INTEGRATED LIFE-CYCLE AND RISK ASSESSMENT FOR INDUSTRIAL PROCESSES

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Dedication

For Marc Sonnemann Riba
and the next generations

Foreword

Much of the 20th century's focus has been on economic progress, toward which humankind has made giant steps. Increasingly, unwanted side effects such as acid rain, climate change and various forms of environmentally induced toxic pollution are becoming more manifest and demanding increased attention. With the need for equal development opportunities for all, the issue of sustainability is becoming increasingly important; in my opinion, it will be one of the key issues for the 21st century. As such, sustainable development is about the welfare of human beings and a natural environment that does not reduce the possibilities of future generations, without losing sight of economic continuity of the current generation.

We are beginning to recognize that the path toward sustainability requires a life-cycle approach. The internal logic of life-cycle thinking extends the traditional focus of environmental engineering on production facilities to all stages of the value chain, which are relevant from an environmental point of view, including the production, consumption, use and waste management phases. This implies a holistic, system-analytical point of view and the cooperation between the different stakeholders throughout the life of the product.

A largely used instrument in the assessment of environmental impacts is *risk assessment* for chemical substances applied, for instance, to accident forecasting and regulatory monitoring of industrial facilities. There is a clear necessity to link this approach with the existing environmental analysis tool that applies a life-cycle perspective, *life-cycle assessment*, that is alive and well in the U.S.

This is why I enthusiastically welcome *Integrated Life-Cycle and Risk Assessment for Industrial Processes*. In this book Guido Sonnemann, Francesc Castells and Marta Schuhmacher provide not only an updated introduction to life-cycle assessment, but also to risk assessment. While they demonstrate the potential for a further integration of these two approaches, they also show the limits and constraints. I certainly agree with them on the usefulness and need for a toolbox comprised of varied environmental system analysis methods and approaches.

There are quite a lot of endeavors to be undertaken in environmental system analysis. We are still at the beginning. I look forward to our continued journey together.

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Preface

The simplistic perspective that emission sources control is a way to improve the environmental behavior of an industrial process has to be replaced by a more systematic approach that integrates all of the evaluations of environmental effects that can be assigned to a product. This book is a direct response to this challenge, developing environmental impact analysis with a global systems approach.

Life-cycle assessment is a methodology used to evaluate the environmental impacts of a product during the span of its life-cycle, and *risk assessment* is a tool to evaluate potential hazards to human health and the environment introduced by pollutant emissions. This textbook does not focus on only one specific analytical tool, such as life-cycle assessment, nor does it concentrate exclusively on risk assessment, but it looks instead at both of them, embedding them in a toolbox for environmental evaluations. Such a convergence of tools is necessary due to potential contradictory results when applying each tool separately.

In recent years life-cycle impact assessment has continuously enhanced its performance through the incorporation of elements stemming from risk assessment. It has also been increasingly embedded in environmental management practices, but until now these developments have not been reflected in the literature for graduate students, lifelong learners and interested professionals. This book aims to fill in that demand. We have identified a clear need for information provided in the form of a textbook with exercises designed for checking the understanding of the content, and a manual with flowcharts and examples for guidance.

The book covers life-cycle assessment, risk assessment and a combined framework of both for environmental damages estimations in industrial process chains and site-dependent impact assessment. The explanations of methods accompany the description of practical applications of environmental impact analysis for industrial processes.

The first four chapters of the book give a general overview of environmental management strategies, describe life-cycle assessment and risk assessment and place them in the so-called environmental management toolbox. The fifth chapter supplies additional information on techniques for data analysis that are commonly used in the analysis of environmental impacts. The sixth and seventh chapters show the interfaces between life-cycle assessment and risk assessment and provide ways of integrating the two. In the final chapter, resolved exercises of integrated life-cycle and risk assessments are presented.

The parallel presentation of both tools, life-cycle assessment and risk assessment, is a unique feature of this book. Only recently have both tools been further developed in a way that allows methods that combine both tools to become fully operational. Therefore, the exercises presented can be considered as pilot projects. As the case study on the Municipal Solid Waste Incinerator of Tarragona (Spain) develops from one chapter to another, the combined application of life-cycle assessment and risk assessment casts new light on the controversy of waste incineration with regard to human health effects.

Some specialized knowledge in environmental engineering or science is necessary to best understand the book in its entirety. Mathematical parts of the book are expressed by clearly explained equations. Figures systematically illustrate the content. Tables provide basic data and exemplary results of the case study and the exercises.

Apart from being a reference book for graduate and postgraduate students in the field of environmental management, the book intends to catalyze communication between life-cycle assessment experts and risk assessment scientists from academia, consultancies, industry and governmental agencies. In this way, the book is a manual for analyzing situations that are relevant for decision-making. The reader profits directly from the practical format of the book including flowcharts, examples, exercises and concrete applications, making the book a useful guide and facilitating the understanding of the content.

The Plan of Implementation agreed upon at the World Summit on Sustainable Development in Johannesburg, 2002 calls for the development of production and consumption policies to improve the environmental performance of products and services provided, using, where appropriate, science-based approaches; and with regard to chemicals management, a life-cycle perspective is asked for. Life-cycle assessment and risk assessment are both analytical systems approaches that allow science-based knowledge creation according to the current state of understanding of environmental mechanisms. This book can be considered as one of the first attempts to illustrate the existing interfaces between life-cycle assessment and risk assessment and to indicate options for the further integration of both tools.

In the introductory chapters, this book compiles the findings of many distinguished researchers in the fields of life-cycle assessment and risk assessment. Many findings on the basics of these tools have been integrated in recent publications from recognized international and national organizations; these include the International Organization for Standardization, the United Nations Environment Programme, the European Commission, and the United States Environmental Protection Agency.

Finally, the authors would like to take this opportunity to thank everyone who has contributed to this book both directly and indirectly, in particular, the researchers in the field of life-cycle assessment and risk assessment who have supported our work through numerous information exchanges and who are not specifically mentioned in the acknowledgments.

Guido W. Sonnemann
Marta Schuhmacher
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General Abbreviations, Symbols, and Indices

ABBREVIATIONS

AA air acidification
ADI acceptable daily intake
AE aquatic eco-toxicity
AETP aquatic eco-toxicity potential
AGTS Advanced Acid Gas Treatment System
AoP area of protection
AP acidification potential
APEA air pathway exposure assessment
APME Association of Plastic Manufacturers in Europe
AT air toxicity
ATSDR agency for toxic substances and disease registry
b_m baseline mortality
BAT best available technologies
BMD10 benchmark dose
BOD biological oxygen demand
BUWAL Swiss Federal Office of Environment, Forests and Landscape
CalTOX California Total Exposure Model
CBA cost benefit analysis
CEA cost effectiveness analysis
CHAINET European network on chain analysis for environmental decisions support
CMA Chemicals Manufacturers Association
CML Centre of Environmental Science at Leiden University
COD chemical oxygen demand
COI costs of illness
CP Cleaner Production
CV coefficient of variation
CVM contingent valuation method
DALY disability adjusted life years
DDE Dynamic Data Exchange
DEAM data for environmental analysis and management
DeNOx control device to reduce NO_x emissions
DfE design for environment
DM dematerialization
DNRR depletion of nonrenewable resources

DOL depletion of ozone layer
D–R dose–response
DT detoxification
DW disability weight
DWD Deutscher Wetterdienst
EB ecobalance
Eco-ind. eco-indicator
ECOSENSE Integrated Impact Assessment Model for Impact Pathway Analyses
ED energy depletion
ED_{10} effect dose, including a 10% response over background
EDIP environmental design of industrial products
EEC external environmental cost
EIA environmental impact assessment
EIME environmental information and management explorer
EL environmental load
ELG eco-labeling
ELV end-of-life vehicles
EMA environmental management and audit
EMAS environmental management and audit scheme
EMPA Swiss Federal Laboratories for Materials Testing and Research
EMS environmental management system
EPD environmental product declaration
EPR extended producer responsibility
EPS environmental priority strategies
Equiv. equivalent
ER excess risk
E–R exposure–response
ERA environmental risk assessment
ERV emergency room visit
ETH Eidgenössische Technische Hochschule
EUSES European Union system for the evaluation of substances
FDA U.S. Food and Drug Administration
FU functional unit
GE Greehouse effect
GenCat Generalitat de Catalunya
GIS geographic information system
GLC ground level concentration
GW Global warming
GWP global warning potential
HQ hazard quotient
HT human toxicity
HTF human toxicity factor
HTP human toxicity potential
HU hazard unit
HWP Hazardous waste production
IA Inventory analysis

IE industrial ecology
IOA input–output analysis
IP intermediate product
IPA impact pathway analysis
IRIS integrated risk information system
ISC industrial source complex
ISCST Industrial Source Complex Short Term Model
ISO international standard organization
I-TEQ international toxic equivalence
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 multicriteria evaluation
MEI maximal exposed individual
MFA material flow accounting
MHSW mixed household solid waste
MIPS material intensity per service unit
MRL minimum risk level
MSWI municipal solid waste incinerator
mUS$ 10^{-3} US$
NA not available
NOAEL no observed adverse effect level
NOEC no effect concentration
NOEL no observable effect level
NP nutrification potential
OD ozone depletion
ODP ozone depletion potential
PAF potentially affected fraction
PAH polyaromatic hydrocarbons
PCDD/Fs dioxins and furans
PE population exposure
PEC predicted environment concentration
PM particulate matter (PM_{10}: particle with an aerodynamic diameter of ≤10)
PNEC predicted noneffect concentration
POC photochemical ozone creation
POCP photochemical ozone creation potential
POF photochemical oxident formation
PP pollution prevention
PR process
PWMI European Center for Plastics in the Environment
QSAR quantitative structure activity relationship
R*Y reserve size times remaining years
RA risk assessment

RCW relative concentration weighted
RDW relative deposition weighted
RE receptor exposure
REW relative exceedance weighted
RfC reference concentration
RfD reference dose
RM raw material
RMD raw material depletion
RME reasonable maximum exposure
SAR structure activity relationship
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
SPOLD Society for the Promotion of LCA Development
ST short term
TC Technical Committee
TD tumor dose
TE terrestrial eco-toxicity
TEQ toxicity equivalent
TEAM tool for environmental analysis and management
TOC total organic carbon
TQM total quality management
TSP total suspended particulate matter
UF uncertainty factor
UNEP United Nations Environment Program
US-EPA United States Environmental Protection Agency
UWM uniform world model
VLYL value of year lost
VOC volatile organic carbon
VSL value of statistical life
WBCSD World Business Council of Sustainable Development
WD water depletion
WE water eutrophication
WHO World Health Organization
WISARD waste integrated systems assessment for recovery and disposal
WT water toxicity
WTA willingness to accept
WTM Windrose trajectory model
WTP willingness to pay
YLD years of life disabled
YOLL years of life lost
yr year

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 residence 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 insensitivity 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 modeling area
T duration
tkm transport unit equivalent to a mass of 1 t (1000 kg) transported along 1 km
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
$\mathbf{X_o}$ damage today
$\mathbf{X_t}$ damage through the years
$\mathbf{z_s}$ height above mean sea level

INDICES

1 primary pollutant
2 secondary pollutant
CR concentration response
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 product
Prod production
r receptor
RE relative exceedance
s stack
sed sediment
uni uniform
y lateral
z vertical

Table of Contents

Chapter 4 Environmental Risk Assessment

1 Basic Principles in Environmental Management

1.1 PROBLEM SETTING

Now and in the coming years, industry must play a paramount role with respect to the environment, not only as one of the main sources of environmental impact but also as one of the main actors in the proposal of new solutions. Industry-related environmental policy was originally intended to control emissions in various environments. It was widely thought that corrective technical measures at the end of the pipe would sufficiently reduce environmental impact. However, as we have seen through the years, this is insufficient in stopping progressive environmental degradation and also lacks flexibility for an evolving industry. On the one hand, it is necessary to take a quantitative leap in this approach, including the expression of risk; on the other hand, environmental considerations must be included in the entire range of industrial management. That means we must consider environmental impact within all phases of production, marketing, use and end of life once a product's life is over. Figure 1.1 presents an overview of these conceptually related methods in environmental management. They will be explained in this chapter starting with overall strategies.

1.2 OVERALL STRATEGIES

For the world to make substantial progress toward becoming a safer planet, it is necessary to introduce environmental considerations in all aspects of industrial management practices for all phases of production, marketing, use and end of life of a product. Based on these reflections, different general objectives have been formulated as programs that intend to encompass the idea of good environmental management as shown in Figure 1.1.

1.2.1 Sustainable Development

Sustainable development is understood as satisfying the needs of the present generation without compromising the needs of future generations. Sustainability takes into account three aspects:

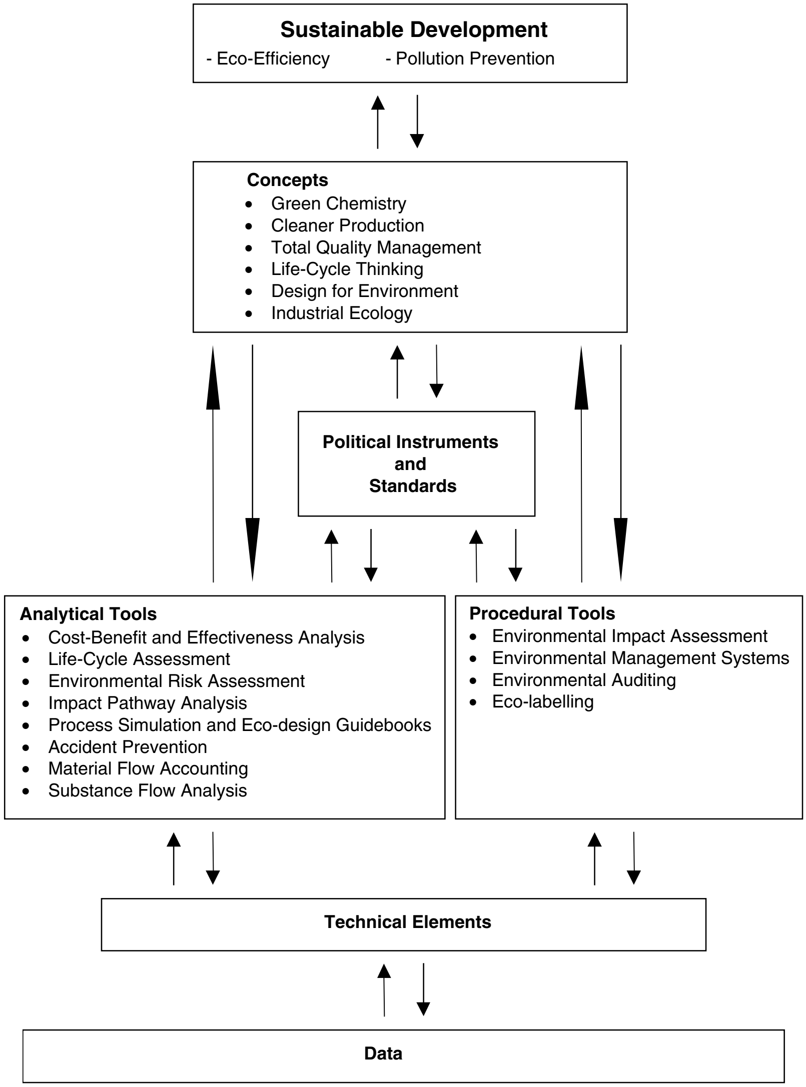


FIGURE 1.1 Conceptually related methods in environmental management. (Adapted from De Smet, B. et al., Life-cycle assessment and conceptually related programs, working group draft, SETAC-Europe, Brussels, 1996.)

1. Economic: we need economic growth to assure our material welfare.
2. Environmental: we need to minimize environmental damage, pollution, and exhaustion of resources.
3. Social: the world's resources should be shared more equitably between the rich and the poor.

Agenda 21 is a strategic document adopted by the United Nations Conference on Environment and Development (UNCED) held in Rio de Janeiro in 1992. In the Rio Summit, or Earth Summit as it is known, representatives from 179 nations gathered in what would become the end of a 2-year effort intended to define a model for sustainable development. The Earth Summit was a historical event: a new global commitment for sustainable development was established in Rio. This commitment respects the fact that environmental protection and the development process are indivisible. It is based on political commitment and global consensus at its highest level: the Agenda 21, which is an action plan for the 1990s and the early years of the 21st century. At the same time, it stands as a global alliance of humankind regarding environment and development, that is, for sustainable development. Agenda 21 is a large document divided into 40 chapters and written to foster an action plan. The goal of this project is to see that development becomes sustainable in social, economic and environmental terms.

In 2002 the World Summit for Sustainable Development in Johannesburg reviewed the implementation of Agenda 21 over the past 10 years. In 2002, the world's political situation was far different from the one that marked the Rio Earth Summit of 1992. One positive outcome was the new partnership among governments, civil society, industry and the United Nations (UN) in areas such as corporate responsibility and environmental standards. In the implementation plan, the development of a 10-year framework with programs in support of sustainable consumption and production patterns, using science-based approaches such as life-cycle analysis, has been agreed upon. It is an encouragement to industries to improve their social and environmental performance, taking into account the International Standards Organization (ISO) and the Global Reporting Initiative.

The philosophy of sustainable development has turned into a valuable guide for many communities that have discovered that traditional methods for planning and development create more social and ethical problems than the ones they solve, while sustainable development offers them real and long-lasting solutions to consolidate their future. Sustainable development makes possible the efficient use of resources, building of facilities, quality of life protection and enhancement, and the establishment of new businesses to strengthen economies. It may help in building healthy communities capable of sustaining present and future generations. Sustainability can be seen as a triangle with each of its cornerstones representing environmental, economic and social elements (Figure 1.2). Put simply, sustainability is the balance among these three elements; achieving a steady balance demands equal attention to each element.

1.2.2 Eco-Efficiency

The philosophy of eco-efficiency was first introduced in 1992 in *Changing Course,* a report by the World Business Council for Sustainable Development (WBCSD). In 1993, eco-efficiency was defined in more detail at the first workshop, held in Antwerp under the name "eco-efficiency," in which the council arrived at the conclusion that eco-efficiency is

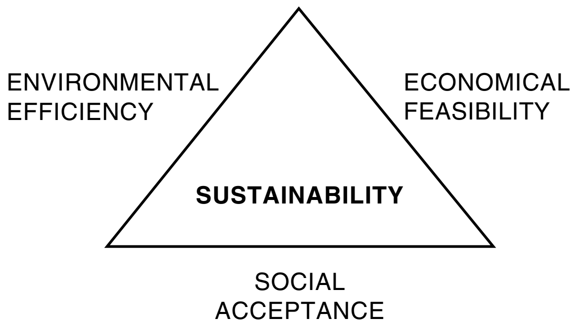


FIGURE 1.2 The three elements of sustainability.

> …reached by the delivery of competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing ecological impacts and resource intensity throughout the life-cycle to a level at least in line with the earth's estimated carrying capacity.

Based on this definition, eco-efficiency may be understood as a philosophy aimed at setting a framework for measuring the degree of sustainable development attained and for which indicators are developed. The WBCSD (1999) has identified seven actions to attain eco-efficiency:

1. Reduce material intensity of goods and services.
2. Reduce energy intensity of goods and services.
3. Reduce toxic dispersion.
4. Enhance materials' ability to be recycled.
5. Maximize sustainable use of renewable resources.
6. Extend product durability.
7. Increase service intensity of goods and services. This philosophy can be applied at a business level; for instance, Dow Chemical has developed a six-point eco-efficiency compass for its eco-innovation efforts that challenges its managers to:
   - Dematerialize to achieve reduction of raw materials, fuels and utilities in the product-service system.
   - Increase energy efficiency to determine where larger quantities of energy are consumed.
   - Eliminate negative environmental impact and reduce and control dispersion of pollutants related to the system's end of life.
   - Redesign products or their use for significant reduction of energy, material consumption and pollutant emission.
   - Close the loop by means of effective and efficient recycling.
   - Mirror natural cycles by designing the system as a part of a longer natural cycle in which materials taken from nature can be returned to it (UNEP and WBCSD, 2000).

1.2.3 Pollution Prevention

Pollution prevention (PP2) entails avoiding pollutant production right before pollutants are issued at ends of pipes, through stacks or into waste containers. The principle of prevention says that prevention is better than cure and is related to concepts such as waste reduction, waste minimization, and reduction at source. The PP2 principle may be easier to understand than to implement because establishing the boundaries between wise and unwise prevention procedures is not as easy as it may seem. This principle was created before the emergence of the philosophy of sustainable development. The waste management hierarchy list, established by the 1990 Federal Pollution Prevention Act (U.S.), may serve as reference:

1. Whenever feasible, pollution or waste should be prevented or reduced at the source.
2. If the pollution or waste cannot be prevented, reusing or recycling is the next preferred approach.
3. If the pollution or waste cannot be prevented or recycled, safe treatment must be carried out.
4. Disposal or other release into the environment should be employed as a last resort and accomplished in a safe manner.

Some examples of PP2 measures are (US-EPA, 2000a):

- Raw material replacement
- Product replacement
- Process redesign
- Equipment redesign
- Waste recycling
- Preventive maintenance (i.e., pump-end lock leaks)
- Stock minimization to prevent future wastes
- Solvent adsorption or distillation in water and later recycling

1.3 BUSINESS GOALS

In the face of social demands toward sustainable development, businesses may behave mainly in two different ways. Businesses may be content simply with watching regulations and standards or they may undertake an aggressive approach by establishing their own environmental strategies beyond mere implementation of regulations.

1.3.1 Implementation of Regulations and Standards

Administrations adopt regulations on the basis of this philosophy: each business must comply with established legislation. ISO and other regulations are not mandatory for businesses although increasingly more businesses are demanding that their providers get ISO or other certification.

1.3.2 Toward Establishment of More Ambitious Strategies than Those Given by Legislation

Establishing environmental strategies more ambitious than current legislation results in benefits for businesses as well, although very often those benefits will only be noticeable within the mid- or long run. This necessitates investing in the future. In many markets a green product sells better than a "regular" product, that is, good environmental management may also contribute to better marketing. Businesses seeking to survive in the global market must not let their competitors have a more advanced environmental policy; they must always ascertain how they stand in comparison to them (benchmarking). One of the criteria currently used by financial assessment organizations is environmental competence. Reviewing development through the past years, we may say that more technologically advanced businesses are also more environmentally competent businesses. Existing at the crest of the wave avoids losing ground and then struggling later to regain it. Future benefits include cost reduction, image enhancement and increased staff motivation. Specifically, the following advantages can occur:

- Enhancement in optimization and control of energy and raw material consumption
- Optimization in costs derived from waste and emission management and treatment
- Expense reductions in transportation, storage and packing
- Savings in environmental cleaning and repair tasks derived from accidental leaks
- Reductions in insurance policies for environmental risk
- Better conditions in negotiating bank loans
- Savings in fines from violations of law
- Decreases in accident risk and therefore in derived costs
- Enhanced business image with clients, administration, staff, investors, media, environmental defense organizations and the general public
- New marketing tool by using an environmental compliance label
- Adoption of an active policy before current legislation and future environmental regulations affect the business
- Increased possibilities to obtain public funding for environmental activities
- Involvement of staff in a system aimed at achieving common goals
- Increased training for staff

1.4 CONCEPTS

The previously mentioned philosophies and businesses goals to establish environmental strategies are reflected in and can be applied to various concepts.

