten, die durch Monetarisierung ermittelt werden können. Das Verhältnis von Monetarisierung zu anderen Bewertungssystemen wird dargestellt.

Danach wird die Ökobilanzmethodik näher erläutert. Im Hinblick auf LCIA wird zunächst der allgemeine methodische Überbau dargestellt. Die verschiedenen Methoden, die in den letzten Jahren entwickelt worden sind, insbesondere solche mit einem einzelnen Bewertungsindex, werden diskutiert. Neue Entwicklungen schliessen LCIA Methoden ein, die am Schaden orientiert sind. Schliesslich werden die letzten Entwicklungen in LCIA als eine weitere Differenzierung und eine Ausweitung der Komplexheit beschrieben. Damit ist die Möglichkeit verbunden, sehr genaue und umfassende Berichte erstellen zu können, welche die möglichen Auswirkungen für jeden denkbaren Anwendungsfall untersuchen.

Im nächsten Abschnitt werden die Umweltrisikoanalyse ERA und der Wirkungspfadansatz IPA beschrieben. Während die herkömmliche Umweltrisikoanalyse die Wahrscheinlichkeit eines Schadensereignisses untersucht, bietet der Wirkungspfadansatz die Möglichkeit, die Schäden direkt abzuschätzen. Die Anwendung von Umweltrisikoanalysemethoden in dieser Arbeit verlangt den Einsatz von komplexen technischen Elementen wie einem Gaussianischen Dispersionsmodel und Multimedia-Ausbreitungs- und Expositionsmodellen, die hier beschrieben werden. Ausserdem wird eine Einführung in das integrierte Umweltschadensbewertungsmodell EcoSense gegeben, das bei der Durchführung von dem Wirkungspfadansatz angewandt wird.

Schliesslich analysiert der letzte Abschnitt von Kapitel 2 den Charakter der Lebenswege im Hinblick auf die Anzahl der Prozesse und legt die Grundlage für eine Differenzierung zwischen industriellen Prozessketten mit weniger als 100 Prozessen und einem kompletten Produktsystem. Die Bedeutung der Differenzierung für die Art und Weise der Umweltwirkungsanalyse wird aufgezeigt.

Das Kapitel 3 führt den Leser in das Fallbeispiel der Müllverbrennung unter besonderer Berücksichtigung der Gesundheitsrisiken ein. Auf der Basis des gleichen Datensatzes wurden Ergebnisse ermittelt für die Anwendung der verschiedenen üblicherweise genutzten Umweltwirkungsanalysemethoden LCA, ERA und IPA auf die Müllverbrennungsanlage (MVA) von Tarragona und ihre Prozesskette in zwei Situationen: mit und ohne weiterführende saure Abgasreinigungsanlage. Ausserdem wird ein modulares Modell kurz beschrieben, das erlaubt, ein Scenario für eine MVA mit einer weiteren Rauchgasreinigungsstufe für Stickstoffdioxidemissionen DeNOx zu erstellen. Als Teil des Fallbeispiels in diesem Kapitel wird festgestellt, das der Lebensweg der MVA tatsächlich als eine industrielle Prozesskette mit weniger als 20 wichtigen Prozessen an verschiedenen Orten definiert werden kann und dass es sich nicht um ein komplettes Produktsystem, wie z.B. einen Computer handelt.