1.4.1 Green Chemistry

Green chemistry is the use of chemistry for PP2. In more detail, it is aimed at designing chemical substances and, at the same time, production processes respectful of the environment. This includes reducing or eliminating use and production of dangerous substances. The concept of green chemistry was coined in 1995 and encompasses all aspects and types of chemical processes such as synthesis, catalysis, analysis, monitoring, reaction separators and conditions; PP2 minimizes their negative impact on human health and the environment. Therefore, this concept emphasizes the U.S. law concerning chemical substances (US EPA, 2000b).

1.4.2 Cleaner Production

The concept of cleaner production (CP) was first introduced by UNEP-IE (United Nations Environment Program, Division of Industry and Environment) in 1989. CP means continuous use of a preventive and integrated environmental strategy. This concept is applied to processes, products and services to increase eco-efficiency and reduce risks to population and the environment. It is intended to preserve energy and raw materials, as well as to eliminate toxic wastes and reduce the amount and toxicity of all emissions and wastes generated in every process. CP demands attitudes different from current ones in order to undertake responsible environmental management in the creation of adequate national policies and assessment of technology options. Currently, administrations tend to demand the use of best available technologies (BAT). This strategy helps reduce costs as much as it does risks, as well as to identify new opportunities. The aim of CP is to avoid pollution before it is produced in every process or in the corresponding process chains (UNEP-IE, 2000).

Although eco-efficiency is based on aspects of economic efficiency that have environmental benefits, cleaner production is based on aspects of environmental efficiency with economical benefits. Therefore, UNEP and the WBCSD are fostering similar concepts and have decided to join efforts. This new initiative combines UNEP interests within the public sector with participation of the industrial sector at the WBCSD. Their first joint action took place at the annual meeting of UNCSD (United Nations Commission on Sustainable Development) in April and May, 1996, in New York, under the title of Eco-Efficiency and Cleaner Production: Charting the Course to Sustainability (UNEP and WBCSD, 2000).

1.4.3 Total Quality Environmental Management

Total quality environmental management (TQEM) derives from quality activities and describes processes following ISO 9000 and other standards. A few of the many businesses that have implemented total quality management (TQM) include Ford Motor Company, Phillips Semiconductor, SGL Carbon, Motorola, and Toyota Motor Company. The following information helps to understand the key elements in the TQM process:

- Total = each single person and activity within a business is involved.
- Quality = satisfying client demands.

- Management = quality can and must be managed.
- TQM = a process of quality management that must be implemented in a continuous manner and with the philosophy of permanent enhancement regarding every single activity.

Although ISO 9000 stands as a series of rules for quality assurance, TQM stands as a concept for continuous improvement (Hansen, 2000). The environmental equivalent to the ISO 9000 standards for TQM is the ISO 14000 series of standards applicable to the various phases of environmental management. TQEM means applying TQM to the environmental issues of a business. This is obviously a comprehensive approach because it encompasses the business as a whole plus its management system (Vasanthakumar, 1998).

1.4.4 Life-Cycle Thinking

Life-cycle thinking is a way of addressing environmental issues and opportunities from a system or holistic perspective and evaluating a product or service system with the goal of reducing potential environmental impacts over its entire life-cycle, as illustrated in Figure 1.3. The concept of life-cycle thinking implies linking individual processes to organized chains starting from a specific function. According to this type of thinking, everyone in the entire 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 of this product on the environment and try to reduce it as much as possible. The impacts of all life-cycle stages need to be considered when making informed decisions on production and consumption patterns, policies and management strategies (UNEP, DTIE, 1999).

Many people refer to life-cycle management emphasizing "end of life management." A particular subject area for end-of-life management is the study of the limits of recyclability, whose environmental benefit disappears when the energy, materials and pollution involved in the collection, production and recycling processes exceed those necessary to produce the product. Extended producer responsibility (EPR) is one aspect of life-cycle thinking. This principle was introduced by Thomas Lindhqvist, from the Swedish Ministry of Environment, in two reports in 1990 and 1992, when the first world seminar on EPR was organized in cooperation with UNEP-IE (IIIEE, 2000):

> Extended producer responsibility is an environmental protection strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer of the product responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product. The extended producer responsibility is implemented through administrative, economic and informative instruments. The composition of these instruments determines the precise form of the extended producer responsibility.

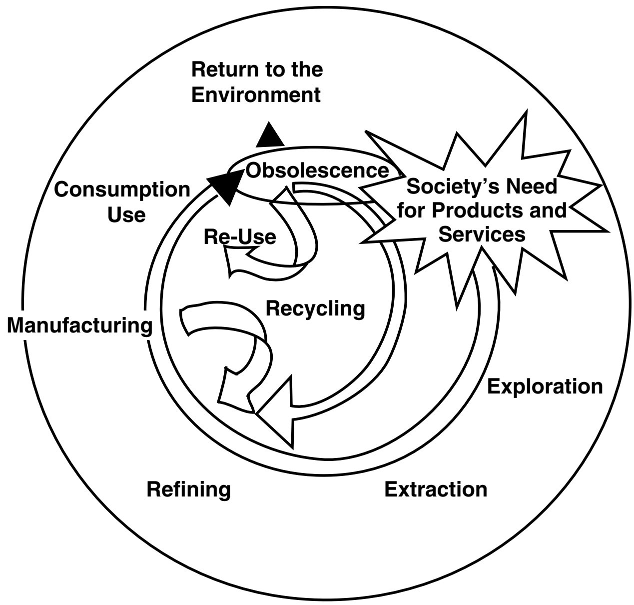


FIGURE 1.3 The life-cycle of a product system. (UNEP/SETAC, Background paper of the UNEP/SETAC life-cycle initiative, UNEP DTIE, Paris, 2001.)

At least two similar principles exist: responsible care and product stewardship. Responsible care is an initiative undertaken by the U.S. Chemical Manufacturers Association (CMA) in 1989 that commits every CMA member not only to improving health, security and environment protection, but also to responding to concerns from the public about their products. The product stewardship code is designed to change health, security and environment protection into an integral part of design, manufacturing, distribution, use, recycling and disposal. This code reinforces sharing information about adequate product use, storage and disposal. It has also been designed to include the widest possible extent of the trade chain to help prevent inadequate use likely to harm human health and the environment (SOCMA, 2000).

1.4.5 Design for Environment

Eco-design, or design for environment (DfE), allows approaching environmental problems associated with a given product within its phase of design. That is, it implies considering the environmental variable as one of the many product requirements in addition to the conventional design goals: cost, utility, functioning, security, etc. The purpose of DfE is to manufacture products with a lower global environmental load associated with their life-cycle (material/component parts acquisition/purchase, production, distribution, use and end of life). Implementation of the environmental variable in the manufacturing process of the product must be undertaken without sacrifice of the remaining product properties and combining price and quality in a sound manner (Fiksel, 1997).

Eco-design is intended to be the dematerialization and detoxification of the design process. It has to do with taking off the highest possible amount of materials — specifically, dangerous substances — without the product's losing its function (Fiksel, 1997; Brezet and Hemel, 1997; Rodrigo and Castells, 2002). Because a product life-cycle is decided during the design and production planning phase, influencing it in further phases is much more difficult.

1.4.6 Industrial Ecology

Industrial ecology (IE) entails an approximation of industrial systems to natural systems. It deals with the systematic analysis of material and energy flows in industrial systems with the purpose of minimizing waste generation and negative environmental effects (Graedel, 1994).

IE provides a holistic approach (similar to that of life-cycle assessments [LCAs]) of industrial systems, based on its analogy with natural systems: wastes produced by a living being become a source of raw material for another being. Industrial ecology may be defined as a network of industrial systems that cooperate in reusing waste energy and material within the same network; a waste flow from one of the industrial members becomes a source of raw material useful for another member. This industrial approximation is mainly associated with the concepts of eco-park, industrial symbiosis, and industrial clustering. Close cooperation among different industrial systems provides each member with higher efficiency levels in its industrial activity at the same time that it provides a higher level of use of its waste energy and material flows. This cooperation renders significant environmental and economic benefits for each member.

A specific aspect of IE is the concept of industrial metabolism proposed by Ayres in 1989; this concept consists of the process through which energy and materials flow through industrial systems, starting from the source through various industrial processes to the consumer and to final disposal (Ayres, 1996). Although IE is the most comprehensive concept of those presented until now, it is based on the idea that energy and materials flow within local, regional or global systems; however, systems encompass several processes and are not exclusively intended for a given product or service. According to the International Society for Industrial Ecology (http://www.yale.edu/isie/), IE asks how an industrial system works, how it is regulated, and how it interacts with the biosphere. Then, on the basis of what is known about ecosystems, the goal is to determine how the system could be restructured to make it compatible with the way natural ecosystems function. This way it also includes socioeconomic issues related to sustainable development.

1.5 TOOLS

The preceding concepts, which have been developed to direct environmental management, are quite abstract. This is why we need tools to transfer them into action and make environmental aspects more concrete, taking into account 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 tools and analytical tools.

In general, to be applied, the tools need technical elements like dispersion and other pollutant fate models, damage functions, weighting schemes, and available data, e.g., about emissions and resource consumption, as well as technical specifications and geographic location. The application of these tools provides consistent environmental information that facilitates adequate decision-making toward sustainable development. An overview of the conceptually related methods in environmental management is presented in Figure 1.1. 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 European Union Environment and Climate Program. The mission of this action was to promote the common use of the different tools and to facilitate information exchange among the relevant stakeholders (CHAINET, 1998).

1.5.1 Political Instruments

Generally, political instruments are adopted by political administrations. We will present typical samples of laws for a given activity to environmental management that are related.

1.5.1.1 Law on Chemical Substances

Within the framework of green chemistry and chemical substances, we could mention regulations such as the European Parliament and Council Directive 1999/45/EC, May 31, which serves as a means to homogenize standards among EU member countries concerning classification, packing and characterization of dangerous substances (Federal Environment Agency Ltd. of Austria, 2000).

1.5.1.2 Law on Process Security

An example is the National Industrial Security Program EO 12829 of the U.S. This order establishes a National Industrial Security Program to safeguard federal government classified information that is released to contractors, licensees, and grantees of the U.S. Government.

The Chemical Security Act (S.1602) of the U.S. was designed to protect communities from terrorism and accidents involving hazardous industrial chemicals. This bill establishes the first national effort to reduce industrial chemical hazards that endanger nearby neighborhoods, schools, hospitals, senior centers, or other public and business areas. It was foreseen that in September 2002, the U.S. Senate would take up the Chemical Security Act as an amendment to the homeland security bill. The Chemical Security Act, S. 1602, introduced by Senator Corzine (D-NJ), would require high-priority facilities to conduct vulnerability and hazard assessments and develop plans for improving site security and reducing chemical hazards (www.cpa.com/teampublish/uploads/S1602.pdf).

1.5.1.3 Law on Integral Intervention of Environmental Management

Directive 96/61 EU on Integrated Pollution Prevention and Control binds businesses to:

- Achieve high-level human and environmental protection by establishing an administrative intervention system applicable to those activities that may affect human security and health as well as the environment
- Integrate all current licenses regarding emissions, wastes and air, and those related to fire prevention, critical accidents and health protection, into a sole license
- Ensure the public's access to environmental information

This intervention system is based on the simplification of administrative procedures to grant businesses environmental licenses as necessary, making involved administrations co-responsible for granting licenses.

1.5.1.4 Environmental Policy Oriented to Product Recovery or Collection at End of Life

Adopted July 12, 1996, the European Parliament and Council Directive governs the management of end-of-life vehicles (ELV) that EU member states must implement with adequate measures to assure manufacturers comply with all or a significant part of the costs resulting from the treatment of end-of-life vehicles, or collect them following the conditions stated in the directive.

1.5.1.5 Advance of Costs for End-of-Life Management

EU-directive on packaging and packaging waste: European Parliament and Council Directive 94/62/EC of 20 December on packaging and packaging waste.

The Packaging Waste Regulations came into force on March 6, 1997. These new regulations are the U.K.'s approach to meeting the recycling and recovery targets set out in the EU Directive on Packaging and Packaging Waste. The aim is to recover 50% of the 8 million tonnes of packaging waste produced in the U.K. It is estimated that the cost is likely to be 0.02 U.S. $ per 10 U.S. $ of every shopping bill. Businesses are only affected if they fall into three or four of the following categories (EU directive on packaging waste, 1994):

- Manufacture raw materials used for packaging
- Convert raw materials into packaging
- Pack or fill packaging
- Sell packaging or packaged products to the final user or consumer
- Own and handle more than 50 t of packaging materials or packaging each year (including imported packaging materials and goods, but not exports)
- Annual turnover of more than 3.2 million (from January 1, 2000)

1.5.1.6 Special Protection Plans for Specific Areas

The National Oceanic and Atmospheric Administration's National Coastal Zone Management (CZM) Program (http://www.nos.noaa.gov/ocrm/czm/) is a voluntary partnership between the U.S. government and U.S. coastal states and territories. It is authorized by the Coastal Zone Management Act of 1972 to preserve, protect, develop and, where possible, restore and enhance the resources of the nation's coastal zone.

1.5.2 International Standards

The International Standards Organization (ISO) was founded in 1946 in Geneva, Switzerland. ISO has established nonmandatory international standards for the manufacturing, communication, trade and administrative sectors. It has also created ISO 9000 for management and quality assurance; this has been adopted by more than 90 countries and implemented by thousands of industries and service providers. Accordingly, for environmental management, ISO has created the ISO 14000 series, a new generation of standards to foster national and international trade in compliance with international standards to protect the environment. In this way, some common guidelines and similarities between environmental management and business management are established for all businesses regardless of size, activity or geographical location. Several of the ISO 14000 standards refer to the previously mentioned procedural and analytical of tools. The five groups of environmental standards governed by the ISO 14000 series are:

- ISO 14001–04: Environmental management systems — general guidelines on principles, systems and supporting techniques
- ISO 14010–14012: Guidelines for environmental auditing
- ISO 14020–14024: Environmental labels and declarations
- ISO 14031: Environmental performance evaluation — guidelines
- ISO 14040–14043: Life-cycle assessment

These standards will be reflected in some of the procedural and analytical tools explained later.

1.5.3 Procedural Tools

Important procedural tools described next are the environmental impact assessment, environmental management system, eco-audit and eco-labels. Some procedural tools come from current legislation and others from international standards. The latter can be applied at process, product and service levels.

1.5.3.1 Environmental Impact Assessment

As stated in Council Directive 85/337/EU, modified by Council Directive 57/11/EU, environmental impact assessment (EIA) is a set of research papers and technical systems used to estimate the effects on the environment of implementing a given

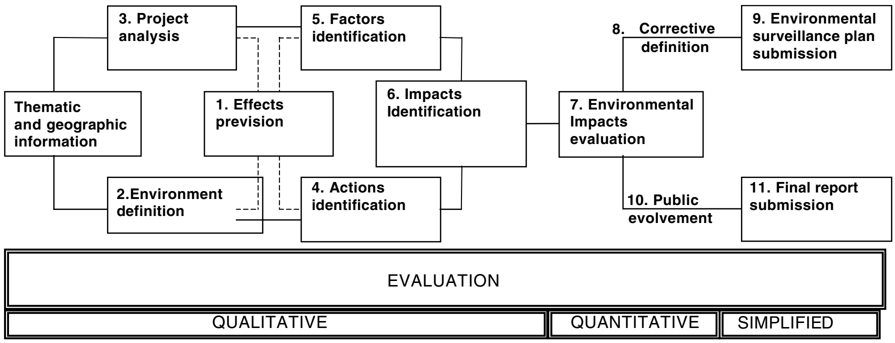


FIGURE 1.4 General structure of an environmental impact assessment.

project, work or activity. Thus EIA is an analytical procedure oriented to determine objectively the consequences of impacts derived from a given activity on the environment. The idea behind EIA is to obtain an objective judgment of the consequences due to the impacts generated by accomplishing a given activity. The main part of such an evaluation is the environmental impact study (Coneza, 1997).

Environmental impact research is a technical interdisciplinary assessment aimed at foreseeing, identifying, determining and correcting the environmental impact or consequences that certain activities may have on the quality of human life and the environment. It has to do with presenting objective reality to determine the influence on the environment of implementing a given project, work or activity. In sum, EIA is an analytical tool fundamental for information gathering and necessary to submit an environmental impact declaration.

The different phases of EIA are summarized in Figure 1.4. The first six phases are related to qualitative assessment. During Phase 7 a quantitative assessment is carried out, which partially continues during Phase 8 and Phase 9; in Phase 10 and Phase 11 more simplified results are produced. The first nine phases are related to the environmental impact study (Coneza, 1997).

1.5.3.2 Environmental Management System

The environmental (or eco-)management and audit scheme (EMAS) is an EU-based system related to Council Regulation (EEC) 1836/93 for the continuous improvement of environmental aspects in businesses. Internationally it corresponds in many features to ISO 14001, although the latter does not have the same recognition as that of the environmental authorities (Zharen, 1995).

An environmental management system (EMS) is a means of ensuring effective implementation of an environmental management plan or procedures in compliance with environmental policy objectives and targets. A key feature on any effective EMS is the preparation of documented system procedures and instructions to ensure effective communication and continuity of implementation. There are certification systems for EMS as the ISO 14001 and EC EMAS scheme (EMAS is now compatible with ISO 14001), which demonstrate that a system is operated to an internationally recognized standard. Alternatively, a customized system can be developed addressing the particular needs of the operation (EC, 1999).

An environmental management system allows businesses to:

- Assure a high level of environmental protection
- Continuously improve their environmental performance
- Obtain competitive advantages out of these improvements
- Communicate their progress with the publication of an environmental declaration showing their efforts

The environmental management department is a recognized instrument in all EU member states, based on Council Resolution 1836/1993 as of June 29, 1993, which allows industries nonmandatory adherence to an EU environmental management and audit scheme. EMAS has been developed for organizations involved in industrial activities, energy generation, recycling and solid and liquid waste treatment. Additionally, it can be applied to other sectors such as energy, gas and water supply, construction, trade, transportation, financial services, public administrations, entertainment, culture, sports, education and tourism.

At present on the international scene, the ISO 14001 standard about environmental management systems is mostly used. This standard is not against that established in EU Resolution 1836/1993 and can be seen as a previous step for EMAS adherence. As a standard with international application, the ISO 14001 has a more general nature. In Europe EMAS enjoys official recognition on the side of political administrations.

1.5.3.3 Eco-Audit

An eco-audit, or environmental audit, is an "independent and methodical test carried out to determine whether the activities and results concerning the environment meet previously established regulations and prove to be adequate for attaining the foreseen goals." As explained in the ISO 14010 standard, eco-audit is "a process of systematic testing and objective assessment of evidence to determine whether environmental activities, events, conditions and systems, or information about these, conform to audit criteria and communication of results to customers."

Eco-audits are carried out to implement environmental management systems. Audits help acknowledge the current business position in the face of existing legislation. Therefore, the eco-audit may be defined as a management tool to test whether activities and results related to the environment are accomplished, that is, established goals attained and standards met, and if the latter are adequate to attain those goals. It is an important tool to enhance environmental management and has a preventive nature; thus it is not an inspection and control activity or a witch hunt, but is intended for problem detection and solution. Environmental audits are generally carried out to accomplish one or several of these goals (Umweltbundesamt, Germany [1998]):

- Determine suitability and effectiveness of an organization's environmental management system to attain environmental goals.
- Provide the audited organization with an opportunity to enhance its environmental management systems and, as a result, contribute to continuous improvement of its environmental performance.
- Check conformity to existing regulations.
- Internally assess the organization's environmental management system within the framework of a given environmental management standard.
- Assess an organization prior to establishing a contractual party relationship with that entity.

1.5.3.4 Eco-Label

The idea of eco-labeling is to guarantee the environmental quality of certain properties or characteristics of the products that obtain the eco-label (Alfonso and Krämer, 1996) in order to provide consumers with better information on green products and promote the design for environment (EC, 1997). An EU eco-label scheme is laid down in Council Regulation 880/92 (EEC) 1980/2000.

Products and services that meet previously established environmental criteria are allowed to use various official labels for easier recognition. On the one hand, the eco-labeling scheme provides consumers and end-users with enhanced and more reliable information; on the other hand, it fosters design, manufacturing, marketing, use and consumption of products and services exceeding existing mandatory environmental quality requirements. A product with a lower environmental impact is that with a composition and/or manufacturing, operation, elimination that cause lower damage or impact to the environment. Some examples are paper manufactured without emission of organic chloride compounds, a washing machine with low water and energy consumption, a refrigerator manufactured with recyclable component parts, etc. A service with a lower environmental impact would be a small business (store, repair shop, etc.) whose operation, management or service supply (from the use of environmental quality products to adequate waste management) is respectful of the environment. Examples of other eco-labels are the Blue Angel in Germany and the White Swan in the Nordic countries.

The status of eco-labeling and product information on the international scene is far from coherent, however, particularly among the various stakeholders. At the global level, including food and nonfood products, we can count on 700 labels and 2000 green claims; only 17 of these green claims are part of a systematic eco-labeling scheme. These cover product categories ranging from laundry detergents, household cleaners, paints and varnishes, household paper, sanitary items, wood, textiles, white domestic appliances, and garden products, as well as tourism, energy production or efficiency and services. ISO classified labels or indicators for environmental claims use into three categories:

1. National eco-labels (also called “Label Type I” in ISO 14024); examples: EC eco-label, Nordic Swan and Blue Angel
2. Self-environmental declarations (also called “Label Type II” in ISO 14021); examples: ozone friendly label, green dot and animal cruelty free
3. Indicators based on life-cycle assessments (also called “Label Type III” in ISO/TR 14025); only example: environmental product declaration (EPD) promoted by the governments of Sweden, Norway, Canada, Korea and Japan, and companies such as Volvo and ABB

The related environmental product declarations gain popularity as tools, especially for business-to-business communication, and have great potential to be used widely by institutional buyers in their efforts for green procurement. In general, for the communication to consumers, one overall environmental Type I label based on a single indicator is considered the most effective option to influence consumer choices. However, consumers are likely to be interested in more detailed environmental information for durable goods such as cars or electronics; a Type III label might be provided to influence the purchase for these types of items.