Mit dem Kapitel 4 beginnt die Methodikentwicklung. Das Bestehen von verschiedenen Arten von Unsicherheiten wird oft als Hauptbeschränkung für die klare Interpretation von Umweltschadensabschätzungen genannt. Deshalb wird in Kapitel 3 eine Strategie für eine stochastische Unsicherheitsanalyse mit Monte Carlo Simulation präsentiert. Ein methodischer Rahmen für solche Analyse wird vorgestellt für die Sachbilanz und den Wirkungspfadansatz. Das Verfahren wird angewandt auf die Sachbilanz des Fallbeispiels und die Schadensabschätzungen für die MVA Emissionen auf lokaler Ebene durch den Wirkungspfadansatz. Die Anwendung des Wirkungspfadansatzes zeigt, dass die berechneten Unsicherheiten und Variationen geringer sind als diejenigen, die mit analytischen Methoden ermittelt worden sind. Die Sensibilitätsanalyse zeigt, dass der Hauptanteil des Umweltgesundheitsrisikos nicht von PCDD/Fs und Schwermetallen stammt, sondern von Partikeln und NOx, wenn toxikologische Dosis-Wirkungsbeziehungen und epidemiologische Expositions-Wirkungsbeziehungen benutzt werden. Die parallele Anwendung der Unsicherheitsanalysentechnik sowohl auf LCI als auch auf IPA zeigt, dass die Unsicherheiten in der Sachbilanz weniger wichtig sind als die aus den Schadensbewertungen stammenden. Besonders die Anwendung von Dispersionsmodellen, den Dosis- und Expositions-Wirkungsbeziehungen und die verschiedenen Bewertungsschema bringen eine hohe Unsicherheit mit sich.

Kapitel 5 liefert einen mathematischen Rahmen und ein Flussdiagramm, die zusammen eine Methodik beschreiben, die räumliche Differenzierung in Abhängigkeit verschiedener Detaillierungsgrade erlaubt und die teilweise eine Integration von LCA, IPA und Umweltkosten vorsieht. Insbesondere ist diese Methodik auf die so genannten industriellen Prozessketten anwenbar, d.h. nicht auf vollständige Lebenswege von Produktsystemen. Deshalb heisst die vorgeschlagene Methodik „Umweltschadensabschätzungen für Industrielle Prozessketten". Ein wichitger Teil der Methodik ist der mathematische Rahmen, der erlaubt, die Umweltwirkungen auf die verschieden Lebenswegabschnitte zu verteilen und die Schäden der entsprechenden Region, in der die Emission stattfindet, zuzuschreiben. Das Herz der Methodik ist das entwickelte Flussdiagramm, welches die unterschiedlichen Schritte deutlich macht. Verschiedene Optionen werden gegeben, um einen geeigneten Kompromiss zwichen ortsspezifischen und ortsunsspezifischen Umweltwirkungsanalysen zu finden und um die Umweltschadensschätzungen in bedeutungsvolle Ergebnisse, wie Umweltkosten, zu überführen.

Die Methodik stellt Mittpunkt- und Endpunkt-Methoden in LCIA in einen gemeinsamen methodischen Rahmen. Auf diese Weise wird der direkte Vergleich von Mittpunkt- und Endpunkt-Ergebnissen ermöglicht. Das Flussdiagramm ist unterteilt in verschiedene

Abschnitte, die teilweise Ähnlichkeit mit den Phasen einer Ökobilanz haben. Die erste Phase ist die Definiton des Ziels und des Umfangs. Sie schliesst die Wahl des Bewertungs- und Aggregationsschemas durch den Entscheidungsträger ein. Die zweite Phase ist die Sachbilanz und die dritte die Umweltwirkungsbilanz. An diesem Punkt hört die Ähnlichkeit mit der LCA Methodik auf. Anstatt der Interpretation kommen als nächste Schritte die Dominanzanalyse und räumliche Differenzierung, Ausbreitungs- und Expositionsanalyse und Konsequenzanalyse sowohl als auch Schadensprofil und ökologische Effizienz. In dem vorgestellten methodischen Grundgerüst werden ein Vielzahl von analytischen Methoden wie ERA und technische Elemente wie die Unsicherheitsanalyse durch Monte Carlo Simulation genutzt. Als Wahloption ist auch die Berücksichtigung von Unfällen vorgesehen. Schliesslich wird die entwickelte Methode auf das Fallbeispiel der MVA Prozesskette angewandt, um ihre pinzipielle Anwendbarkeit unter Beweis zu stellen.