1.5.4 Analytical Tools

The following analytical tools are relevant methods for environmental management:

- Life-cycle assessment (LCA) is a tool standardized according to ISO series 14040 for product-oriented environmental impact assessment and will be further explained in Chapter 2.
- Environmental risk assessment (ERA) and impact pathway analysis (IPA) are the tools generally used for the impact analysis in site-specific environmental impact assessment. These tools will be described in more detail in Chapter 3.
- Cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA) are technoeconomic tools to support decision-making towards sustainability. They refer to environmental costs, a topic explained in Chapter 3.
- Process simulation (and the related re-engineering) is an important tool for the improvement of industrial processes. It allows foreseeing environmental effects resulting from changes in process design before implementation.
- Accident prevention requires determining the environmental risk that implies installation and operation of an industrial process due to undesirable events. Undesirable events are caused by unforeseen emissions of pollutants for accidental reasons.
- Material Flow Accounting (MFA) and Input–Output Analysis (IOA) have been developed to look at the life cycle of material substances in industrial systems and the environment.

Analytical tools differ depending on the specific aspect of their focus; few of them have been standardized by ISO.

1.5.4.1 Life-Cycle Assessment (LCA)

In order to consider environmental impacts of a product's life-cycle systematically, the life-cycle assessment (LCA) methodology has been developed. This is the only standardized tool currently used to assess product environmental loads. The steps of LCA are: goal and scope definition, inventory analysis, impact assessment, and interpretation.

The goal, the motivation for research, must be clearly defined from the very beginning because the following phases will be influenced by its early definition. The creation of life-cycle inventories (LCIs) is intended to identify and assess the environmental load associated with the full life-cycle of a product, process or activity. In the case of a product, the inventory starts at the extraction process of raw materials from the environment, continues in the production, consumption and use of end products, and ends when the product or its derivate turns into waste. Operations such as transportation, recycling, maintenance, etc. must also be considered in the inventory.

Life-cycle impact assessment (LCIA) allows for easier interpretation of environmental information produced during the inventory analysis phase. LCIA includes several phases:

- Classification of environmental loads within the different categories of environmental impact
- Categorization of environmental loads by means of a reference pollutant typical of each environmental impact category
- Normalization of the data obtained from the characterization, dividing it into real or foreseen magnitude for its corresponding impact category within a geographical location and a point in time for reference
- Quantitative or qualitative assessment of the relative significance concerning different categories of impacts

These categories, plus the level of detail and methodology, are chosen depending on the goals and scope of the research.

Following this analysis, more objective and transparent decisions can be made concerning environmental management for the creation of guidelines for new product development and guidebooks to define environmental priorities (SETAC, 1993). See Chapters 2 and 3 for more details.

1.5.4.2 Environmental Risk Assessment (ERA)

The usual point in introducing risk assessment is to emphasize that risk is part of everything we do and that the risk derived from pollutant exposure should be paid the attention it deserves. In the U.S. during the late 1980s, about 460,000 of 2.1 million deaths per year were due to cancer. Without taking the age factor into consideration, the risk of dying from cancer equals 22% (460,000/2,100,000 = 0.22). Individuals who smoke one package of cigarettes per day have approximately a 25% risk of dying from heart disease. Meanwhile, the U.S. Environmental Protection

Agency (U.S. EPA) intends to control exposure to toxic substances with risk levels ranging from 10^{-7} to 10^{-4} (0.00001 to 0.01%) throughout life (Masters, 1991).

Environmental Risk Assessment is a process for determining the probability for negative effects on human health or the environment as a result of exposure to one or more physical, chemical or biological agents. ERA requires knowledge about the negative effects of exposure to chemical substances or materials, as well as knowledge about the intensity and duration necessary for these to cause negative effects on population and the environment. Decision-making within sound risk management entails examining the various choices for risk reduction. The risk assessment scope is generally local; environmental impacts are presented in the form of risk per researched recipient because it is the case with the value for exposure to toxic substances at levels entailing a risk ranging from 10^{-7} to 10^{-4} throughout life.

Environmental risk assessment will be explained in detail in Chapter 4.

1.5.4.3 Impact Pathway Analysis (IPA)

Contrary to this approach, the Impact Pathway Analysis (IPA) estimates the overall damage, including even relatively small contributions, in locations more than 1000 km away from the source of emissions. The damage is presented in the form of costs, but because these costs are not included in the price of the product or service responsible for the emissions, they are known as external costs.

An example of an IPA scheme is shown in Figure 1.5. After the location and technology producing the pollutant have been selected, Phase 1 deals with the calculation of emissions and the plant's demand for resources. Phase 2 concerns the distribution of emissions to several recipients. The following phases consider the transition processes and impact assessments, for example, a bad fruit crop or higher incidence of asthma attacks. Once all physical impacts have been calculated, economic assessments are applied to the impacts to estimate damages in currency amounts (euros, dollars, yens). Impact pathway analysis is explained as a special form of ERA at the end of Chapter 4.

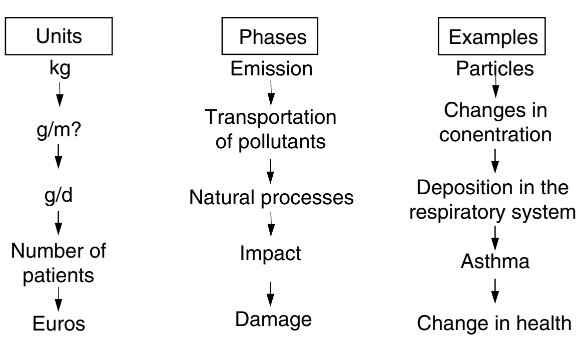


FIGURE 1.5 Impact pathway analysis (IPA) scheme. (Adapted from European Commission — DGX11), *ExternE – Externalities of Energy*, ECSC-EC-EAEC, Brussels-Luxembourg, 1995.)

1.5.4.4 Cost-Effectiveness Analysis (CEA) and Cost-Benefit Analysis (CBA)

An efficient emission level is the point at which marginal damages (external costs) are equivalent to marginal abatement costs (internal costs), i.e., both types of costs are neutralized. Again, this point is the efficient level of emissions, a consideration illustrated in Figure 1.6. Because marginal damage costs and marginal abatement costs are equal, both hold the W value at that level of emissions. Marginal damages have their threshold at emission level E, while the uncontrolled emission level is at E_d. It is possible to analyze the results in terms of total values because we know the totals are the areas under marginal curves. Thus, the triangular area distinguished with an *a* (marked by points E_u, E_e and the marginal damage curve) schematizes the existing total damages when emissions are at level E_e, while the *b* area shows a reduction of the total costs at this level of emissions. The addition of both areas is a measurement of total costs of E_e — tones per year in this case. E_u is the only point at which this addition is minimized, but the *a* area is not necessarily equal to the *b* area; this depends on the shape of the marginal curves, which may have wide variations (Field, 1995). The preceding explanations serve as a basis to introduce the cost-effectiveness analysis (CEA) and the cost-benefit analysis (CBA) methods.

CEA considers only the internal costs, i.e., the costs resulting from emission reduction technologies. These costs are compared to the reduction of the environmental load due to the economic investment. If we want to invest this money to enhance the environment we will invest it in the most effective choice.

CBA is another economic tool intended to provide decision-making support for long-term investments from a social perspective instead of the business perspective. The field for its application includes the environmental selection of technologies and legislation strategies. CBA is intended to correct dysfunctions caused by market defects. In the sphere of environmental management, the main interest is that external effects be considered external costs. The conversion of damages into costs is based

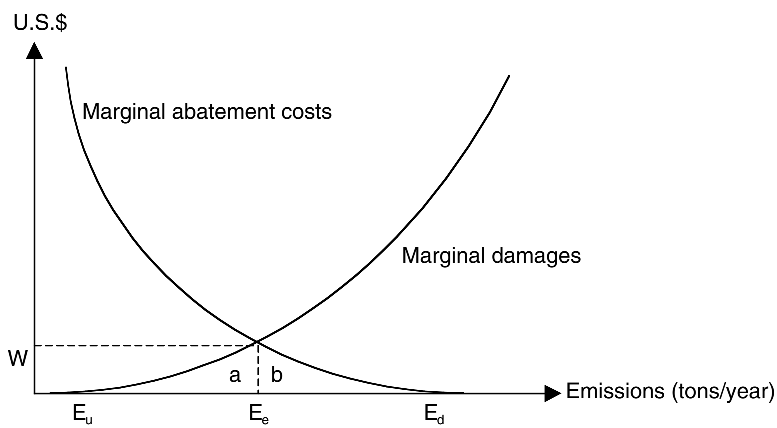


FIGURE 1.6 Efficient emission level.

generally on Paretian's theory of well-being, in which individuals facing external effects judge their importance for their quality of life. Preferences may be shown in monetary terms, just as in market choices. In this way, the final assessment of the marginal damage balance (external costs) and marginal abatement costs (internal costs) may be carried out in one monetary unit: currency (dollar, euro, yen), which is a globally accepted reference in decision-making (Dasgupta and Pearce, 1972; Nas, 1996).

1.5.4.5 Process Simulation

Process re-engineering is one of the applications within chemical engineering that shows increases in interest year after year. The vast majority of plants have been built and are currently operating so as to obtain the highest possible return using available equipment and reduce investment. New process synthesis tools allow analysis, in a relatively simple and fast way, of whether the optimum current process setting is the best and to assess possibilities for improvement. To attain this, it is necessary to count on models with an exact and precise representation of equipment operation. In the first place, it is necessary to adjust these models using plant data to ensure successful predictions. After the base case has been obtained, exploring alternative design choices and assessing their applicability through equipment sizing begins. At this point, it is necessary to assess in detail the possibilities for implementing the proposed alternative by means of detailed research of the most promising alternatives (detailed sizing, economic estimate, cost analysis, etc.). Apart from these applications, process simulation has proved to be the fundamental tool for detecting bottlenecks in processes and operation problems, and in obtaining ideal operating conditions for the environment.

15.4.6 Accident Prevention

To prevent accidents, it is necessary to determine the environmental risk associated with a facility or process operation resulting from undesirable events or accidents. Undesirable events are those unforeseen events that accidentally cause the emission of pollutants into the environment. Analysis of undesirable events (accidents) must include the following phases:

- Analysis of facilities and information gathering
- Identification of the most representative accident scenarios
- Establishment of accident probability for these scenarios and its evolution
- Establishment of consequences associated with each accident scenario assessed (scope of associated physical effects)
- Establishment of the impact these physical effects have on the environment

Fail trees and event trees are applied when accident prevention assessments are undertaken (AICHE, 1985).

1.5.4.7 Material Flow Accounting (MFA) and Input-Output Analysis (IOA)

Material Flow Accounting (MFA) refers to accounting in physical units (usually in tons); the extraction, production, transformation, consumption, recycling and deposition of materials in a given location (i.e., substances, raw materials, products, wastes, emissions into the air, water or soil). Within the range of the present work, MFA encompasses methods such as substance flow analysis (SFA) and other types of balance of materials for a given region (Fuster et al., 2002). Examples of flow assessments are:

- Eco-toxic substances such as heavy metals that may cause environmental problems due to their accumulation capacity
- Nutrients such as nitrogen and phosphates due to their critical influence over eutrofication
- Aluminum, the economic use, recycling and reuse of which are to be improved (Bringezu et al., 1997; Fuster et al., 2002)

As a part of establishing statistical accounts on a national scale, the input–output analysis (IOA) has been under development since the 1930s. One of the main applications of this analysis is to show the interrelationship of all flows of goods and services within a given economy; it also shows the connection between producers and consumers and interdependence among industries (Miller and Blair, 1985). Since 1993, different environmental applications have been designed. Nowadays, this macroeconomic method is frequently applied to environmental analysis (Proops et al., 1993).

1.5.5 Application-Dependent Selection of Analytical Tools

According to Wenzel (1998), the governing dimensions for applications of analytical tools such as LCA are site specificity, time scale and need for certainty, transparency and documentation. Possible applications can be positioned in relation to these governing dimensions. In the case of LCA, Sonnemann et al. (1999) have examined in which cases LCA is an integrated element of another concept and for which goals other tools of environmental management should accompany it. They correspond to the following points:

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 environmentally friendly product (re)design and development is called design for the environment (DfE). LCA provides the information to support it.

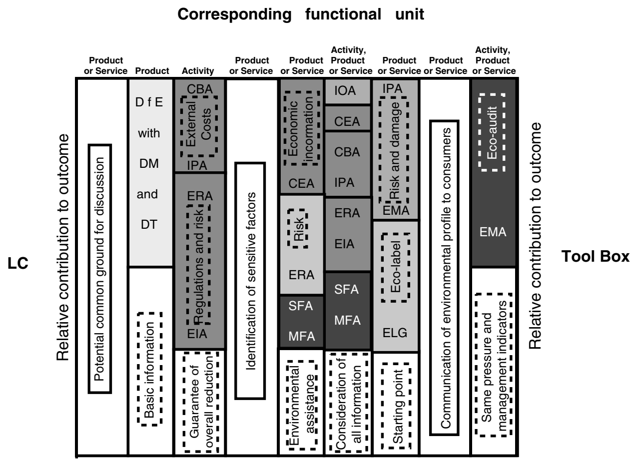
3. **Production technology assessment.** LCA helps to ensure that overall reductions are achieved and pollutants are not shifted elsewhere in the life-cycle, although other tools are needed for the assessment of the actual impacts of the technology.
4. **Improving environmental program.** LCA can be 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 from other tools.
7. **Environmentally friendly purchasing support.** LCA is obviously a starting point for eco-labeling, but the current emissions and resource indicators do not give the full picure of sustainable performance. LCA information should be complemented with data from other tools.
8. **Marketing strategies.** By using LCA it is possible to develop an environmental profile of a product or service that can be communicated to the consumers.
9. **Environmental performance and liability evaluation.** The combination of an environmental management system with LCA is an interesting topic for the future. For this reason, it is necessary to use the same pressure and management indicators.

By using the proposed guide in the form of a matrix (Figure 1.7), the environmental practitioner should be able neither to overestimate the possibilities of LCA nor to be discouraged from using it because of its inherent limitations.

Therefore, we can say that life-cycle thinking is the right concept to evaluate the environmental impacts of a functional unit (product, service or activity) but that LCA is often not the only tool to consider. The practical guide (Figure 1.7) for environmental decision-makers helps select the tool that corresponds to a particular application.

1.6 EXAMPLE: DECISION-MAKING SITUATION IN ENVIRONMENTAL MANAGEMENT

Political administration or business managers very often make decisions of environmental relevance. As can be seen in Figure 1.8, their decisions may enhance or worsen the environmental performance of a chemical substance, process, product or whole region.



Applications
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. Environmentally 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: Dematerialization
EMA: Environmental Management
MFA: Material Flow Accounting
DT:Detoxification and Audit
SFA:Substance Flow Analysis

FIGURE 1.7 Matrix guide for the inclusion of the life-cycle concept in environmental management practices. 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) environmentally friendly purchasing support; 8) marketing strategies; 9) environmental performance and liability evaluation. **CBA**: cost-benefit analysis; **EIA**: environmental impact assessment; **IOA**: input–output analysis; **CEA**: cost-effectiveness analysis; **ELG**: eco-labeling; **IPA**: impact pathway analysis; **DfE**: design for environment; **ERA**: environmental risk assessment; **LCA**: life-cycle assessment; **DM**: dematerialization; **EMA**: environmental management; **MFA**: material flow accounting; **DT**: detoxification and audit; **SFA**: substance flow analysis.

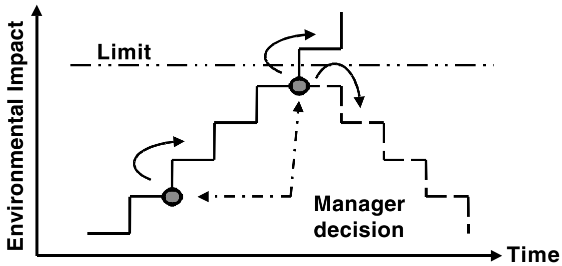


FIGURE 1.8 Decision-making situation in environmental management.

A risk is that, after several decisions, the acceptable level of resource consumption and pollution is exceeded. In the end, in the case of business managers, their decisions may cause problems with the administration, neighbors of their facilities or the consumers of their product. In the case of administrations, decisions may raise protests among the population and lost elections. In general, managers lack sufficient time to apply environmental management tools. They entrust internal and/or external specialists (consultants) the corresponding projects or assessments and base the obtained results to support and justify their decision-making under rational arguments. When managers do not master environmental management tools, they are at risk of choosing a specialist who applies a certain tool that will render a result subject to the methodology on which it is based. Consider, for example, the question of whether to build a new thermal power plant near carbon mines in a very populated area or far from mines in a sparsely populated area.

ERA applied to population exposure would choose the second option, while with an LCA the first option would seem more appropriate due to reduced transportation. In sum, we must not trust these tools blindly; instead we must understand their inherent limitations and apply each one, or a combination of them, to the right context.

1.7 CASE STUDY: WASTE INCINERATION AS ENVIRONMENTAL PROBLEM — THE CASE OF TARRAGONA, SPAIN

1.7.1 Waste Incineration as Environmental Problem

In recent years, waste incineration has been frequently preferred to other waste treatment or disposal alternatives due to advantages such as volume reduction, chemical toxicity destruction and energy recovery. However, strong public opposition to waste incineration often impedes the implementation of this technology. One of the main reasons for this opposition has been the perception that stack emissions are a real and serious threat to human health (Schuhmacher et al., 1997). In past years, the environmental consequences of incineration processes and their potential impact on public health by emissions of trace quantities of metals and polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs), as well as other emission products, have raised much concern. Unfortunately, information presented to the public about health risks of incineration is often incomplete, including only data on PCDD/Fs levels in stack gas samples (Domingo et al., 1999). In order to get overall information on the environmental impact of a municipal solid waste incinerator (MSWI), a wider study must be performed. Next we present and analyze the case of an MSWI in Tarragona, Spain. This case study is presented with three different alternatives explained below.

1.7.2 Municipal Solid Waste Incinerator (MSWI) in Tarragona, Spain

Our case study will focus on an MSWI (SIRUSA) located in Tarragona (northeastern Spain) that has operated since 1991. In 1997 an advanced acid gas removal system was installed. Thus, two situations (or scenarios) were studied: the operation of the plant during 1996 (later called former situation or scenario 1) and the current operation with the advanced acid gas removal system working (later called current situation or scenario 2).

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 950 and 1000°C. In the secondary postcombustion chamber, the temperature is 650 to 720°C and the output temperature of the flue gas is 230 to 250°C. The minimum incineration conditions are 2 s of incineration time at 850°C 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 solids. The average composition of the municipal waste is shown in Table 1.1 and a schematic overview of the plant is given in Figure 1.9.

The flue gas cleaning process is a semidry 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, ensures 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 1.2.

The emissions are also controlled by the local authorities (Delegació Territorial del Departament de Medi Ambient de la Generalitat de Catalunya.). Thus, the plant is under continuous, real-time control, which guarantees independent information on the emissions.

TABLE 1.1
Waste Composition of the MSWI Plant in Tarragona, Spain[a]

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

[a] Average, 1999.

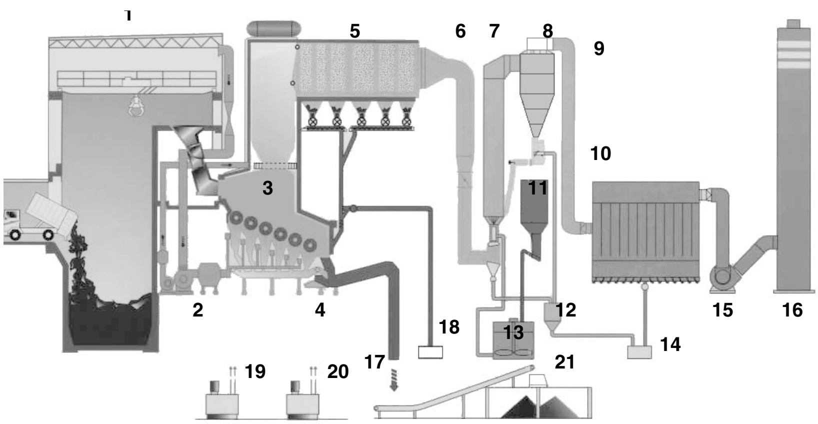


FIGURE 1.9 Scheme of the MSWI plant in Tarragona. (From Nadal, 1999.) 1. crane bridge; 2. fans; 3. oven; 4. slag extractor; 5. boiler; 6. combustion gases; 7. reactor; 8. separation cyclone; 9. active coal; 10. recycling; 11. silo; 12. silo; 13. heat pump; 14. fan; 15. fan; 16. chimney; 17. slag treatment; 18. ashes; 19. feeder hydraulic station; 20. extractor hydraulic station; 21. slag.

Material inputs and outputs for subprocesses important for the process chain and taken into account with their site-specific data in the LCA (see Chapter 2) are mentioned in Tables 1.2 and 1.3. An overview of the process chain and the origin of the material is given in Figure 1.10.

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 Envi-

TABLE 1.2
Overview of Data from the MSWI Plant in Tarragona, Spain

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[a] (g/Nm^3)	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 ($\mu g/Nm^3$)	20	5.6
Cd ($\mu g/Nm^3$)	20	6.6
Heavy metals[b] ($\mu g/Nm^3$)	450	91
Ni ($\mu g/Nm^3$)	30	8.4
PCDD/Fs (ng/Nm^3) as toxicity equivalent (TEQ)	2	0.002
Materials		
In		
CaO (t/yr)	0	921
Cement (t/yr)	88.5	518
Diesel	148.8	148.8
Out		
Slag (t/yr)	42,208	42,208
Scrap for treatment (t/yr)	2,740	2,740
Ashes for treatment (t/yr)	590	3,450
Ashes for disposal (t/yr)	767	4,485
Plant data		
Gas volume (Nm^3/h)	90,000	90,000
Gas temperature (K)	503	503
Stack height (m)	50	50
Stack diameter[c] (m)	1.98	1.98
Latitude (°)[d]	41.19	41.19

(*continued*)

TABLE 1.2 (CONTINUED)
Overview of Data from the MSWI Plant in Tarragona, Spain

Situation	Without new filters	With new filters
Longitude (°)[d]	1.211	1.211
Terrain elevation (m)	90	90

[a] Corresponds to the measured value, not to the adjusted one used in the LCA study (see Chapter 2).
[b] Heavy metals is a sum parameter in the form of Pb equivalents of the following heavy metals: As, B, Cr, Cu, Hg, Mn, Mo, Ni, Pb, and Sb. Cd is considered apart for its toxic relevance and As and Ni for their carcinogenic relevance.
[c] Although there are two stacks with 1.4 m, due to the limitations of the dispersion models used, one stack with a diameter of 1.98 was considered.
[d] Initially the data were in UTM, the Mercator transversal projection. The conversion was made using the algorithm in http://www.dwap.co.uk/welcome.

ronmental Management and Analysis (AGA) Group of the Universitat Rovira i Virgili in Tarragona, Spain. The MS–Excel-based model takes into account the elementary waste input composition and relevant plant data, such as plant layout and process specific constants.

In the model, the steam generator consists of grate firing and heat recovery systems and a regenerative air preheater. Energy production is calculated using the heating value of the waste input and the state points of the steam utilization 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. Because 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 a semidry 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 1.11.

Based on the modular model, a future scenario, scenario 3 to be used in Chapter 7, was created for an 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 1.3. All the

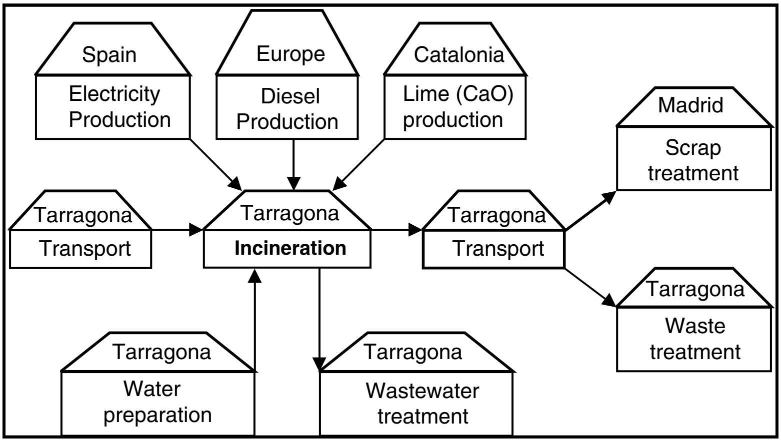


FIGURE 1.10 Overview of the MSWI process chain.

calculated emissions are lower than the current situation 2, scenario of the MSNI in Tarragona, Spain. Also, the corresponding transport distances are presented because they are necessary for the estimation of environmental damages within industrial process chains in Chapter 6.

1.8 QUESTIONS AND EXERCISES

Several typical situations for environmental management decisions are given. Students are asked to think of a tool, or a combination of tools, for environmental management in order to apply them to a manager's or consultant's situation.

1. The population neighboring an industrial area is becoming increasingly sensitive to issues on environmental pollution. Business A wants to protect itself against possible population claims for environment pollution due to their industrial activity.
2. Business B wants to make sure the implementation of a new gas treatment in an existing plant will be environmentally friendly.
3. Business C is developing a new electronic product that will be introduced to the market within a year.
4. Business D is developing a new detergent that will be introduced to the market within a few years.
5. Business E is developing a marketing campaign under the slogan of "Join the Struggle against Climate Change" to increase sales of its product manufactured with fully renewable resources.
6. Business F intends to undertake an environmental enhancement for its galvanization process.
7. Business G intends to purchase a civil responsibility insurance policy for its waste outlet.

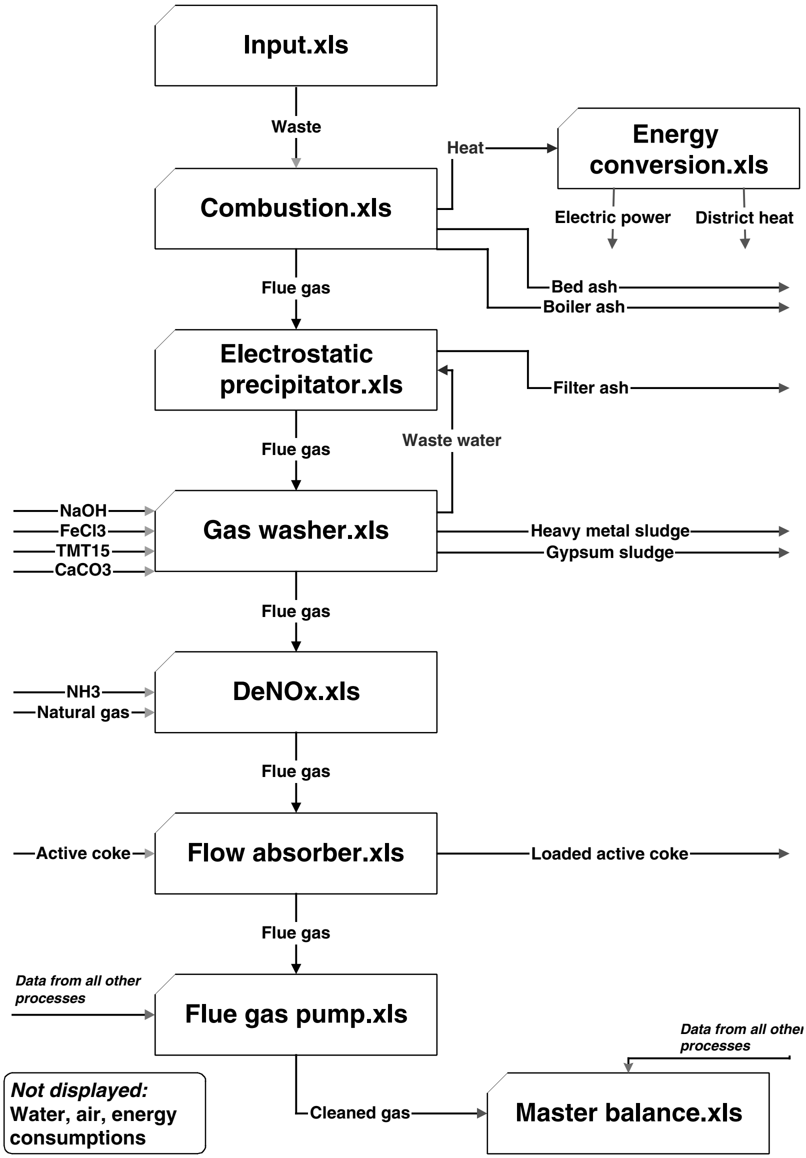


FIGURE 1.11 Workbook structure of the molecular spreadsheet model (Hagelüken, 2001.)

8. Administration H must decide whether it will grant a license for the construction of a new 50-km highway linking city X with city Y.
9. Administration I is searching for a location at which to build an urban solid waste incineration plant.
10. The European Commission must decide whether a subsidy for electrical automobiles is justified from the environmental perspective.

TABLE 1.3
Overview of Inputs and Outputs of Scenario 3

Overall transport (tkm/yr)[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 /yr)	
Consumption	18.41
Production	290.11
Materials (t/yr)	
In	
CaO	2241
Ammonia	686
Cement	1341
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 ($\mu g/Nm^3$)	0.035
Cd ($\mu g/Nm^3$)	1.50
Ni ($\mu g/Nm^3$)	0.51
NOx (mg/Nm^3)	58
PM10 (mg/Nm^3)	0.45

[a] tkm is equivalent to a mass of 1 t (1000 kg) transported 1 km.

REFERENCES

AICHE (American Institute of Chemical Engineers) (1985), Guidelines for Hazard Evaluation Procedures, prepared by Batelle Columbus Division for The Center for Chemical Process Safety.

Alfonso, P. and Krämer, L. (1996), *Derecho Medioambiental de la Unión Europea*, McGraw-Hill, Madrid.

Ayres, R.U. and Ayres, L.W. (1996), Industrial ecology, in Ayres, R.U. and Simonis, U.E., Eds., *Industrial Metabolism: Restructuring for Sustainable Development*, United Nations University Press, New York, 1994. Edward Elgar Publishing Co., Brookfield, VT.

Brezet, H. and Hemel, C. (1997), *Ecodesign — a Promising Approach to Sustainable Production and Consumption*, UNEP-TIE, Paris.

Bringezu, S., Fischer-Kowalski, M., Kleijn, R., and Palm, V. (1997), Regional and national material flow accounting — from paradigm to practice of sustainability, *Proceedings of the ConAccount Workshop*, Leiden, The Netherlands.

CHAINET (European Network on Chain Analysis for Environmental Decision Support) (1998), Definition document, CML, Leiden University, The Netherlands.

Ciroth, A. (1998) Beispielhafte Anwendung der Iterativen Screening – Ökobilang-Diplomarbeit Institut für Technischen Umweltschutz, Fachgebeit Abfallvermeidung, Technische Universität Berlin, Germany.

Coneza, F. (1997), *Guía Metodológica para la Evaluación del Impacto Ambiental*, Ediciones Mundi-Prensa, Madrid.

Dasgupta, A.K. and Pearce, D.W. (1972), *Cost-Benefit Analysis — Theory and Practice*, MacMillan Press, Cambridge.

De Smet, B., Hemming, C., Baumann, H., Cowell, S., Pesso, C., Sund, L., Markovic, V., Moilanen, T., and Postlethwaithe, D. (1996), Life-cycle assessment and conceptually related programs, working group draft, SETAC-Europe, Brussels.

Domingo, J.L., Schuhmacher, M., Granero, S., and Llobet, J.M. (1999), PCDDs and PCDFs in food samples from Catalonia, Spain. An assessment of dietary intake, *Chemosphere*, 38, 3517–3528.

EC (European Commission) (1995), *ExternE – Externalities of Energy*, 6 vol., EUR 16520-162525. DG XII Science, Research and Development, Brussels.

EC (European Commission) (1997), Guidelines for the application of life-cycle assessment in the EU eco-label award scheme. A report prepared for the EC by the "Groupe des Sages," Brussels.

EU directive on packaging and packaging waste (1994), European Parliament and Council Directive 94/62/EC, http://www.efpa.com/eulaws/B1.pdf, September 1, 2003.

EC (European Commission) (1999), Integrating environmental concerns into development and economic cooperation, Brussels, http://glossary.eea.eu.int/EEAGlossary/E/environmental_management_system, September 25, 2000.

Federal Environment Agency Ltd. of Austria (2000), http://www.ubavie.gv.at/publikationen/uba-aktuell/archiv/1999/08/TM_1999-08-06-1.htm, September 25, 2000.

Field, B.C. (1995), *Economía Ambiental — Una Introducción*, McGraw-Hill, Santafé de Bogotá, Columbia.

Fiksel, J. (1997), *Ingeniería de Diseño Medioambiental*, Mc Graw Hill, Madrid.

Fuster, G., Schuhmacher, M., and Domingo, J.L. (2002), Human exposure to dioxin and furans. Application of the substance flow analysis to health risk assessment, *Environ. Int. Sci. Pollut. Res.*, 9(4), 241–249.

Graedel, T. (1994), Industrial ecology — definition and implementation, in *Industrial Ecology and Global Change*, Socolow, R. et al., Eds., Cambridge University Press, New York, Chap. 3.

Hagelüken, M. (2001), Effects of different models of municipal solid waste incinerators on the results of life-cycle assessments, student research project, cooperation of TU Berlin and Universitat Rovira i Virgili, Tarragona, Spain.

Hansen, D.A. (2000): Total Quality Management (TQM)–Tutorial Help Page, http://home.att.net/~iso9k1/tqm/tqm.html, September 25, 2000.

IIIEE (International Institute for Industrial Environmental Economics) (2000), *Extended Producer Responsibility*, Lund University, Sweden, http://www.lu.se/IIIEE/research/products/epr/epr_old.html, September 25, 2000.

ISO 14001 (1996), Environmental Management Systems — Specification with Guidance for Use, International Organization for Standardization, Geneva, Switzerland.

ISO 14010 (1996), Guidelines for Environmental Auditing — General Principles, International Organization for Standardization, Geneva, Switzerland.

Kremer, M., Goldhan, G., and Heyde, M. (1998), Waste treatment in product-specific life-cycle inventories, *Int. J. LCA*, 3(1), 47–55.

Masters, G. (1991), *Introduction to Environmental Engineering and Science*, Prentice Hall, Englewood Cliffs, NJ.

Miller, R.E. and Blair, P.D. (1985), *Input–Output Analysis: Foundations and Extensions*, Prentice-Hall, Englewood Cliffs, NJ.

Nadal, R. (1999), Planta incineradora e residuos sólidos urbanos de Tarragona. Report for Master en Enginyeria i Gestio Ambiental (MEGA), Universitat Rovira i Virgili, Tarragona, Spain.

Nas, T.F. (1996), *Cost-Benefit Analysis — Theory and Application*, SAGE Publications, New York.

Proops, J., Faber M., and Wagenhals, G. (1993), *Reducing CO_2 Emissions — a Comparative Input–Output Study for Germany and the U.K.*, Springer-Verlag, Heidelberg, Germany.

Rodrigo, J. and Castells, F. (2002), *Electrical and Electronic Practical Ecodesign Guide*, Universitat Rovira i Virgili, Tarragona, Spain.

Schuhmacher, M., Meneses, M., Granero, S., Llobet, J.M., and Domingo, J.L. (1997), Trace element pollution of soils collected near a municipal solid waste incinerator: human health risk, *Bull. Environ. Contam. Toxicol*., 59, 861–867.

SETAC (Society of Environmental Toxicology and Chemistry) (1993), *Guidelines for Life-Cycle Assessment — a Code of Practice*, Sesimbra/Portugal SETAC Workshop report, SETAC Press, Pensacola, FL.

SOCMA (Synthetic Organic Chemical Manufacturers Association) (2000), product stewardship, http://www.socma.com/respcare/codes/prodstew.html, 25/09/2000.

STQ (Servei de Tecnologia Química) (1998), *Análisis del Ciclo de Vida de la Electricidad Producida por la Planta de Incineración de Residuos Urbanos de Tarragona*, technical report, Universitat Rovira i Virgili, Tarragona, Spain.

Sonnemann, G.W., Castells, F., and Rodrigo, J. (1999), Guide to the inclusion of LCA in environmental management, 8th Congreso Mediterráneo de Ingeniería Química, Expoquimia, Barcelona.

Umweltbundesamt, Germany (1998), http://www.umweltbundesamt.de/uba-info-daten-e daten-e/environmental-audit.htm, September 25, 2000.

UNEP and WBCSD (2000), Eco-Efficiency and Cleaner Production: Charting the Course to Sustainability, http://www.iisd.ca/linkages/consume/unep.html, September 25, 2000.

UNEP DTIE (Division of Technology, Industry and Economics) (1999), Towards the global use of life-cycle assessment, technical report, United Nations Environment Program, Paris.

UNEP-IE (Division of Industry and Environment) (2000), Production and Consumption Unit, http://www.unepie.org/Cp2/what_is/general_info.html, September 25, 2000.

UNEP/SETAC (2001), Background paper of the UNEP/SETAC Life-Cycle Initiative, UNEP DTIE, Paris.

U.S. EPA (2000a), Information on pollution prevention P2: http://www.epa.gov/opptintr/p2home/, September 25, /2000.

U.S. EPA (2000b), Information on "What is Green Chemistry?" http://www.epa.gov/opptintr/greenchemistry/whatis.htm, September 25, 2000.

Vasanthakumar, N.B. (1998), *Total Quality Environmental Management: an ISO 14000 Approach*, Quaum Books, Westport, CT.

WBCSD (1999), Eco-efficiency indicators and reporting, technical report, Working Group on Eco-Efficiency Metrics & Reporting, Geneva, Switzerland.

Wenzel, H. (1998), Application dependency of LCA methodology — key variables and their mode of influencing the method, *Int. J. LCA*, 3(5), 281–288.

Zharen, W.M. (1995), *ISO 14000 — Understanding the Environmental Standards*, Government Institutes, Rockville, MD.

2 Life-Cycle Assessment

2.1 INTRODUCTION

2.1.1 CONCEPT OF LIFE-CYCLE ASSESSMENT

Life-cycle assessment (LCA) of a product comprises the evaluation of the environmental effects produced during its entire life-cycle, from its origin as a raw material until its end, usually as a waste. This concept goes beyond the classical concept of pollution from the manufacturing steps of a product, taking into account the "upstream" and "downstream" steps. These steps can be illustrated by using the life-cycle of a chair as an example.

Let us imagine that our chair would be manufactured in polished wood fixed by iron screws, and that the seat would be made of a low-density foam layer covered by polyamide fabric. If we carried out an evaluation of the chair based only on its manufacturing stage, the study would show insignificant environmental impact. This would be justified by the simplicity of the production process, in many cases reduced to electricity consumption, dust generation and wood waste production due to the assembly of all components. However, according to the life-cycle approach, we must consider all the previous operations carried out in order to transform natural resources into the intermediate products that will make up the chair. In this case, the study would start with primary activities like wood planting, iron mining and crude oil extraction and continuing through fabric manufacturing to the final assembly of the chair. Moreover, we must include later stages such as use and final disposal from the point of view of environmental impact. This means that we need to evaluate each aspect related to natural resources consumption or waste releases from the entire life-cycle of the chair. The most recognized and well-accepted method of carrying out environmental assessment of products and services along their life-cycles is the methodology of LCA. This chapter will present an overview of its conceptual framework, common applications and importance for eco-design and environmental management solutions.

In this framework, 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 manufacture, distribution and use to the final processing of the disposed product. Throughout all these stages, extraction and consumption of resources (including energy) and releases into air, water and soil are identified and quantified. Subsequently, the potential impact contribution of these resources' extraction and consumption, as well as environmental releases causing several important types of environmental impacts, is assessed and evaluated (Curran, 1996; EEA, 1998).

2.1.2 History of Life-Cycle Assessment

An overview of LCA history can be found in Assies (1992), Vigon et al. (1993), Pedersen (1993), Boustead (1992) and Castells et al. (1997). It is not easy to determine exactly when studies related to the methodology that would later be known as life-cycle assessment started. In the opinion of Vigon et al. (1993), one of the first studies was H. Smith's, whose calculations of energy requirements for manufacturing final and intermediate chemical products entered the public domain in 1963. Later, other global studies such as those by Meadows et al. (1972) and the Club of Rome (1972) predicted the effects of an increase in population and energy and material resources. These predictions (which foretold fast consumption of fossil fuels and the climate changes resulting from it), together with the oil crisis of the 1970s, encouraged more detailed studies, focused mainly on the optimum management of energy resources. As explained by Boustead and Hancock (1979), because of the necessity of solving material balance in the process in order to undertake such assessments, it was necessary to include raw material consumption as well as waste generation. The so-called "energy assessments" date back to these years; Assies (1992) quotes, Boustead (1974) and IFIAS (1974). Although these studies focused basically on the optimization of energy consumption, they also included estimations on emissions and releases. More references about these assessments can be found in Boustead (1992).

Vigon et al. (1993) highlighted the 1969 Coca-Cola™ study carried out by the Midwest Research Institute (MRI) aimed at determining the type of container with the lowest environmental effect. However, Assies (1992) considers MRI's assessment conducted by Hunt (1974) for the U.S. Environmental Protection Agency (EPA) in order to compare different drink containers to be the forerunner of LCA studies. This study uses the term "resource and environmental profile analysis" (REPA) and is based on the analysis of a system following the production chain of the researched products from "cradle" to "grave" in order to quantify the use of resources and emissions to the environment. The study was also to develop a procedure enabling comparison of the environmental impacts generated by those products.

In 1979 the SETAC (Society for Environmental Toxicology and Chemistry), a multidisciplinary society of professionals with industrial, public and scientific representatives, was founded. One of SETAC's goals was, and continues to be, the development of LCA methodology and criteria. In the same year, Boustead and Hancock (1979) published a study describing the methodology of energy assessment with the idea of making energy treatment more systematic and establishing criteria to compare various energy sources.

In 1984 the EMPA (Swiss Federal Laboratories for Materials Testing and Research) conducted research that added the effects on health to emission studies and took into account a limited number of parameters, thus simplifying assessment and decision-making. Products were assessed on the basis of their potential environmental impact expressed as energy consumption, air and water pollution, and solid wastes. It also provided a comprehensive database with access to the public that, according to Assies (1993), catalyzed the implementation of LCA (EMPA, 1984; Druijff, 1984).

Due in part to higher access to public data and an increased environmental awareness of the population, new LCA projects were developed in the industry as well as in the public sector. Such growth resulted in the "launch" of this subject at an international level in 1990. That year, conferences about LCA were held in Washington, D.C. (organized by the World Wildlife Fund and sponsored by the EPA), Vermont (organized by SETAC) and Leuven, Belgium (organized by Procter & Gamble).

On their part, public organizations such as the Swiss Federal Office of Environment, Forests and Landscape (BUWAL) started to study industrial sectors or products at the time they made the achieved results public. Among these, it is worth mentioning BUWAL's 1991 and 1994 reports. Also, in the private sector, companies such as Franklin Associates, Ltd. published their studies on materials used for container manufacturing and material transportation (Franklin Associates, Ltd. 1990, 1991). Other organizations, for example, the Association of Plastics Manufactures in Europe (APME) and the European Center for Plastics in the Environment (PWMI), also published their studies on plastic materials (Boustead, 1993a, b, c; 1994a, b, c). SPOLD (Society for the Promotion of LCA Development, an association established by 20 large European businesses with the aim of fostering and standardizing the use of LCA) was founded in 1992.

Growth in the number of studies, as well as in organizations devoted to this subject matter, allowed the publication of works intended to standardize the criteria to be applied in LCA studies. Among these were Fava et al. (1991), Heijungs et al. (1992), Boustead (1992), Fecker (1992), Vigon et al. (1993), SETAC (1993) and Guinée et al. (1993a, b).

In June 1993, the ISO created the Technical Committee 207 (ISO/TC 207) with the goal of developing international norms and rules for environmental management. The fifth of the six subcommittees created, the LCA SC5, was assigned standardization within the field of LCA. Its aim is to prevent the presentation of partial results or data of questionable reliability from LCA studies for marketing purposes, thus ensuring that each application is carried out in accordance with universally valid structure and features. As a result of this work, we rely on the different ISO standards mentioned in Chapter 1 of this book (ISO 14040, 1997).

In the 1990s, the annual conferences by SETAC and the Working Groups on LCA played a paramount role in developing this methodology to its current status. For an overview on the results of this work, see Udo de Haes et al. (1999) and Udo de Haes et al. (2002a).

More recently, the Life-Cycle Initiative (UNEP/SETAC Life-Cycle Initiative, 2002; Udo de Haes et al., 2002b) has been launched jointly by UNEP and SETAC. This initiative builds on the ISO 14040 standards and aims to establish approaches with best practice for a life-cycle economy, corresponding to the call of governments in the Malmö declaration of 2000. In May 2000, UNEP and SETAC signed a letter of intent and established the Life-Cycle Initiative. The mission of the initiative is

> … to develop and implement practical concepts and tools for evaluating the opportunities, risks and trade-offs associated with products and services over their entire life-cycle.

This initiative will allow laying the foundations for LCA methodology to be used in a practical manner by all product and service sectors around the globe.

Although this methodology is currently beginning to consolidate, the application patterns of the technique to practitioners is still very much in debate. In the past years, however, there has been a growing confidence in the LCA community that the emerging tool has a real future in the Life Cycle Management (LCM) toolbox (Saur et al., 2003).