In Kapitel 6 wird eine existierende Methode zur ortsabhängigen Umweltwirkungsanalyse benutzt und weiter entwickelt. Diese Methode kann als ein Weg gesehen werden, einen Kompromiss zwischen ortspezifischen Schadensendpunktvorhersagen und Wirkungs-mittpunktpotentialen als Lebenswegindikatoren zu finden. Die Hauptgründe für die Unterschiede zwischen den berechneten Wirkungspotentialen und den Erwartungen von tatsächlichen Wirkungen in LCIA sind auf der einen Seite die Nichtbeachtung der räumlichen Verteilung der Rezeptoren und auf der anderen Seite das Fehlen von Information über den Emissionspunkt und die entsprechenden Dispersionbedingungen im jeweiligen Medium. Um diese Hauptfaktoren zu verbinden, benutzt die vorgestellte Methode repräsentative allgemeine Wirkungsklassen, die mehreren Rezeptorenverteilungen und Dispersionsbedingungen entsprechen und auf statische Überlegungen basieren. Der existierende Ansatz wurde für Berechnungen mit komplizierteren Modellen angepasst: detallierteres Dispersionprogramm und geographisches Informationssystem. Trotz dieser ersten Anpassungen wurde klar, dass weitere Forschung notwendig ist, um die Eigenschaften der Emissionsquelle statistisch mit wenigen Parametern zu bestimmen. Die bestehende Methode wurde verallgemeinert, um mehr Rezeptoren berücksichtigen zu können. In der Studie wird sie für Gesundheitsrisiken durch Luftemissionen in Katalonien angewandt. Wirkungsfaktoren auf Landkreisniveau werden berechnet.

Die ortsabhängige Umweltwirkungsanalysemethode fügt sich perfekt in das im Kapitel 5 entwickelte methodische Grundgerüst ein und wird deshalb benutzt, um die Wichtigkeit des Transports im Verhältnis zum MVA Prozess, der sich in der Anwendung im Kapitel 5 als besonders interessant für weitere Untersuchungen erwiesen hat, zu vergleichen. Räumliche Differenzierung in LCIA hat sich mit einem akzeptablen Arbeitsaufwand und wenig zusätzlicher Information im Vergleich zur herkömmlichen Sachbilanz als möglich erwiesen. Folgende zusätzliche Information wird benötigt: Name des Landkreises, Schornsteinhöhe, Information über die Verteilung des Partikeldurchmessers. Der Vergleich der Ergebnisse, die mit den Schadensendpunktindikatoren berechnet worden, mit denen für den Mittpunktindikator Humantoxizitätspotential (HTP) zeigt, dass das HTP die Umweltschadenswirkung der Transportprozesse unterschätzt. Das ist insbesondere relevant für den Fall der am weitesten fortgeschrittenen Abgasreinigungsanlagen, d.h. mit zusätzlicher

DeNOx, in dem die verbesserte Reinigungskapazität stark durch den zusätzlichen Verbrauch von Rohmaterialien und die damit verbundenen zusätzlichen Transporte kompensiert wird.

Das Kapitel 7 diskutiert die möglichen Anwendungen und Beschränkungen der präsentierten Methodik. Sie ist zumindest im Hinblick auf zwei Gesichtspunkte beschränkt: die innewohnenden Unsicherheiten und ihre reduzierte Anwendbarkeit auf industrielle Prozessketten, die in dieser Arbeit als Ketten mit 100 oder weniger Prozessen definiert wurden. Als am besten geeignete Anwendungen des entwickelten Systems zur Umweltschadensabschätzung für industrielle Prozessketten werden empfohlen: Abfallwirtschaft und das damit verbundene Management von Produkten, die nicht mehr im Gebrauch sind, sowie die umweltwirkungsoptimierte Installation von neuen Anlagen. Es ist klar, dass eine grosse Anzahl von Umweltproblemen, auch solche mit einer Prozesskettenperspektive, im Moment nicht mit dieser Methodik untersucht werden können, denn zur Zeit sind nicht genügend Daten vorhanden vorhanden und deshalb scheint der Arbeitsaufwand zu gross zu sein. Auf der Basis dieser Schlussfolgerungen und anderer Überlegungen wird eine Strategie für generelle Ökobilanzanwendungen für Produkte vorgeschlagen. Diese Strategie bezieht sich auf die Entwicklung von einem international akzeptierten methodischem Rahmen für Mittpunkt- und Endpunkt-Methoden in LCIA.

In den Schlussfolgerungen, Kapitel 8, wird die Methodikentwicklung als Hauptleistung der Dissertation herausgestellt. Schliesslich werden interessante Perspektiven für weitere Forschungsarbeiten in diesem Bereich vorgestellt. Es wird ermutigt weitere Arbeiten über die Differenzierung von Lebensweganalysemethoden im Hinblick auf ihre Prozessanzahl in der betrachteten Kette durchzuführen. Es wird festgestellt, dass die Basis für die Schaffung einer neuen Generation von integrierten Abfallwirtschaftsmodellen geschaffen wurde, welche die umweltwirkungsorientierte Optimierung der Ortsansiedlung von Abfallbehandlungsanlagen und deren Transportstrecken ermöglicht, auch wenn noch erhebliche Arbeit zu leisten ist, bevor dieser Ansatz vollständig operativ sein wird.

Abstract

Environmental damage estimations in industrial process chains need the assessment of environmental impacts in two perspectives: process chain-orientated and site-orientated. For both perspectives environmental assessment tools exist: Life Cycle Assessment (LCA) and Environmental Risk Assessment (ERA). LCA is a fairly new chain-orientated tool to evaluate the environmental performance of products focussing on its entire life cycle. In the Life Cycle Impact Assessment (LCIA) phase a product system's Life Cycle Inventory (LCI) results are evaluated to better understand their environmental relevance. ERA is a tool to assess the risk of chemicals. In the exposure analysis the risk of a process at one location is evaluated. The Impact Pathway Analysis (IPA) is a method related to ERA that has been developed for the assessment of environmental damages by the terms of physical impact parameters like cancer cases. Usually in the IPA the physical impact parameters are converted into external environmental costs, but depending on personal values individuals may prefer other existing weighting schemes to express different types of environmental damages.

Products are manufactured in a ramified chain of processes. While specific tools exist for the environmental assessment of products and processes, this is not the case for the assessment of a number of industrial processes with a common functional unit such as end-of-life cycles. However, the level of sophistication in the assessment can be much higher for industrial process chains with a quite limited number of processes involved than for the life cycles of complex products. Only little efforts have been made so far to systematically explore the inherent uncertainties, interfaces and possibilities for integration and communication of the chain-orientated and site-orientated environmental assessment methods in the case of such industrial process chains. Therefore, the objective of this thesis is to find an adequate trade-off between process chain-orientated and site-orientated environmental impact assessment and to convert environmental damage estimates in meaningful results like environmental costs.

The thesis proposes a mathematical framework and a flowchart that allows spatial differentiation at different levels of detail based on the integration of LCA, ERA and IPA with environmental costs. This methodology called "Environmental Damage Estimations for Industrial Process Chains" puts the conventional potential midpoint LCIA indicators in a common framework with damage endpoint IPA indicators. As a trade-off between site-specific damage assessments and potential life cycle indicators a currently existing site-dependent impact assessment is further developed and integrated in the methodology proposed. The site-dependent impact assessment method is based on statistical reasoning and uses representative generic impact classes, corresponding to receptor distribution and dispersion conditions. As part of the methodology development, uncertainties in the LCI and IPA are analysed using Monte Carlo Simulation. This parallel analysis permits to show that the uncertainties in the inventory analysis are less important than those in the damage assessment.

The presented methods and the developed methodology were successfully applied in several ways to a case study on waste incineration with a special focus on human health. In a comparison of the results obtained by endpoint indicators with midpoint indicators it was found that for the situation of the case study apparently the midpoint indicators underestimate the environmental impact of the transport processes. A new generation of integrated waste management tools seems to be feasible that takes into account the setting of the waste treatment installations and the sites affected by the transport routes, allowing in this way an overall environmental optimisation.