2.1.3 Common Uses of Life-Cycle Assessment

In a first approach, the uses of LCA can be classified as general and particular:

General:

- Compare alternative choices.
- Identify points for environmental enhancement.
- Count on a more global perspective of environmental issues, to avoid problem shifting.
- Contribute to the understanding of the environmental consequences of human activities.
- Establish a picture of the interactions between a product or activity and the environment as quickly as possible.
- Provide support information so that decision-makers can identify opportunities for environmental improvements.

Particular:

- Define the environmental performance of a product during its entire life-cycle.
- Identify the most relevant steps in the manufacturing process related to a given environmental impact.
- Compare the environmental performance of a product with that of other concurrent products or with others giving a similar service.

The use of LCA allows defining the environmental profile of a product throughout its life-cycle. Thus, the consumption of natural resources or releases into air, water and soil can be identified, quantified and expressed in terms of impacts on the environment. LCA does not necessarily need to be applied to the entire life-cycle of a product. In many cases, this kind of evaluation is applied to a single process such as a car assembly or to a service such as raw material transportation. Depending on the context, LCA is useful as a conceptual framework or as a set of practical tools. "Life-cycle thinking" can stimulate creativity and ability to see the extensive dimensions of a problem. In terms of strategic management, a business can find important product improvements, new approaches to process optimization and, in some cases, radically new ways of meeting the same need (only with a new product or a service) while carrying out an LCA. In this context, LCA can be seen as a support tool in decision-making processes. In addition, life-cycle management (LCM), one of the newest concepts, allows an integrated approach to minimizing environmental loads throughout the life-cycle of a product, system or service.

From a different point of view, LCA can be applied in establishing public policy. Sustainable development has been included as a major item on most governmental agendas since the Rio Summit in 1992. It is obvious that the LCA approach must be used to ensure that actions toward a more sustainable future will have the desired effect. In this framework, the main governmental applications regarding LCA are product-oriented policies, deposit-refund programs (including waste management policies), subsidy taxation and general process-oriented policies. Finally, anything we do to make LCA useful will not really help unless the world believes it is efficient. For this reason, LCA experts admit the necessity of giving more information about LCA issues in order to increase credibility of the tool and gain greater acceptance from the public. A great interest exists about what other people think of the discipline and its implications for the future.

Approaches to consumption have been valuable to the analysis of current conditions and have promoted novel strategies for future development. This has exposed the limitations of isolated production-focused strategies. What is urgently needed is to change the systems of production and consumption in an integrated way. The recent Life-Cycle Initiative (UNEP/SETAC Life-Cycle Initiative, 2002; Udo de Haes et al. 2002) mentioned in Section 2.1.2 is going in that direction, which means a life-cycle approach is needed for changing unsustainable consumption and production patterns.

2.2 LCA FRAMEWORK AND THE ISO 14000 PATTERN

The ISO standardized the technical framework for the life-cycle assessment methodology in the 1990s. On this basis, according to ISO 14040 (1997), LCA consists of the following steps (Figure 2.1):

- Goal and scope definition
- Inventory analysis
- Impact assessment
- Interpretation

LCA is not necessarily carried out in a single sequence. It is an iterative process in which subsequent 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 steps of LCA are distributed along ISO patterns. For example, ISO14040 (1997) provides the general framework for LCA. ISO 14041 (1998) provides guidance for determining the goal and scope of an LCA study and for conducting a life-cycle inventory (LCI). ISO 14042 (2000) deals with the life-cycle impact assessment (LCIA) step and ISO 14043 (2002) provides statements for the interpretation of results produced by an LCA. Moreover, technical guidelines illustrate how to apply the standards.

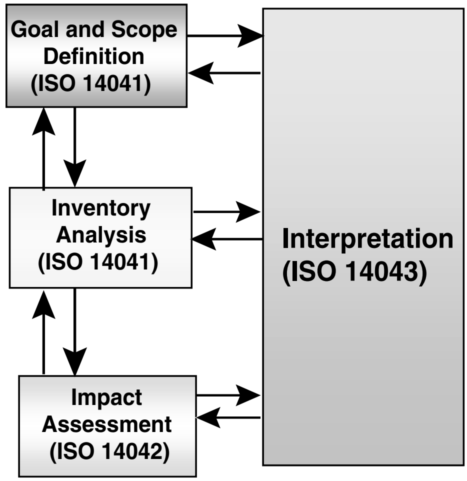


FIGURE 2.1 The phases of LCA according to ISO 14040 (1997) (available at www.afnor.fr).

2.2.1 Goal and Scope Definition

The goal and scope definition is designed to obtain the required specifications for the LCA study. During this step, the strategic aspects concerning questions to be answered and identifying the intended audience are defined. To carry out the goal and scope of an LCA study, the practitioner must follow some procedures:

1. Define the purpose of the LCA study, ending with the definition of the functional unit, which is the quantitative reference for the study.
2. Define the scope of the study, which embraces two main tasks:
   - Establish the spatial limits between the product system under study and its neighborhood that will be generally called "environment."
   - Detail the system through drawing up its unit processes flowchart, taking into account a first estimation of inputs from and outputs to the environment (the elementary flows or burdens to the environment).
3. Define the data required, which includes a specification of the data necessary for the inventory analysis and for the subsequent impact assessment phase.

2.2.2 Inventory Analysis

The inventory analysis collects all the data of the unit processes within a product system and relates them to the functional unit of the study. In this case, the following steps must be considered:

1. Data collection, which includes the specification of all input and output flows of the processes within the product system (product flows, i.e., flows to other unit processes, and elementary flows from and to the environment)
2. Normalization to the functional unit, which means that all data collected are quantitatively related to one quantitative output of the product system under study; usually, 1 kg of material is chosen, but often other units such as a car or 1 km of mobility are preferable
3. Allocation, which means the distribution of emissions and resource extractions within a given process throughout its different products, e.g., petroleum refining providing naphtha, gasolines, heavy oils, etc.
4. Data evaluation, which involves a quality assessment of the data (e.g., by eventually performing a 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.

2.2.3 Impact Assessment

The impact assessment phase aims at making the results from the inventory analysis (IA) 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 in this regard are:

1. Select and define impact categories, which are classes of a selected number of environmental impacts such as global warming, acidification, etc.
2. Classify by assigning the results from the IA to the relevant impact categories.
3. Characterize by aggregating the inventory results in terms of adequate factors (so-called characterization factors) of different types of substances within the impact categories; therefore a common unit is defined for each category. The results of the characterization step are known as the environmental profile of the product system.

More details will be given in Chapter 3.

2.2.4 Interpretation

The interpretation phase aims to evaluate the results from the inventory analysis or impact assessment and compare them with the goal of the study defined in the first phase. The following steps can be distinguished within this phase:

1. Identification of the most important results of the IA and 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 reports, including a definition of the final outcome, a comparison with the original goal of the study, drawing up 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 GOAL AND SCOPE DEFINITION

Section 2.2 briefly described the different steps of an LCA according to ISO 14040. This section describes how to run an LCA goal and scope definition in practice.

2.3.1 Purpose of an LCA Study

The main purpose of the study — the reason why an LCA is developed — must be clearly defined at the very beginning because it has a strong influence on further steps. If the study is designed to compare a product with another product that has already been submitted to an LCA, the structure, scope, and complexity of the first product's LCA must be similar to those of the other product so that a reliable comparison can be made. If the aim is to analyze the environmental performance of a product to determine its present status and to enable future improvements, the LCA study must be organized by carefully dividing the manufacturing process into well-defined sections or phases, to identify afterwards which parts of the process are responsible for each environmental effect.

2.3.2 The Functional Unit

The functional unit is the central concept in LCA; it is the measure of the performance delivered by the system under study. This unit is used as a basis for calculation and usually also as a basis for comparison between different systems fulfilling the same function. Table 2.1 presents examples of functional units related to the function performed by different systems.

An important point regarding the functional unit concerns the function carried out by the system. (When different alternatives for manufacturing products or providing service are possible, the functional unit must be clear and constantly enable a sound comparison of the options considered.) For example, let us evaluate the environmental impact produced by the transportation service of a person from Barcelona to Paris, cities separated by 1000 km. The system's function is clear: transfer a passenger. Nevertheless, the transfer can be done by different modes, except by ship.

TABLE 2.1
Examples of Functional Units

Class of products, process and services	System function	Functional unit
Goods use	Light generation	kWh/day
	Laundry washing	5 kg washed clothes
Process	Gasoline production	m^3 produced/h
	Liquid effluents treatment	t of removed COD/day
Transportation	Goods transport	tkm*
	Passengers transferring	Person km

* 100 kg transported 1 km.

Orange juice provides another good example. When the function of the system under study is orange juice consumption, the production of orange juice, its transport, processing, packaging, distribution, storage, sewage treatment and final disposal are considered. If the aim is to compare two different processes of juice production, 1 or 1000 L of orange juice will serve as the functional unit, taking into account that only the manufacturing system presents different alternatives. However, if the aim was to compare the use of different types of packaging systems, the functional unit should be consumption from a 1-L orange juice container.

Flowers are a classic example because people usually want "a bunch of flowers," rather than "750 g of flowers" or "flowers for 1 week." Thus, the functional unit should be defined as accurately as possible, considering that it should comprise the selected products and their end use, and that it is compatible with the nature of the application.

Finally, in practice, the functional unit must be measurable and, when two products with different life spans are compared, e.g., a match and a lighter, it is important that the period of use be considered for its establishment.

2.3.3 The System Boundaries

When the goal and scope definition of an LCA is made, it is crucial to define the system boundaries. They define the range of the system under study and determine the processes and operations it comprises, such as prime material extraction, manufacturing and waste disposal. In this substep, the inputs and outputs to be taken into account during the LCA study must be established. According to Lindfors et al. (1995), these can be the overall input to production as well as input to a single process; the same is true for output. Even for a quite subjective operation, the definition of system boundaries can be carried out according to the following criteria: life-cycle boundaries, geographical boundaries, and environmental load boundaries.

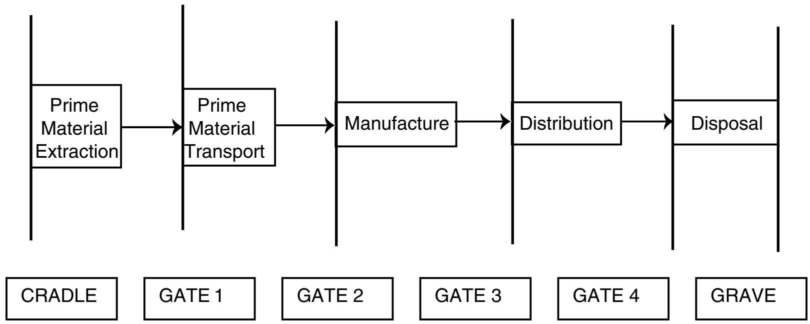


FIGURE 2.2 Product life-cycle span steps.

2.3.3.1 Life-Cycle Boundaries

Let us assume that the life-cycle span of a product is composed of the steps shown in Figure 2.2. Different system boundaries can be defined according to the life-cycle step; if it is considered the entire life from the prime material extraction until the final disposal, the limits will be defined as "cradle to grave." When the destination of a product is not known, the analysis will be stopped after manufacture and the limits will be cradle to gate 3. Other studies regarding product "stewardship" will take care of the product from manufacture until disposal, defining gate 3 to grave. In a situation of mature LCA practices, each life-cycle step will carry out its own gate-to-gate analysis and the entire cradle-to-grave process will be the result of the composition of a set of gate-to-gate systems.

2.3.3.2 Geographic Boundaries

These boundaries consider geographic limits to establish the limits of product system. They can be considered life-cycle boundaries when the different life-cycle steps are confined in some region. These criteria are well recommended in cases of site-specific studies of LCA, as will be discussed later.

2.3.3.3 Environmental Load Boundaries

Different types of environmental load are renewable and nonrenewable raw materials, air and liquid emissions, solid waste, energy losses, radiation and noise. LCA can be carried out considering the entire list of inputs and outputs (complete LCA) or taking into account air and liquid emissions (partial LCA). In Figure 2.3, partial LCA 1 considers only air and liquid emissions and is carried out from the beginning (cradle) until gate 2. Partial LCA 2 takes into account only solid waste and energy losses and goes from gate 1 to the end of life (grave).

2.3.4 Data Requirements

The quality of data used in the life-cycle inventory is naturally reflected in the quality of the final result of LCA. In this frame, it is important that the data quality be

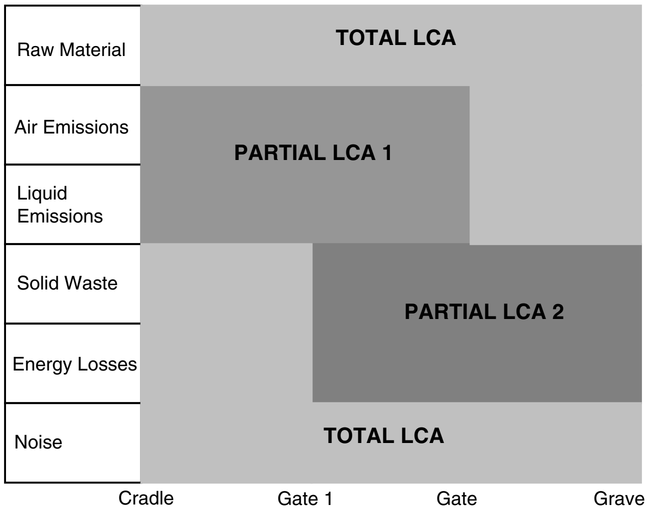


FIGURE 2.3 Boundaries in LCA.

described and assessed in a systematic way that allows other practitioners to understand and reproduce the actual LCA. Initial data quality requirements must be done with the assistance of the following parameters:

- Time-related coverage concerns a representative age (e.g., a period of 5 years) and minimum time frequency (e.g., annual).
- Geographical coverage means a geographic area from which data for a process unit should be collected (local, regional, national, continental or global) in order to satisfy the goal and approach (site specific or chain specific) of the study.
- Technology coverage reflects the nature of the technology used in process units. Depending on the product system and the goal of the study, a technological mix should be used. In this case, the technology coverage is considered, for example, as weighted average of the actual process mix, best available technology or worst operating unit.

Further description that defines the nature of the data collected from specific sites vs. data from published sources and whether the data are be measured, calculated (e.g., by material or energy balance) or estimated by similarity (from other process units with similar operational conditions to that presented by the system in study) will also be considered.

Data quality indicators such as precision, completeness, representativeness, consistency and reproducibility should be taken into consideration in a level of detail depending on the premises of the goal and scope definition step. Chapter 5 presents and discusses some techniques of data analysis applied in order to assess the uncertainty of the data to be used in the life-cycle inventory.

2.4 LIFE-CYCLE INVENTORY (LCI)

2.4.1 Introduction

Within LCA, life-cycle inventory is considered the step in which all the environmental loads or environmental effects generated by a product or activity during its life-cycle are identified and evaluated. Environmental loads are defined here as the amount of substances, radiation, noises or vibrations emitted to or removed from the surroundings that cause potential or actual harmful effects. Within this definition can be found: raw materials and energy consumption, air and water emissions, waste generation, radiation, noise, vibration, odors, etc. — what is commonly known as environmental pollution. Environmental loads must be quantifiable (valuable). Although other types of effects such as aesthetic, social, etc. must often be taken into account, they are not considered in LCI.

To prepare an LCI, each environmental load (EL) generated by the process must be added to the ELs due to material and energy inputs and the result assigned to the product. Thus, the inventory basically consists of an environmental load balance in which the ELs assigned to a product are the sum of ELs assigned to inputs plus the ones generated by the process. To illustrate this procedure, the CO_2 assigned to the product of Figure 2.4 is the sum of the CO_2 emitted by the process plus the CO_2 produced during the production of the input raw materials and the generation of the energy input.

The development of an LCI can be divided into process flow diagram and data collection, application of allocation criteria and environmental loads calculation. Each of these substeps will be briefly discussed next.

2.4.2 Process Flow Diagram and Data Collection

Data must be collected based on a process flow diagram of the system under study according to the defined life-cycle boundaries. A typical example of flow diagram applied to chair manufacturing is shown in Figure 2.5. Data collection is the most

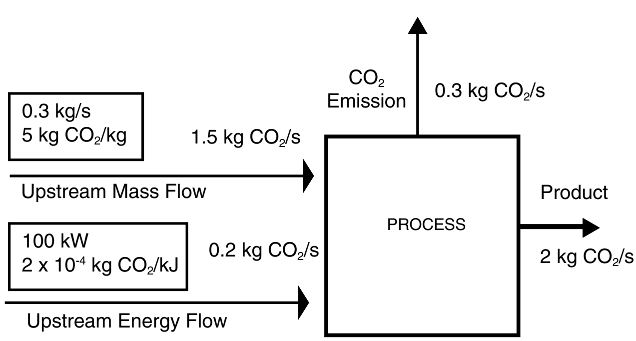


FIGURE 2.4 CO_2 assigned to a process in the life-cycle of a product.

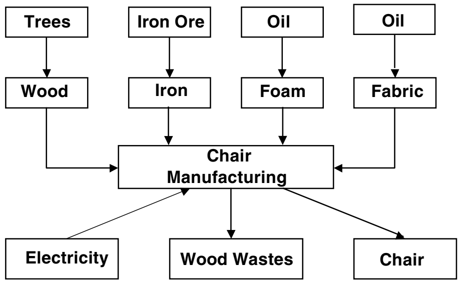


FIGURE 2.5 Simplified process flow diagram for chair manufacturing.

time-consuming task in an LCA study; establishing qualitative and quantitative information concerning the process and its elementary flows requires a lot of work. The data collection can build on data from different kinds of data sources, which can be divided into four main categories, as presented in Table 2.2.

When using data from an electronic database or literature, it is important to ensure that they concern the relevant processes and come from secure sources, and are updated and in accordance with the goal and the system range previously established.

Experience with data collection shows large differences in the availability of input and output data. Input data are the most readily available because energy and raw material consumption are registered by the companies. Also, for cases of companies with uniform production profiles, energy consumption per product unit can be calculated on the basis of the company's total energy consumption. For nonuniform profiles, energy consumption must be estimated for each individual process.

Output data, with the exception of the main product and sometimes some by-products, are difficult to find. This difficulty is due in many cases to the absence of control registers of all releases, and the impossibility of allocating the existing data to the individual product. This feature is typically dependent on the size of the company in the study. Nevertheless, as recommended by Hauschild and Wenzel

TABLE 2.2
Various Types of Data Sources

Data sources	
Electronic databases	Several databases provided by commercial and public software and Internet sources on LCA
Literature data	Scientific papers, public reports and existing LCA studies
Unreported data	Provided by companies, laboratories, authorities and correlated sources
Measurements and/or computations	Calculated or estimated where data are nonexistent or should be improved

Adapted from Hauschild and Wenzel (1998).

(1998), the problem of the availability of output data can be avoided in some cases by carrying out mass and energy balances from some inputs in order to calculate output values. This can be entirely adequate in many cases, and sometimes, even better than using data from direct measurements of the releases or emissions.

Finally, all data must be well specified concerning type and amount. Because the data collected are originally intended for other purposes, they must be processed. Units must be converted to a standard set, preferably to SI units, and the data normalized, i.e., expressed in relation to a given output from a stage or operation comprised by the product system. The previous procedure of data normalization for each step of the system is helpful and will make the further step of environmental loads calculation easy.

2.4.3 Application of Allocation Criteria

In LCA, the term "allocation" means distribution of environmental loads. If we consider a manufacturing process for only one product, there is no allocation problem because all the environmental loads must be assigned to that product. In a very common case in process industries, the same process delivers several products, so allocation criteria need to be created in order to distribute the environmental load.

For example, considering the production of eggs on a chicken farm, we will have different sizes of eggs to be sold at the market as products at corresponding prices. The application of the LCI of this farm allows distributing the environmental load of inputs (animal food, water, electricity) and the ones of the farm itself (emissions, odors, wastes, etc.) into the eggs' production, according to some distribution criteria. Instead of considering the distribution based on the number of eggs, it appears to be more reliable to distribute the environmental load according to the weight of the eggs. Weight has been used here as an allocation criterion.

Let us now consider the example of the allocation of environmental loads due to road transport of goods in a truck. If we are dealing with the transport of foam mattresses of different density, the more appropriate allocation criteria would be in this case the volume of each mattress, provided the capacity of transport is not limited by the weight but by the volume.

Another illustration is provided by a detergent industry that sells a detergent in two different concentrations: standard and concentrated. In this case it is clear that the allocation criterion to distribute the environmental load would be the weight of active product (detergent) in each type of commercial product.

It is difficult to define general rules for environmental load allocation because of the variety of options; sometimes different criteria can be used for the same process. An extended treatment of allocation criteria can be found in SETAC (1993), Pedersen (1993) and Ekvall and Finnveden (2001). Nevertheless, some indicative rules can be used:

1. The observed product bears the entire EL.
2. EL is divided proportionally to the weight of the product.
3. EL is divided proportionally to the energy content of the product.

4. Environmental loading is divided proportionally to the volume of the product.
5. In cases in which chemical reactions are involved, the EL is divided according to a molar or heat reaction basis.
6. Allocation among different types of product flows can be solved by using avoided environmental loads.
7. Allocation is avoided with the concept of system expansion.

2.4.4 Environmental Loads Calculation

After data collection and the selection of the allocation criteria, a model of environmental loads calculation is set up for the product system. One of the most efficient alternatives using an eco-vector will be presented next.

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 realization of the corresponding balance are carried out by a methodology based on an eco-vector. The eco-vector ($\boldsymbol{v}$) is a multidimensional mathematical operator in which each dimension or element corresponds to a specific EL. In an LCA study, each elementary flow is associated with an eco-vector with information about natural resource depletion and/or waste releases generated along the product system in study.

In this framework, each mass flow from the system (kg/s) has an associated eco-vector ($\boldsymbol{v}$) whose elements are expressed in specific mass bases. The most common alternative units are kilograms of pollutant per kilogram of product in cases of mass units, and kiloJoule per kilogram of product regarding energy units. ELs that cannot be expressed in terms of mass or energy (e.g., radiation or acoustic intensity) are transduced in terms of eco-vector as ELs per product mass unit (EL/kg product).

An important aspect of the use of eco-vectors to calculate environmental load is that each eco-vector must be expressed in units that can be accumulated in order to carry out material and energetic balances.

$$\boldsymbol{v}_m = \begin{bmatrix} \text{(kg / kg) or (EL / kg)} \\ \text{Renewable Raw Material} \\ \text{Not Renewable Raw Material} \\ \text{Air Emissions} \\ \text{Liquid Emissions} \\ \text{Solid Waste} \\ \text{Energy Losses} \\ \text{Radiation} \\ \text{Noise} \\ \text{Other Environmental Impacts} \end{bmatrix} \tag{2.1}$$

Expression 2.1 shows a mass eco-vector ($\boldsymbol{v}_m$) in which different kinds of environmental loads are grouped. Examples of database lists with environmental loads that constitute elements of the eco-vectors can be found in Expression 2.1 (for instance, Boustead, 1993a, b, c and Frischknecht et al., 1996).

The product of any process mass flow M [kg/s] and its corresponding eco-vector ($\boldsymbol{v}_{m,M}$) gives the rate of pollutants ($\boldsymbol{P}$) — expressed in kg/s or EL/s — generated by this mass flow until the life-cycle phase of the system, and shown in Expression 2.2:

$$\boldsymbol{P} = M \cdot \boldsymbol{v}_{m,M} \tag{2.2}$$

In parallel, an eco-vector ($\boldsymbol{v}_e$) is defined for the energy flows. The elements of ($\boldsymbol{v}_e$), in turn, are expressed in specific energy bases, e.g., kilograms of pollutant per kiloJoule. The rows of ($\boldsymbol{v}_e$) have analogous elements compared to those of the mass eco-vector, as presented by Expression 2.3:

$$\boldsymbol{v}_e = \begin{bmatrix} \text{(kg / kg) or (EL / kg)} \\ \text{Renewable Raw Material} \\ \text{Not Renewable Raw Material} \\ \text{Air Emissions} \\ \text{Liquid Emissions} \\ \text{Solid Waste} \\ \text{Energy Losses} \\ \text{Radiation} \\ \text{Noise} \\ \text{Other Environmental Impacts} \end{bmatrix} \tag{2.3}$$

The product of an energy flow E [kW] and its corresponding vector ($\boldsymbol{v}_{e,M}$) gives the pollutant flow ($\boldsymbol{P}_e$) regarding an energetic flow of the system in study. The equation to calculate ($\boldsymbol{P}_e$) is shown in the following expression:

$$\boldsymbol{P} = E \cdot \boldsymbol{v}_e \tag{2.4}$$

The use of Expressions 2.2 and 2.4 makes it possible for the environmental loads of the mass and energy flow, both measured in the same units, to be handled together, once the pollutant flows obtained by these treatments are expressed respectively in terms of natural resource consumption rate and waste release rate.

In this framework, each of the system's inputs has an associated eco-vector and its content must be distributed to the output of the system. The balance of each of the elements of the eco-vector must be closed. This means that total amount of output from a process is equal to the pollutant quantity that entered with the inlets plus the amount of pollutant generated during its operation.

To enable this balance, the output is divided into products and wastes. In order to differentiate both classes, a convention establishes that the waste flows have eco-

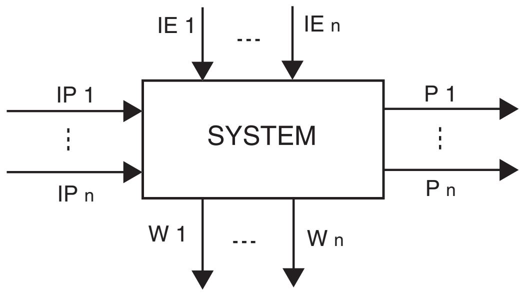


FIGURE 2.6 Representation of a generic product system.

vectors with negative elements corresponding to the pollutants they contain. The environmental load of the input and waste flows must be distributed among the products of the process.

In this way, the LCI or the balance of environmental loads of the product system under study is carried out similar to a material balance. Thus, in the case of the whole and complex plant, this can be divided into its units or subsystems, and the system of equations obtained for each of them is solved in order to calculate the eco-vectors for every intermediate or final product. The solution of the equation's systems allows detailed knowledge of the origin of the pollution, which can be assigned to each product of a plant.

The balances are carried out in a similar way for discontinuous processes, only changing the basis of the computation. For example, instead of considering a pollutant rate, the calculations are carried out in mass of pollutant per mass of obtained product. An illustration of a generic discontinuous system taken with $\boldsymbol{n}$ inputs of raw materials and energy and $\boldsymbol{n}$ outputs of products and waste releases is presented in Figure 2.6.

The algorithm resulting from the global balance of EL is given as follows:

$$\sum_{i=1}^{n}\left(Pi\cdot \boldsymbol{v}_{m,pi}\right)=\sum_{i=1}^{n}(IPi\cdot \boldsymbol{v}_{m,IPi})+\sum_{i=1}^{n}(IEi\cdot \boldsymbol{v}_{e,IEi})-\sum_{i=1}^{n}(W_i\cdot \boldsymbol{v}_{m,Wi}) \qquad (2.5)$$

where:

IP_i = mass inputs
IE_i = energy inputs
P_i = outputs (products and by-products)
W_i = wastes
$\boldsymbol{v}_{m,e}$ = mass and energy eco-vectors of the flows

The only unknowns in Expression 2.5 are the eco-vectors associated with the products. If only one product is assumed, the correspondent eco-vector will be calculated by:

$$v_{m,p} = \frac{\sum IP_i \cdot v_{m,IP_i} + \sum IE_i v_{i,IE_i} - \sum W_i v_{m,W_i}}{P} \tag{2.6}$$

In the next section, we present a simple example to learn how to calculate an LCI of a product. Nevertheless, many LCA tools in the market can help the user carry out an LCI of a big variety of systems and products. The following section briefly describes some of these existing software packages.

2.5 LCA-BASED SOFTWARE TOOLS

We are concerned with LCA-based software tools developed for relatively easy application because they may be applied by a wider range of users. Since it is not the objective of this book to present an exhaustive list of all the existing tools in the market, we have only selected some considered representatives of different application fields. We have divided the tools into four sections:

1. **Life-cycle inventory tools.** Given the data for a product, these tools will create the LCI. Thus, the life stages of the product that contributes mostly to the environmental loads considered can be identified. However, in some cases a small amount of one type of environmental load can prove to be more harmful than large amounts of others. An LCI will only provide the raw data; it will not help to identify which emissions are most significant. To do so, a full LCA is required. The LCI tools included are:
   - The **Boustead Model** is a basic MS DOS-based software package, one of the largest database available. All information is collected from industry through questionnaires. Data from over 23 countries are available, which makes Boustead a very internationally oriented tool.
     Contact details:
     Dr. Ian Boustead
     Boustead Consulting
     Tel: +44 1403 864561
     Fax: +44 1403 865284
     http://www.boustead-consulting.co.uk
   - The **Euklid** developers followed the methodology of the ISO standards and therefore limited the process to an inventory. The software package is based on an SQL database with object-oriented program structure.
     Contact details:
     Frauenhofer-Institut für Lebensmitteltechnologie und Verpackung
     Tel: +498161 491 300
     Fax: +498161 491 33
     http://www.ilv.fhg.de
   - **JEM-LCA** is an inventory tool aimed at the electronics sector with a limited database developed at NEC. This software system is based on an inventory and process tree principle.
     Contact details:

Mr. Shigeyuki Miyamoto
Ecology-Based Systems Research Laboratory, NEC Corporation
Tel: +81 3 38327085
Fax: +81 3 38327022
http://www.nec.co.jp

2. **Full LCA.** A full LCA produces results (often in graphical form) showing, for instance, the degree of global warming potential caused when a product is manufactured, or the likely damage to the ozone layer caused during use of a product. Some LCAs stop at this point; however, a few LCA methods include a third stage: weighting, in which the environmental impacts are translated into a single index (see Chapter 3 for more details). Various aspects differentiate LCA tools, such as the size of the database (varying from 100 to 6000+ materials) and the amount of support provided to the user. Most are in software format and some enable results to be exported to other software applications. LCA tools included are:
   - **EDIP LCA** tools have been developed for use in product development. EDIP is a software tool based on three groups: database, modeling tool, and calculation facilities and is available in English and Danish.
     Contact details:
     Institute for Product Development (IPU)
     Tel: +45 45932522
     Fax: +45 45932529
     http://www.dtu.dk/ipu
   - **LCAiT** is a simple graphics-based software that allows the user to set up a product life-cycle graphically and allows material and input–output balances. Because of a windows-type drop and drag system, it is simple to copy cards between different studies.
     Contact details:
     Chalmers Industriteknik CIT
     Tel: + 46 31 7724000
     Fax: + 46 31 827421
     http://www.lcait.com
   - **GaBi** is a software system designed to create life-cycle balances covering environmental and economical issues. The structure can be set up to support ISO 14040 standards. There are two possible databases and additional add-on modules.
     Contact details:
     Institut für Kunststofprüfung und Kunststofkunde & Product Engineering GmbH (IKP), Universität Stuttgart
     Tel: +49711 6412261
     Fax: +49711 6412264
     http://www.ikp.uni-stuttgart.de
   - **LCAdvantage** is a software system consisting of a graphical interface based on links, representing material and energy flows between modules that represent product components. The software also has a report

generator and contains a high degree of transparency and documentation on the information provided.

Contact details:
Pacific Northwest National Laboratory
Tel: +1 5093724279
Fax: +1 5093724370
http://www.battelle.com

- **PEMS** is based on graphical flowcharts representing a product life-cycle in four units: manufacture, transportation, energy generation and waste management. The database is transparent and allows the user to insert new information.

  Contact details:
  PIRA International
  Tel: +44 1372 802000
  Fax: +44 1372 802245
  http://www.pira.co.uk
- In **Simapro**, the database is transparent and the program allows the results to be displayed in different formats such as after classification or characterization. Simapro is a software package that comes with extensive instruction material, including an operating manual for the program, the database and the methodology.

  Contact details:
  Pré Consultants BV
  Tel: +31 33 4555022
  Fax: +31 33 4555024
  http://www.pre.nl
- **TEAM** is a software package with an extensive database and a powerful and flexible structure that supports transparency and sensitivity analyses of studies. The manufacturer offers to insert company data into the database.

  Contact details:
  Ecobalance UK, The Ecobilan Group
  Tel: +44 1903 884663
  Fax: +44 1903 882045
  http://www.ecobalance.com
- **Umberto** is a multipurpose LCA package capable of calculating material flow networks. This package uses a modular structure and offers clear transparent results. The user starts by setting up a life-cycle model, after which the process units and materials can be selected.

  Contact details:
  IFEU Institut für Energie und Umweltforschung Heidelberg GmbH
  Tel: +496221 47670
  Fax: +496221 476719
  http://ourworld.compuserve.com/homepages/ifeu_heidelberg/ifeu_eng.htm

3. **Abridged LCA.** The main criticism of LCA is the time, expertise, and data needed if the assessment is to be thorough. Therefore, abridged LCA tools have been developed that are essentially simplified and cheaper versions of LCAs. Often, the simplicity is achieved because they are (at least partly) qualitative. By using qualitative methods or displaying results only in evaluated form, these tools help the user to perform the same function as a full LCA without the need for huge amounts of data, etc. Although these methods can save a substantial amount of time and money, a certain level of background knowledge is necessary and results are not as trustworthy as those provided by full LCA. Abridged LCA tools included are:
   - ***Eco Indicator '95*** is both an LCIA method (see Chapter 3) and a manual for designers with background information on LCA. It contains a limited amount of data but allows simple LCA evaluation studies and helps designers understand the fundamentals of life-cycle thinking.
     Contact details: See Goedkoop (1995)
     PRé Consultants BV
     Tel: +31 33 4555022
     Fax: +31 33 4555024
     http://www.pre.nl
   - **MET matrices method** is a simple method of assessing and prioritizing environmental impacts of products or processes. By filling in two simple 4 × 4 matrices, the main causes of environmental impact can be determined (a reasonable level of background knowledge is required).
     Contact details: See Brezet and van Hemel (1997)
   - **AT&T product improvement matrix and target plot** is similar to the MET matrices, but more systematic. The matrix consists of questions and a scoring system requiring the user to grade certain aspects of a product or process design. The scoring system produces a target plot that indicates the areas most suited for improvement.
     Contact details: See Graedel (1995)

4. **Specialized LCA tools.** Specialized LCA tools are basically the same as normal LCA tools, but the databases are oriented toward a particular product. The majority is for the packaging sector, but they can be used and adapted for other products. (Most of them have an interactive database to which the user can add.) Specialized LCA tools included are:
   - **ECOPACK 2001-06-22** is the successor of Ecopack 2000, based on the data sets created by the Swiss EPA, BUWAL. The sets SRU 133 and SRU 250 are based on material production, energy carriers and transportation, which are all used in packaging industry.
     Contact details:
     Max Bolliger Consulting
     Tel: +41 41 6722477
     Fax: +41 41 6722477

- **Ecopro 1.4** is a software based on the flow chart principle; systems can be built out of process or transport modules. The user can add information to the database and several methods of impact assessment are available.
  Contact details:
  EMPA/Sinum GmbH
  Tel: +41 71 2747474
  Fax: +41 71 2747499
  http://www.empa.ch
- **KCL ECO** operates on a process of modules and flows. Each flow consists of a number of equations that represent masses and energies moving between two modules. The software works especially well when applied to small products and has a clear presentation style.
  Contact details:
  The Finnish Pulp and Paper Research Institute KCL
  Tel: +358943711
  Fax: +3589464305
  http://www.kcl.fi
- ***Repaq*** is an LCI tool with a database containing information on packaging materials from U.S. conditions. The user can set up a functional-unit type of description of a packaging system, specify the materials and fabrication method and insert additional information.
  Contact details:
  Franklin Associates Ltd.
  Tel: +1913 6492225
  Fax: +1913 6496494
  http://www.fal.com/
- **EIME** can be used by several persons and allows sharing of design information. By using a network set-up, environmental managers can select priority issues that will be enforced by "to do" and "do not" reminders during the design process, especially of electronic products.
  Contact details:
  Schneider Electric
  Tel: +33 1 41 29 7000
  Fax: +33 1 41 29 7100
  http://www.schneider-electric.com
- **WISARD** is an LCA software tool combined with waste management priorities. It is equipped with LCI capabilities but also allows comparison of different waste management scenarios.
  Contact details:
  Ecobalance UK, The Ecobilan Group
  Tel: +44 1903 884663
  Fax: +44 1903 882045
  http://www.ecobalance.com

2.6 EXAMPLE: SCOPE DEFINITION AND INVENTORY CALCULATION

Following the example of life-cycle assessment of the chair presented in Section 2.1.1, we will illustrate the LCA phases according to ISO 14040. According to the steps of Section 2.3, the aim of the LCA of our chair is to know the environmental performance of our product for comparison with those of concurrent and future improvements. As a functional unit, we will consider the most common product the chair model Tarraco 53, with a total weight of 5.3×10^3 g composed of the following materials:

- Wood: 4852 g (frame and seat)
- Iron: 10 g (screws)
- Foam: 124 g (seat)
- Fabric: 117 g (seat)

2.6.1 Scope of the Study

In this case, we will consider a simplified entire life-cycle taking the raw materials from its source as wood from pine trees, iron from ore, foam from oil and fabric from polyamide. The simplified process flow diagram is presented in Figure 2.5.

2.6.2 Inventory Calculation

In this simplified LCA only data relative to the material and energy inputs, and wastes, according to Figure 2.6, will be considered.

In the IA, data were collected based on existing databases obtained from TEAM®. All the input and output flows were referred to the chosen functional unit, that is, one chair. In the allocation procedure we have considered that all the manufacturing facilities produce only one type of chair, so all flows must be assigned to only one product.

Let us consider an example of the calculation of environmental load assigned to this chair. The environmental load balance will be based on the scheme presented in Figure 2.7. Material and energy inputs and outputs associated with the manufacturing of the chair (1 functional unit, FU) are characterized in Table 2.3. To simplify the application example, a condensed eco-vector containing the following environmental loads is selected:

- Raw materials:
    - Coal (in ground)
    - Natural gas (in ground)
    - Oil (in ground)
- Air emissions:
    - Carbon dioxide (CO_2, fossil)
    - Nitrogen oxides (NO_x as NO_2)
    - Sulfur oxides (SO_x as SO_2)

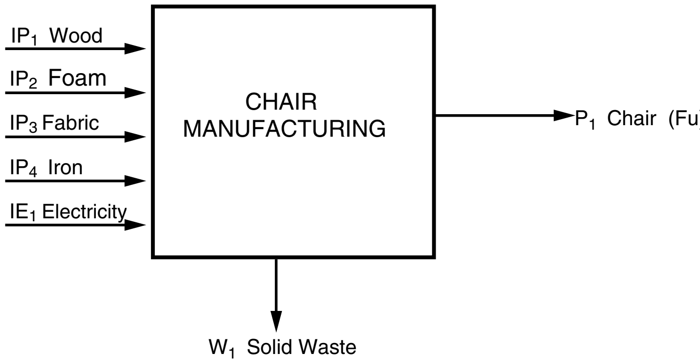


FIGURE 2.7 Environmental load balance in the manufacturing of a chair.

TABLE 2.3
Input and Output Energy and Material Flow Assigned to a Chair

Flow	Variable	Units	Value
Wood	IP1	kg	4.852
Foam (polyurethane)	IP2	kg	0.124
Fabric (polyamide)	IP3	kg	0.117
Iron	IP4	kg	0.010
Electricity	IE1	kWh	4.0
Solid waste	W1	kg	0.485
Chair	P1	FU	1

- Water emissions:
  - BOD5 (biochemical oxygen demand)
  - COD (chemical oxygen demand)
  - Nitrates (NO_3^-)
- Total Solid Waste

Based on data from software TEAM and Frischknecht et al. (1996), we obtain the eco-vectors shown in Table 2.4.

Each element of the eco-vector is presented in units of environmental load, EL, per unit of mass or energy depending on whether we are dealing with a mass or energy stream. W1 is a waste stream, representing the wood wastes generated by the chair manufacture. Provided this stream is a pure waste its elements are –1 or 0, depending on whether the corresponding environmental load of the eco-vector is present. To close the environmental balance assigning the values of the waste eco-vectors, $\boldsymbol{v}_{m,W1}$, to the product, P, of the system, the non zero elements of $\boldsymbol{v}_{m,W1}$ must be negatives (–1).

TABLE 2.4
Eco-Vectors of Input and Output Streams of Chair Manufacturing System

	Stream	**IP1**	**IP2**	**IP3**	**IP4**	**IE1**	**W1**
	Vector	$v_{m, IP1}$	$v_{m, IP2}$	$v_{m, IP3}$	$v_{m, IP4}$	$v_{e, IE1}$	$v_{m, W1}$
EL		Wood	Foam	Polyamide	Iron	Electricity	Solid waste
Load	Units	EL/kg	EL/kg	EL/kg	EL/kg	EL/kW.h	EL/kg
Coal	kg	7.93×10^{-2}	3.89×10^{-1}	7.46×10^{-1}	6.87×10^{-1}	1.39×10^{-1}	0
Natural gas	kg	3.27×10^{-2}	8.32×10^{-1}	1.41	9.10×10^{-2}	4.07×10^{-2}	0
Crude oil	kg	5.10×10^{-2}	7.38×10^{-1}	8.12×10^{-1}	5.50×10^{-2}	2.88×10^{-2}	0
CO_2	kg	1.05	4.06	7.00	2.05	5.27×10^{-1}	0
NO_x	kg	1.80×10^{-3}	1.78×10^{-2}	2.60×10^{-2}	3.60×10^{-3}	9.50×10^{-4}	0
SO_x	kg	2.01×10^{-3}	1.91×10^{-2}	2.90×10^{-2}	4.50×10^{-3}	2.57×10^{-3}	0
BOD5	kg	9.26×10^{-7}	4.64×10^{-4}	3.60×10^{-3}	1.70×10^{-4}	1.34×10^{-7}	0
COD	kg	1.45×10^{-5}	2.78×10^{-3}	1.50×10^{-2}	4.63×10^{-4}	1.99×10^{-6}	0
NO_3^-	kg	5.00×10^{-6}	6.30×10^{-3}	3.00×10^{-2}	6.90×10^{-6}	7.70×10^{-7}	0
Solid waste	kg	1.50×10^{-1}	5.95×10^{-1}	3.05×10^{-1}	7.60×10^{-1}	8.04×10^{-2}	–1

By the application of Expression 2.6, the value of the eco-vector, $\boldsymbol{v}_{m,P}$, corresponding to the environmental load associated with the chair can be obtained. Table 2.5 shows the calculation of the environmental loads as a function of inputs and outputs.

These results show the procedure to calculate the environmental load assigned to a product as a function of the environmental data of the different process inputs and the loads generated by the process. From Table 2.5 it is possible to determine the relative contribution of each input to the total value of the corresponding environmental loads, as presented in Table 2.6.

2.7 CASE STUDY: LCA STUDY OF AN MSWI IN TARRAGONA, SPAIN

For the MSWI plant introduced in Chapter 1, an LCA study (STQ 1998) was developed as an Excel spreadsheet model on the basis of the Code of Practice (SETAC 1993) and according to the steps in ISO 14040 (1997). The inventory was based on providers' information, literature data on raw materials and a detailed analysis of the incineration process.

2.7.1 Goal and Scope Definition of the MSWI LCA Study

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 Scenarios 1 and 2 (described earlier) in order to analyze the environmental efficiency of the investment in an advanced acid gas removal system. Next, the project is described according to the points indicated in ISO 14040 (1997).

TABLE 2.5
Calculation of EL Associated with the Chair as a Function of EL of Input and Output Streams

EL	Value units	IP1.v_{mIP1} (kg)	IP2.v_{mIP2} (kg)	IP3.v_{mIP3} (kg)	IP4.v_{mIP4} (kg)	IE1.v_{eIE1} (kg)	W1.v_{mW1} (kg)	v_{mP} (kg)
Coal	kg	3.85×10^{-1}	4.82×10^{-2}	8.72×10^{-2}	6.87×10^{-3}	5.58×10^{-1}	0	1.08
Natural gas	kg	1.59×10^{-1}	1.03×10^{-1}	1.65×10^{-1}	9.10×10^{-4}	1.63×10^{-1}	0	5.90×10^{-1}
Crude oil	kg	$2.47 \text{ x } 10^{-1}$	9.15×10^{-2}	9.50×10^{-2}	5.50×10^{-4}	1.15×10^{-1}	0	5.50×10^{-1}
CO_2	kg	5.09	5.04×10^{-1}	8.19×10^{-1}	2.05×10^{-2}	2.11	0	8.55
NO_x	kg	8.71×10^{-3}	2.20×10^{-3}	3.04×10^{-3}	3.60×10^{-5}	3.80×10^{-3}	0	1.78×10^{-2}
SO_x	kg	9.75×10^{-3}	2.37×10^{-3}	3.39×10^{-3}	4.50×10^{-5}	1.03×10^{-2}	0	2.58×10^{-2}
BOD5	kg	4.49×10^{-6}	5.75×10^{-5}	4.21×10^{-4}	1.70×10^{-6}	5.35×10^{-7}	0	4.85×10^{-4}
COD	kg	7.04×10^{-5}	3.44×10^{-4}	1.76×10^{-3}	4.63×10^{-6}	7.96×10^{-6}	0	2.18×10^{-3}
NO_3^-	kg	2.43×10^{-5}	7.81×10^{-4}	3.51×10^{-3}	6.90×10^{-8}	3.08×10^{-6}	0	4.32×10^{-3}
Solid waste	kg	7.28×10^{-1}	7.38×10^{-2}	3.57×10^{-2}	7.60×10^{-3}	3.22×10^{-1}	–0.49	1.65

TABLE 2.6
Relative Percentage Contribution of Input and Output Streams in Total EL Assigned to the Chair

EL	Stream	Wood	Foam	Fabric	Iron	Electricity	Manufacture	Chair
Coal	kg	35.5	4.4	8.0	0.6	51.5	0.0	100.0
Natural gas	kg	26.9	17.5	27.8	0.2	27.6	0.0	100.0
Crude oil	kg	45.0	16.6	17.3	0.1	21.0	0.0	100.0
CO_2	kg	59.6	5.9	9.6	0.2	24.7	0.0	100.0
NO_x	kg	48.9	12.4	17.1	0.2	21.4	0.0	100.0
SO_x	kg	37.8	9.2	13.1	0.2	39.7	0.0	100.0
BOD5	kg	0.9	11.9	86.7	0.4	0.1	0.0	100.0
COD	kg	3.2	15.8	80.4	0.2	0.4	0.0	100.0
NO_3^-	kg	0.6	18.1	81.2	0.0	0.1	0.0	100.0
Solid waste	kg	44.0	4.5	2.2	0.5	19.5	29.3	100.0

The function of the incineration process is to reduce the volume and toxicity of the municipal waste treated. The production of electric energy must be seen as an added value to the incineration process. Because 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 2.8. 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 transport to the final localization). The final step, with its emissions associated to the landfill, is not analyzed. The incineration process has been divided into subsystems:

1. Waste incineration plant with combustion process and including gas treatment
2. Water treatment (treatment process applied in the ash bath, demineralization process applied in the kettles by osmosis and refrigeration process by means of a tower)
3. Ash treatment (ionic ashes and waste from the gas treatment filters)
4. Scrap treatment (iron waste recycling process)

The latter were analyzed as processes that generate a useful product for the incineration process. The environmental loads are associated with the consumed product by the principal process (e.g., water) or the treated product (e.g., municipal solid waste). Literature values have been used for raw material production and the transport process (16-t truck).

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.

2.7.2 Data Used in Life-Cycle Inventory

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 analyzed processes. The considered source was the report referred to as ETH 1996 (Frischknecht et al., 1996). The exactness of the report and the agreement of these data with the particular situation in Spain determine the data quality.
- Real data of consumption and emissions associated with the incineration process, average values from 1996 (Scenario 1) and average values for 2 months with the advanced acid gas removal system in operation (Scenario 2). The data quality can be considered reliable because they have been obtained directly from the process.
- Real data of consumption and emissions associated with the waste treatment processes, obtained by visits and questionnaires answered by the

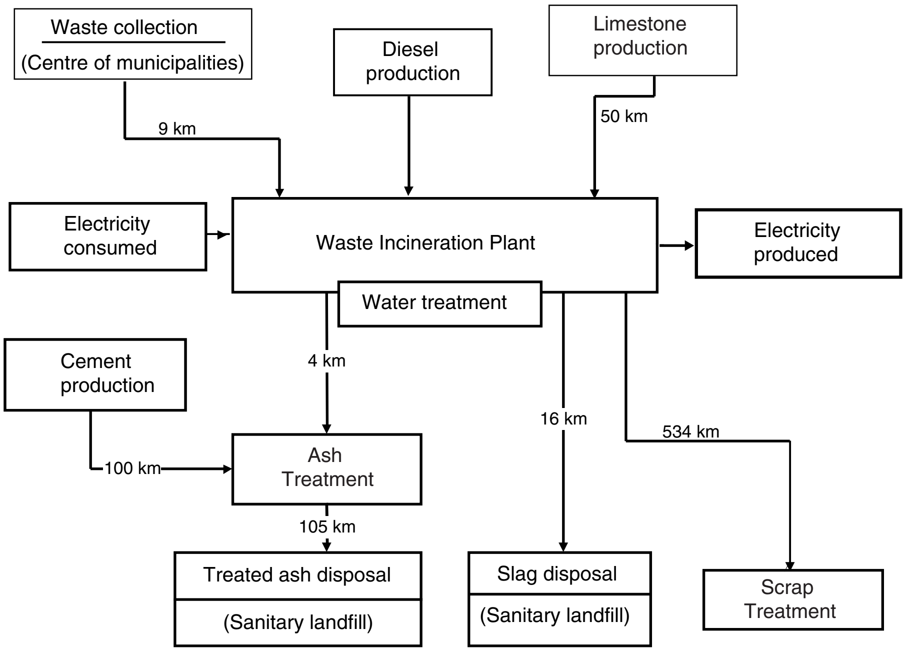


FIGURE 2.8 Processes considered in the LCA study within the boundaries of the system (including transport distances).

treatment companies. The reliability of the data delivered by the companies depends on the available information and the degree of collaboration.

2.7.3 Assumptions and Limitations

The principal assumptions are:

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

The main limitations of the study are:

- There is neither an analysis nor a total characterization of the municipal waste entering the system.
- The environmental loads associated with 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.

2.7.4 Results of the Inventory Analysis

The results for the life-cycle inventory analysis are shown in Table 2.7 and are as follows:

Raw material consumption The current situation is unfavorable for all of the 14 analyzed parameters that are considered according to Frischnkecht et al. (1996) 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 unfavorable due to the higher energetic consumption per produced TJ because of the additional energy consumption of the advanced gas treatment system.

Air emissions From the 37 analyzed parameters, the current situation is unfavorable for all except 9: As, Cd, PCDD/Fs dust, HCl, HF, Ni, SO_2 and other heavy metals. These are basically the parameters reduced by the operation of the advanced gas treatment system.

Water emissions From the 23 analyzed parameters, the current situation is unfavorable for all except 4: BOD, COD, Cd and Hg. These are basically the parameters reduced by the current operation that works without water emissions into the sewage.

2.7.5 Comparison with Spanish Electricity Mix

The electricity produced by MSWI, SIRUSA is fed into the Spanish electricity net. One ton of waste produces about 1 MJ electric 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 otherhand, some of the emissions are reduced in comparison to the Spanish electricity mix.

Therefore, the results of the MSWI LCA were compared 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. Thus, adapting the LCA methodology (Sonnemann et al. 1999), the total amount of CO_2 in the incineration process needed to be distributed in two parts between the carbon containing wastes: one for waste originating 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 carbon content of plastics is 56.43%, according to U.S. EPA (1996b).

The quantitative comparison was made with the software TEAM. The database integrated in TEAM was used for the emissions of Spanish electricity production. The calculation was normalized to 1.016 MJ electricity, which corresponds to the incineration of 1 ton of municipal solid waste as functional unit. The absolute values in Table 2.7 show the results for selected priority air pollutants. 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.

Table 2.8 compares the results of Scenario 2 with the data of the Spanish Electricity mix given by Frischknecht et al. (1996) for selected priority air pollutants. Positive values in Table 2.8 represent higher emissions of the average electricity production in Spain than in the SIRUSA plant. This means that emissions of CO, heavy metals, Ni, particles 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.

2.8 QUESTIONS AND EXERCISES

1. What are the main advantages of LCA?
2. What are the main steps of LCA and in which ISO regulation are they considered?
3. In which steps of LCA are each of the following different functions carried out: a) allocation; b) selection and definition of impact categories; c) identification of the most important results of the IA?
4. What information should be given at the end of the interpretation phase of an LCA?
5. Design the LCA framework scheme for the case of an old fabric factory building with a main structure made of steel and reinforced concrete, which has been in use for 50 years and has subsequently been transformed into a department store. Consider that the department store has been operating for 20 years and that the building is finally knocked down. The owner of the department store invested an additional amount, taking some

TABLE 2.7
Results of LCI Analysis from MSWI in Tarragona, Spain*

Emissions to air	Unit	Electricity Sit. 1[a] (TJ)	Electricity Sit. 2[b] (TJ)	Diff.[c]	Emissions to water	Unit	Electricity Sit. 1[a] (TJ)	Electricity Sit. 2[b] (TJ)	Diff.[c]
Aldehydes	kg	4.54×10^{-5}	5.92×10^{-5}	–30%	AOX	kg	1.15×10^{-3}	1.36×10^{-3}	–18%
Ammonia	kg	2.82×10^{-2}	3.70×10^{-2}	–31%	Aromatics	kg	1.73×10^{-1}	2.06×10^{-1}	–19%
As	kg	1.17×10^{-1}	3.96×10^{-2}	66%	As	kg	5.13×10^{-3}	1.04×10^{-2}	–103%
Benzene	kg	1.32×10^{-1}	1.71×10^{-1}	–30%	B	kg	1.17×10^{-2}	1.51×10^{-2}	–29%
Benzo(a)pyrene	kg	4.94×10^{-5}	5.97×10^{-5}	–21%	Ba	kg	9.27×10^{-1}	1.27	–37%
Cd	kg	1.13×10^{-1}	3.98×10^{-2}	65%	BOD	kg	5.07	8.93×10^{-2}	98%
CO	kg	2.75×10^{-5}	3.01×10^{2}	–9%	Cd	kg	9.30×10^{-3}	$8.17 \text{ x } 10^{-2}$	12%
CO_2	kg	2.33×10^{5}	2.57×10^{5}	–10%	COD	kg	1.51×10^{1}	1.37	91%
C_xH_y aromatic	kg	9.22×10^{-4}	1.28×10^{-3}	–39%	Cr	kg	3.03×10^{-2}	5.48×10^{-2}	–81%
Dichloromethane	kg	5.00×10^{-5}	5.40×10^{-5}	–8%	Cu	kg	1.39×10^{-2}	2.73×10^{-2}	–96%
Dust	kg	1.71×10^{2}	1.61×10^{2}	–8%	Dissolved subst.[d]	kg	1.01	2.14	–112%
Ethanol	kg	2.60×10^{-3}	3.56×10^{-3}	6%	Hg	kg	7.14×10^{-5}	2.73×10^{-5}	62%
Ethene	kg	2.10	2.55	–21%	Mn	kg	7.00×10^{-2}	1.28×10^{-1}	–83%
Ethylbenzene	kg	1.47×10^{-2}	1.78×10^{-2}	–21%	Mo	kg	8.40×10^{-3}	1.64×10^{-2}	–95%
Formaldehyde	kg	9.10×10^{-3}	1.61×10^{-2}	–77%	NH_3	kg	5.23×10^{-1}	6.13×10^{-1}	–17%
H_2S	kg	1.85×10^{-2}	2.91×10^{-2}	–57%	Ni	kg	1.34×10^{-2}	2.69×10^{-2}	–101%
Halon-1301	kg	2.25×10^{-3}	2.66×10^{-3}	–18%	Nitrates	kg	6.75×10^{-1}	8.06×10^{-1}	–19%
HCl	kg	2.89×10^{3}	1.95×10^{2}	93%	Pb	kg	2.56×10^{-2}	3.55×10^{-2}	–39%
Heavy metals	kg	2.65	7.17×10^{-1}	73%	Phosphate	kg	1.51×10^{-1}	3.10×10^{-1}	–105%
HF	kg	9.93	2.80	72%	Sb	kg	4.43×10^{-5}	7.10×10^{-5}	–60%
Methane	kg	3.32×10^{1}	4.92×10^{1}	–48%	SO_4^-	kg	2.31×10^{1}	3.74×10^{1}	–62%
N_2O	kg	1.71	2.06	–20%	Undissolved subst.[d]	kg	2.68×10^{1}	2.94×10^{1}	–10%
Ni	kg	1.80×10^{-1}	6.51×10^{-2}	64%	TOC	kg	3.22×10^{1}	3.82×10^{1}	–19%

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TABLE 2.7 (continued)
Results of LCI Analysis from MSWI in Tarragona, Spain*

Emissions to air	Unit	Electricity Sit. 1[a] (TJ)	Electricity Sit. 2[b] (TJ)	Diff.[c]	Emissions to water	Unit	Electricity Sit. 1[a] (TJ)	Electricity Sit. 2[b] (TJ)	Diff.[c]
Non methane VOC	kg	6.98×10^{1}	8.31×10^{1}	–19%					
NO_x	kg	1.23×10^{3}	1.33×10^{3}	–8%	Bauxite (ore)	kg	3.59×10^{1}	4.27×10^{1}	–19%
PAH	kg	7.37×10^{-4}	1.02×10^{-3}	–38%	Clay	kg	2.12×10^{2}	1.01×10^{3}	–376%
Phenol	kg	7.39×10^{-6}	9.93×10^{-6}	–34%	Coal	kg	1.50×10^{3}	3.15×10^{3}	–110%
Phosphate	kg	1.94×10^{-3}	3.88×10^{-3}	–100%	Copper (ore)	kg	2.71	3.36	–24%
Pentane	kg	5.51×10^{-1}	6.54×10^{-1}	–19%	Iron (ore)	kg	-1.32×10^{4}	-1.38×10^{4}	–5%
Propane	kg	4.35×10^{-1}	5.57×10^{-1}	–28%	Lignite	kg	6.56×10^{2}	8.86×10^{2}	–35%
Propene	kg	3.07×10^{-2}	4.49×10^{-2}	–46%	Limestone (ore)	kg	1.03×10^{3}	1.62×10^{4}	–1473%
SO_2	kg	5.08×10^{2}	2.56×10^{2}	50%	Natural gas	Nm^3	1.06×10^{2}	5.48×10^{2}	–417%
TCDD-Equiv.	ng	1.12×10^{7}	1.43×10^{4}	100%	Nickel (ore)	kg	1.00	1.19	–19%
Tetrachloromethane	kg	2.55×10^{-5}	3.63×10^{-5}	–42%	Oil	kg	5.79×10^{3}	6.86×10^{3}	–18%
Toluene	kg	6.68×10^{-2}	8.39×10^{-2}	–26%	Silica (ore)	kg	8.74×10^{3}	1.04×10^{4}	19%
Trichloromethane	kg	1.93×10^{-6}	3.07×10^{-6}	–59%	Uranium (ore)	kg	5.69×10^{-2}	7.43×10^{-4}	–31%
Vinylchloride	kg	1.19×10^{-5}	1.89×10^{-5}	–59%	Water	kg	3.57×10^{5}	6.41×10^{5}	–80%
					Wood	kg	3.35×10^{1}	4.99×10^{1}	–49%

* All environmental loads referred to the functional unit, 1 TJ of electricity produced.
[a] Sit. 1 = Scenario 1.
[b] Sit. 2 = Scenario 2.
[c] Diff. = (Scenario 1 – Scenario 2)/Scenario 1.
[d] Subst. = substance

TABLE 2.8
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)	kg	3.96×10^{-2}	2.99×10^{-2}	9.75×10^{-3}	–24.6
(a) Cadmium (Cd)	kg	3.98×10^{-2}	1.30×10^{-2}	2.68×10^{-2}	–67.4
(a) Carbon dioxide (CO_2, fossil	kg	2.57×10^{5}	1.63×10^{5}	9.43×10^{4}	–36.7
(a) Carbon monoxide (CO)	kg	3.01×10^{2}	1.33×10^{3}	-1.03×10^{3}	341.2
(a) Heavy metals (sum)	kg	7.17×10^{-1}	3.68	–2.97	413.7
(a) Hydrogen chloride (HCl)	kg	1.95×10^{2}	5.06×10^{1}	1.44×10^{2}	–74.1
(a) Nickel (Ni)	kg	6.51×10^{-2}	2.54×10^{-1}	-1.89×10^{-1}	290.3
(a) Nitrogen oxides (NO_x as NO_2)	kg	1.33×10^{3}	3.32×10^{2}	9.98×10^{2}	–75.0
(a) Particles (unspecified)	kg	1.61×10^{2}	7.72×10^{2}	-6.11×10^{2}	379.8
(a) Sulphur oxides (SO_x as SO_2)	kg	2.56×10^{2}	9.29×10^{2}	-6.73×10^{2}	262.9

marketing initiatives adapting solar PV panels, modern air conditioning system and an extra investment in aesthetic comfort and design features.

6. Explain the differences between reusing, recycling and disposal of a product. How would the corresponding life-cycle inventory change?
7. List all possible applications of LCA to the chemical industry.
8. Compare the use of virgin paper and recycled paper. Compare from an environmental point of view the use of recycled paper and the use of paper obtained from paper pulp. The recycled paper has been obtained from paper wastes transported in 16-t trucks from a distance of 1000 km. The total emissions of carbon dioxide, CO_2, and the generated solids in kg of them are used as comparison parameters. It is desirable to compare the derived effects of purchasing a 400-g pack of new writing paper at the corner stationery shop with the effects derived from driving to buy a similar pack of recycled paper at an establishment 1 km away. Only the ELs due to raw materials and transportation are considered. Use the following environmental loads:

EL	EL/kg of recycled paper	EL/kg of pulp paper	EL/km (car)	EL/tkm (16-t truck)
kg CO_2	1.03	1.61	2.16×10^{-1}	3.46×10^{-1}
kg solids	7.06×10^{-2}	1.73×10^{-1}	7.50×10^{-6}	7.19×10^{-2}

tkm is equivalent to a mass of 1 t (1000 kg) transported 1 km.

9. Compare the use of paper towels and an air dryer to drying the hands. Compare environmentally the use of paper towels for drying hands and the use of a hand-dryer supplied with electricity from a gas thermal power plant or from an eolic power plant. The considered comparison parameters are the emissions of carbon dioxide, CO_2, and sulfur dioxide, SO_2. Suppose that the weight of a paper towel is 7 g and that the electric hand-dryer is 2000 W and works for 30 s. With these data, determine the best way, from an environmental point of view, to dry the hands. The ELs related to the paper and the use of electricity are assumed to be the following:

EL	Units	Paper EL/kg	Electricity gas thermal power plant EL/kWh	Electricity eolic power plant EL/kWh
Lignite	kg	1.10×10^{-1}	1.21×10^{-3}	1.68×10^{-3}
Coal	kg	1.51×10^{-1}	7.06×10^{-2}	1.10×10^{-2}
Natural gas	Nm^3	7.97×10^{-2}	2.09×10^{-1}	2.25×10^{-3}
Crude oil	t	1.84×10^{-4}	2.35×10^{-6}	4.82×10^{-6}
Water	kg	2.60×10^{3}	3.40	5.15×10^{1}
CO_2 (air)	kg	1.46	1.42	3.3×10^{-2}
SO_x (air)	kg	1.04×10^{-2}	2.07×10^{-4}	1.37×10^{-4}
NO_x (air)	kg	3.03×10^{-3}	2.16×10^{-3}	7.1×10^{-5}
CH_4 (air)	kg	2.04×10^{-3}	1.99×10^{-3}	1.1×10^{-4}
HCl (air)	kg	1.30×10^{-4}	1.00×10^{-6}	5.8×10^{-6}

Electricity consumption: Ec (kWh); Ec = (2000 (J/s)/1000); (30/3600) = 1.67×10^{-2} kWh.

REFERENCES

Assies, J.A. (1992), State of art. Life-cycle assessment, 1–20, Workshop Report, Leiden, December 1991. SETAC.

Boustead, I. (1992), Eco-balance methodology for commodity thermoplastics, a report for the European Center for Plastics in the Environment, Brussels.

Boustead, I. (1993a), Eco-profiles of the European plastics industry, report 2, olefin feedstock sources, PWMI.

Boustead, I. (1993b), Eco-profiles of the European plastics industry, report 3: polyethylene and polypropylene, PWMI.

Boustead, I. (1993c), Eco-profiles of the European plastics industry, report 4: polystyrene, PWMI.

Boustead, I. (1994a), Eco-profiles of the European polymer industry, report 5: co-product allocation in chlorine plants, APME.

Boustead, I. (1994b), Eco-profiles of the European plastics industry, report 7: PVDC (polyvinylidene chloride), APME.

Boustead, I. (1994c), Eco-profiles of the European plastics industry, report 8: polyethylene terephthalate (PET), APME.

Boustead, I. (1974), Resource implications with particular reference to energy requirements for glass and plastic milk bottles, *J. Soc. Dairy Technol.*

Boustead, I. and Hancock, G. (1979), *Handbook of Industrial Energy Analysis*, Chichester: John Wiley and Ellis Horwood, New York, chap. 3.

Brezet, H. and van Hemel, C., *Ecodesign: A Promising Approach to Sustainable Production and Consumption,* UNEP, Paris, 1997, ISBN 928071631X.

BUWAL, Swiss Federal Office of Environment, Forest and Landscape (1991), Ecobalance of packaging materials, state of 1990, *Environmental Series* No. 132.

BUWAL, Swiss Federal Office of Environment, Forest and Landscape (1994), Comparative *Ecological Evaluation of Paint Substances in the Building Industry.* Vol. 1: *Method Environ Series,* No. 186, 77 pp.

Castells, F., Aelion, V., Abeliotis, K.G., and Petrides, D.P. (1995), An algorithm for life-cycle inventory, *AICHE Sym. Series Pollut. Prev. via Process Prod. Modifications*, 90(303), 151–160.

Castells, F., Alonso, J.C., and Garreta, J. (1997), Antecedentes y estado actual del Análisis de Ciclo de Vida, *Ingeniería Química*, 29(339), 163–171.

Club of Rome (1972), A blueprint for survival, *Ecologist*, 2(1), 1–44.

Curran, M.A. (1996), *Environmental Life-Cycle Assessment*, McGraw-Hill, New York.

Druijff, E.A. (1984), Milieurelevante produktinformatie (environmental information on products), CML mededelingen 15, Leiden, The Netherlands.

EC (European Commission) (1995), *ExternE — Externalities of Energy*, 6 vols., EUR 16520–162525, DG XII Science, Research and Development, Brussels.

EEA (Eurpean Environmental Agency) (1998), Life-cycle assessment (LCA) — a guide to approaches, experiences and information sources, *Environmental Issues Series* No. 6., European Environment Agency, Copenhagen.

Ekvall, T. and Finnveden, G. (2001), Allocation in ISO 14041. A critical review, *J. Cleaner Prod.*, 9, 197–208.

EMPA (1984), Eidgenössische Materialprüfungs- und Forschungsanstalt. Ökobilanzen von Packstoffen. Schriftenreihe Umweltschutz 24. Herausgegeben vom Bundesamt für Umweltschutz (BUS), Bern.

Fava, J., Consoli, F., Denison, R., Dickson, K., Mohin, T., and Vigon, B. (1993), *A Conceptual Framework for Life-Cycle Impact Assessment*, SETAC Press, Pensacola, FL.

Fava, J.A., Denison, R., Jones, B., Curran, M.A., Vigon, B., Selke, S., and Barnum, J. (1991), SETAC workshop report: a technical framework for life-cycle assessments, August 18–23, 1990. Smugglers Notch, VT. Washington, D.C., January 1991.

Fecker, I. (1992), How to calculate an ecological balance? Report No. 222, EMPA, St. Gallen.

Franklin Associates, Ltd. (1990), Resource and environmental profile analysis of foam polystyrene and bleached paperboard containers: final report, The Council for Solid Waste Solutions.

Franklin Associates, Ltd. (1991), Resource and environmental profile analysis of high-density polyethylene and bleached paperboard gable milk containers: final report, The Council for Solid Waste Solutions.

Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G, Hischier, R., Martin, A., Dones, R., and Gantner, U. (1996), *Ökoinventare von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz*, 3rd ed., ETH Zürich: Gruppe Energie-Stoffe-Umwelt, PSI Villigen: Sektion Ganzheitliche Systemanalysen.

Goedkoop, M.J. (1995), Eco-indicator 95 — final report, NOH report 9523, Pré Consultants, Amersfoort, The Netherlands.

Goedkoop, M. and Spriensma, R. (1999), Eco-indicator 99 — a damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands.

Graedel, T.E. (1995), *Industrial Ecology,* Prentice-Hall, Englewood Cliffs, NJ.

Guinée, J.B., Heijungs, R., van Oers, L., Van de Meent, D., Vermeire, T., and Rikken, M. (1996), LCA impact assessment of toxic releases — generic modeling of fate, exposure, and effect for ecosystems and human beings with data for about 100 chemicals, RIVM report no. 1996/21, Bilthoven, The Netherlands.

Guinée, J.B., Udo de Haes, H.A., and Huppes, G. (1993a), Quantitative life-cycle assessment of products, 1: goal definition and inventory, *J. Cleaner Prod.*, 1(1), 3–13.

Guinée, J.B., Heijungs, R., Udo de Haes, H.A., and Huppes, G. (1993b), Quantitative life-cycle analysis of products, 2: classification, valuation and improvement analysis, *J. Cleaner Prod.*, 1(2), 81–91.

Hauschild, M. and Wenzel, H. (1998), *Environmental Assessment of Products — Scientific Background* (Vol. 2), Chapman & Hall, London.

Heijungs, R., Guinée, J.B., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., and Wegener-Sleeswijk, A. (1992), Environmental life-cycle assessment of products — guide and backgrounds, technical report, CML, University of Leiden, The Netherlands.

Hertwich, E.G., Mateles, S.F., Pease, W.S., and McKone, T.E. (2001), Human toxicity potentials for life-cycle assessment and toxic release inventory risk screening, *Environ. Toxicol. Chem.*, 20, 928–939.

Houghton, J.T., Meira Filho, L.G., Bruce, J., Lee, H., Callander, B.A., Haites, E., Harris, N., and Maskell, K. (1995), *Climate Change 1994 — Radioactive Forcing of Climate Change and an Evaluation of the IPCC IS92 Emissions Scenarios*, Cambridge University Press, London.

Hunt, R.G. (1974), Resource and environmental profile analysis of nine beverage container alternatives, Midwest Research Institute for U.S. EPA (Contract 68-01-1848), Washington, D.C.

IFIAS (International Federation of Institutes for Advanced Study) (1974), Energy analysis workshop on methodology and conventions, Guldsmedshyttan, Sweden.

ISO 14040 (1997), Environmental Management — Life-Cycle Assessment — Principles and Framework, technical standard, ISO, Geneva, Switzerland.

ISO 14041 (1998), Environmental Management — Life Cycle Assessment — Goal and Scope Definition and Inventory Analysis, International Organization for Standardization, Geneva, Switzerland.

ISO 14042 (2000), Environmental Management — Life Cycle Assessment — Life Cycle Impact Assessment, International Organization for Standardization, Geneva, Switzerland.

ISO 14043 (2000), Environmental Management — Life Cycle Assessment — Life Cycle Interpretation, International Organization for Standardization, Geneva, Switzerland.

Itsubo, N. and Inaba, A. (2000), Definition of safeguard subjects for damage-oriented methodology in Japan, presented at the 4th Eco-Balance Conference, Tsukuba, Japan.

Lindfors, L.G., Christiansen, K., Virtanen, Y., Juntilla, V., Janssen, O.J., Rønning, A., Ekvall, T., and Finnveden, G. (1995), Nordic guidelines on life-cycle assessment, *Nord* 1995: 20, Nordic Council of Ministers, Copenhagen.

Marsmann, M. (1993), Standardization in the context of ISO/TC 207, report from the SPOLD members' meeting, Brussels, December 8, 1993.

Meadows, D.H., Meadows, D.L., Randers, J., and Behrens, W. (1972), *The Limits to Growth: a Report for the Club of Rome's Project on the Predicament of Mankind*, ISBN 0-330-241699, Universe Books, New York.

Owens, J.W., Amey, E., Barnthouse, E., Consoli, F., Coulon, R., Fava, J., Humphreys, K., Haaf, B., Hunt, B., Laibson, L., Mones, E., Noesen, S., Norris, G., Todd, J.A., Vigon, B., Weitz, K., and Young, J. (1997), *Life-Cycle Impact Assessment — State of the Art*, SETAC Press, Pensacola, FL.

Pedersen, B. (1993), *Environmental Assessment of Products*, UETP-EEE, IBSN951-9110-83-6.

Powell, J.C. and Pidgeon, S. (1994), Valuation within LCA: a multicriteria approach, in SETAC-Europe (Eds.), *Integrating Impact Assessment into LCA*, SETAC publication, Brussels.

Rabl, A., Spadaro, J.V., and McGavran, P.D. (1998), Health risks of air pollution from incinerators — a perspective, *Waste Manage. Res.*, 16(4), 368–388.

Sage, S. (1993), Industrielle Abfallvermeidung und deren Bewertung am Beispiel der Leiterplattenherstellung, Ph.D. thesis, Institute for Process Engineering, TU Graz, Austria.

Schmitz, S., Oels, H.J., and Tiedemann, A. (1994), Ökobilanz für Getränkeverpackungen, UBA-Texte 52/95, Umweltbundesamt, Berlin.

SETAC (Society of Environmental Toxicology and Chemistry) (1993), *Guidelines for Life-Cycle Assessment — a Code of Practice*, Sesimbra/Portugal SETAC workshop report, SETAC Press, Pensacola, FL.

Sonnemann, G.W., Castells, F., and Rodrigo J. (1999) Guide to the inclusion of LCA in environmental management, 8th Congrés Mediterrani d'Enginyeria Química, Expoquimia, Barcelona, 10–12, November 1999.

Steen, B. (1999), Systematic Approach to Environmental Priority Strategies in Product Development (EPS), Version 2000 – Models and Data of the Default Method, Chalmers University of Technology, Göteborg, Sweden.

Steen B. and Ryding S.O. (1992), The EPS enviro-accounting method, report for Federation of Swedish Industries, Swedish Environmental Research Institute, Göteborg.

STQ (Servei de Tecnologia Química) (1998), Análisis del ciclo de vida de la electricidad producida por la planta de incineración de residuos urbanos de Tarragona, technical report, Universitat Rovira i Virgili, Tarragona, Spain.

Tellus Institute (1992), The Tellus packaging study — inventory of material and energy use and air and water emissions from the production of packing materials, technical report, Tellus Institute, Boston.

Udo de Haes, H.A. (1996), *Towards a Methodology for Life-Cycle Impact Assessment*, SETAC-Europe publication, Brussels.

Udo de Haes, H.A and Lindeijer, E. (2002), The conceptual structure of life-cycle impact assessment, final draft for the Second Working Group on Impact Assessment of SETAC-Europe (WIA-2).

Udo de Haes, H.A. et al. (2002), UNEP/SETAC life-cycle initiative: background, aims and scope, *Int. J. LCA*, 7(4), 192–195.

Udo de Haes, H.A. et al. (2002b), *Towards Best Available Practice in Life-Cycle Impact Assessment,* SETAC Press, Pensacola, FL.

Udo de Haes, H.A., Joillet, O., Finnveden, G., Hauschild, M., Krewitt, W., and Mueller-Wenk, R. (1999), Best available practice regarding categories and category indicators in life-cycle impact assessment — background document for the Second Working Group on LCIA of SETAC–Europe, *Int. J. LCA*, 4, 66–74; 167–174.

UNEP (Industry and Environment) (1996), LCA — what is it and how to use it? Technical report, United Nations Environment Program, Paris.

UNEP/SETAC Life-Cycle Initiative (2001), Background paper of the UNEP/SETAC Life-Cycle Initiative, UNEP, DTIE, Paris.

U.S. EPA (1996), MSW Factbook 3.0. EPA, Office of Solid Waste, Washington, D.C.

Vigon, B.W., Tolle, D.A., Corneby, B.W., Lotham, H.C., Harrison, C.L., Boguski, T.L., Hunt, R.G., and Sellers, J.D. (1993), Life-cycle assessment: inventory guidelines and principles, conducted by Battelle and Franklin Associates for the EPA, Ltd. Office of Research and Development, EPA/600/R-92/245.

Wenzel, H., Hauschild, M., and Alting L. (1997), *Environmental Assessment of Products*. Vol. 1, *Methodology, Tools and Case Studies in Product Development*, Chapman & Hall, London.

3 Life-Cycle Impact Assessment

3.1 INTRODUCTION

The life-cycle inventory offers product-related environmental information consisting basically of a quantified list of environmental loads (raw material consumption, air and water emissions, wastes, etc.) that give the amount of pollutants to be assigned to the product. However, the environmental damage associated with them is not yet known.

Let us consider, for example, well-known air pollutants such as sulfur dioxide, SO_2, nitrogen dioxide, NO_2, and hydrogen chloride, HCl, that generate an environmental impact known as acid rain. The capacity of these pollutants to acidify the atmosphere can be measured by the potential to generate H^+ protons, so the acid concentration could be multiplied by a corresponding factor to obtain a global value of H^+ protons equivalent. In this way an environmental impact category has been measured based on inventory data. The same occurs with air emissions: carbon dioxide, methane, nitrogen oxides, halocarbons, etc. contribute to Earth's global warming and cause the well-known greenhouse effect, measured in CO_2 equivalents. Thus a new type of impact category, global warming potential, is introduced from inventory data.

Thus, the life-cycle impact assessment (LCIA) is introduced as the third step of life-cycle assessment (LCA), described in ISO 14042 (2002) and further outlined in ISO/TR 14047 (2002). The purpose of LCIA is to assess a product system's life-cycle inventory (LCI) to understand its environmental significance better. Thus, LCIA provides information for interpretation — the final step of the LCA methodology.

Jointly with other LCA steps, the LCIA step provides a wide perspective of environmental and resource issues for product systems by assigning life-cycle inventory results to impact categories. For each impact category, impact potentials are selected and category indicator results are calculated. The collection of these results defines the LCIA profile of the product system, which provides information on the environmental relevance of resource use and emissions associated with it. In the same way as LCA as a whole, LCIA builds up a relative approach based on the functional unit.

On the other hand, to compare the potentials for different impacts, it is necessary to evaluate the seriousness of the impact categories relative to one another. This can be expressed by a set of weighting factors — one factor per impact category within each of the main category groups. The weighted impact potential, WP(j), can be calculated by multiplying the normalized impact potential or resource consumption, NP(j), by the weighting factor, WF(j), associated with the impact category.

3.2 PHASES OF LIFE-CYCLE IMPACT ASSESSMENT

The general framework of the LCIA phase is composed of several mandatory elements that convert life-cycle inventory results into indicator results. In addition, there are optional elements for normalization, grouping and weighting of the indicator results and data quality analysis techniques. The LCIA phase is only one part of a total LCA study and should be coordinated with other phases of LCA. An overview of the mandatory and optional elements in LCIA is given in Figure 3.1.

The mandatory LCIA elements are (ISO 14042, 2002):

- Selection of impact categories indicators and models
- Classification of environmental loads within the different categories of environmental impact
- Characterization of environmental loads by means of a reference pollutant typical of each environmental impact category

These categories, plus the level of detail and methodology, are chosen depending on the goals and scope of the research.

Optional elements and information can be used depending on the goal and scope of the LCA study (ISO 14042, 2002):

- Calculating the magnitude of category indicator results relative to reference values (normalization) means that all impact scores (contribution of a product system to one impact category) are related to a reference situation.
- The indicators can be grouped (sorted and possibly ranked).
- Weighting (across impact categories) is a quantitative comparison of the seriousness of the different resource consumption or impact potentials of

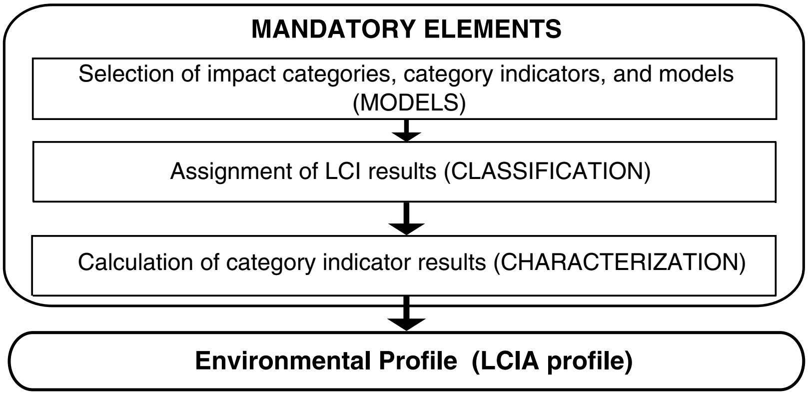

FIGURE 3.1 Mandatory elements of LCIA according to ISO 14042 (2002) (available at www.afnor.fr).

the product, aimed at covering and possibly aggregating indicator results across impact categories.

- Data quality analysis serves to better understand the reliability of the LCIA results better.

The use of models is necessary to derive the characterization factors. The applicability of these 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 characterization factors can be selected.

Models reflect the cause–effect chain, also called environmental mechanism or impact pathway, by describing the relationship among the life-cycle inventory results, indicators and, if possible, category endpoints or damage indicators. For each impact category, the following procedure is proposed in ISO 14042 (2002):

1. Identification of the category endpoints
2. Definition of the indicator for given category endpoints
3. Identification of appropriate LCI results that can be assigned to the impact category, taking into account the chosen indicator and identified category endpoints
4. Identification of the model and the characterization factors

Figure 3.2 illustrates the relationship among the results of the life-cycle inventory analysis, indicators and category endpoints for one impact category for the example of acidification. It clearly shows where a model is needed. These items are explained in detail in this chapter.

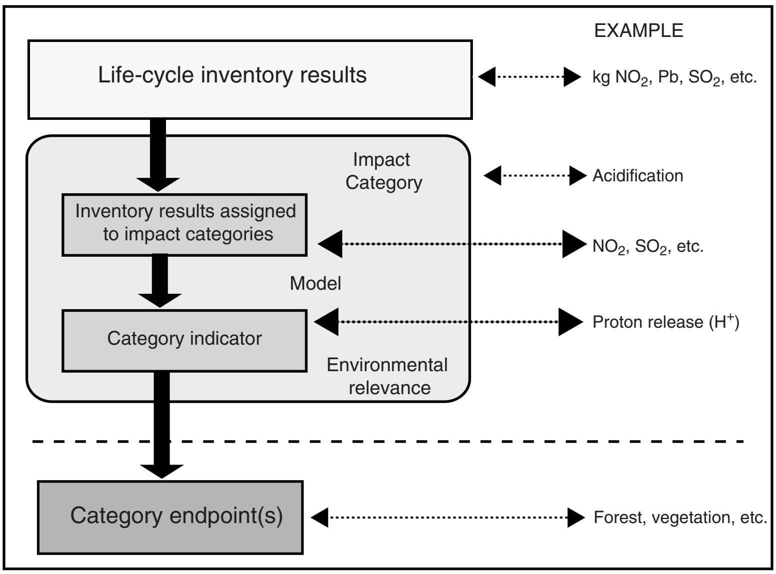

FIGURE 3.2 The concept of indicators (ISO 14042, 2000) (available at www.afnor.fr).

3.3 IMPACT CATEGORIES

An impact category is defined as a class representing environmental issues of concern into which life-cycle inventory results may be assigned. As has been previously mentioned, Udo de Haes et al. (1999) have proposed classifying impacts in input- and output-related categories. Input refers to environmental impacts associated with material or energy inputs to the system and output corresponds to damages due to emissions or pollutants, vibrations, or radiation. Table 3.1 gives an overview of input and output impacts currently used in LCIA with a proposal of possible indicators.

Some of the impact categories mentioned in Table 3.1, such as climate change and stratospheric ozone depletion, have a global effect; others, such as photo-oxidant formation or acidification, have a local effect. This highlights the need for spatial differentiation in the fate and exposure analysis in different impact categories. Figure 3.3 shows the global–local impacts for the different impact categories, each of which is described next.

3.3.1 Extraction of Biotic and Abiotic Resources

This impact category includes the extraction of different types of nonliving material from the natural environment. It is possible to distinguish three different subcategories: extraction of (1) deposits (e.g., fossil fuels and mineral ores), (2) funds (e.g., groundwater, sand and clay), and (3) flow resources (e.g., solar energy, wind and surface water). Examples of used indicator categories are: rareness of resources, energy content of resources, mineral concentrations, degree of use of flow resources in relationship to the size of the flow, total material requirement, and indicators

TABLE 3.1
Impact Categories and Possible Indicators

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 GWP
Stratospheric ozone depletion	kg CFC-11 as equivalence unit for ODP
Human toxicity	HTP
Eco-toxicity	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 AP
Nutrification	Stoichiometric sum of macronutrients as equivalence unit for the nutrification potential (NP)

Source: Udo de Haes, H.A. et al., *Int. J. LCA*, 4, 66–74, 167–174, 1999. With permission.

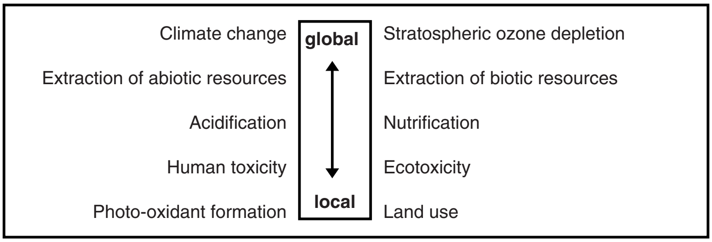

FIGURE 3.3 The need for spatial differentiation in different impact categories. (From UNEP DTIE, 2003).

related to other categories, such as energy requirement or land use. Extraction of biotic resources is mainly related to the extraction of specific types of biomass from the natural environment. The rareness and regeneration rate of the resources is generally used as indicator (SETAC-Europe, 1999).

3.3.2 Climate Change: Global Warning Potential

Most of the radiant energy received by the Earth as short-wave radiation is reflected directly, re-emitted from the atmosphere, or absorbed by the Earth's surface as longer infrared wave radiation (IR). This natural greenhouse effect is increased by manmade emissions of substances or particles that can influence the Earth's radiation balance, thus raising the planet's temperature.

Many of the substances emitted to the atmosphere as a result of human activities contribute to this manmade greenhouse effect and must be classified in this impact category. Listed in order of importance, they are (Hauschild and 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 a substance relative to those of the other gases. The Intergovernmental Panel of Climate Change (IPCC) has developed a characterization factor system that can weight the various substances according to their efficiencies as greenhouse gases (Houghton et al., 1995). This system can be used in political efforts to optimize initiatives to counter manmade global warming.

The system classifies these substances according to their global warming potential (GWP), which is calculated as the anticipated contribution to warming over a chosen time period from a given emission of the substance, divided by the contribution to warming from emission of a corresponding quantity of carbon dioxide (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: the emission of the greenhouse gas expressed as CO_2 equivalents.

TABLE 3.2
GWP for Some Substances Depending on the Time Horizon

Substance	Formula	Lifetime years	GWP ($kg\ CO_2$ eq./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

Source: Data taken from Houghton et al., 1995.

CO_2 was chosen as a reference substance by the IPCC because it makes the most significant contribution to the manmade greenhouse effect. The expected contribution in terms of warming from a greenhouse gas is calculated based on knowledge of its specific IR absorption capacity and its expected lifetime in the atmosphere. The GWP is internationally accepted and well documented, and provides characterization factors for the substances encountered in an LCA. Table 3.2 presents examples of GWP values for direct contribution of three substances mentioned previously: CO_2, CH_4 and N_2O.

3.3.3 Stratospheric Ozone Depletion

Human activities have caused an increase of substances as different chloride and bromide-containing halocarbons, especially CFCs, tetrachloromethane, 1,1,1-trichloroethane, HCFCs, halons and methyl bromide, involved in the breakdown of ozone in the stratosphere. A common characteristic of these compounds is that they are chemically stable because they can survive long enough to reach the stratosphere, where they can release their content of chlorine and bromide under the influence of UV radiation (Solomon and Albritton, 1992).

The Earth's atmosphere receives ultraviolet radiation from the sun but not at full intensity. Ozone molecules in the atmosphere absorb large quantities of UV radiation. The reduction of the ozone layer supposes that more UV radiation reaches the surface of the earth and causes damage, especially to plants, animals and humans.

Table 3.3 presents a list of factors to calculate the stratospheric ozone depletion potential of different chemical substances expressed in kilograms of CFC-11 (Freon 11) equivalent as a reference.

3.3.4 Human Toxicity

Chemical emissions such as heavy metals, persistent organic substances (POPs), volatile organic compounds (VOCs) and others may lead to direct human exposure (inhalation or drinking water) or to indirect exposure (food consumption). Apart from their toxicity, these substances are all persistent as common characteristics (low degradability in the environment and ability to bioaccumulate). In contrast to other impact categories, e.g., global warning and ozone depletion, no common

TABLE 3.3
Characterization Values for Ozone Layer Depletion

	Ozone layer depletion		
Medium	**Chemical Substance**	**u**	**Value**
Air	1,1,1-trichloroethane	kg	1.20×10^{1}
Air	CFC (hard)	kg	1.00
Air	CFC (soft)	kg	5.50×10^{-2}
Air	CFC-11	kg	1.00
Air	CFC-113	kg	1.07
Air	CFC-114	kg	8.00×10^{-1}
Air	CFC-115	kg	5.00×10^{-1}
Air	CFC-12	kg	1.00
Air	CFC-13	kg	1.00
Air	Halon-1201	kg	1.40
Air	Halon-1202	kg	1.25
Air	Halon-1211	kg	4.00
Air	Halon-1301	kg	1.60×10^{1}
Air	Halon-2311	kg	1.40×10^{-1}
Air	Halon-2401	kg	2.50×10^{-1}
Air	Halon-2402	kg	7.00
Air	HCFC-123	kg	2.00×10^{-2}
Air	HCFC-124	kg	2.20×10^{-2}
Air	HCFC-141b	kg	1.10×10^{-1}
Air	HCFC-142b	kg	6.50×10^{-2}
Air	HCFC-22	kg	5.50×10^{-2}
Air	HCFC-225ca	kg	2.50×10^{-2}
Air	HCFC-225cb	kg	3.30×10^{-2}
Air	Methyl bromide	kg	6.00×10^{-1}
Air	Tetrachloromethane	kg	1.08

Source: Goedkoop (1995).

internationally accepted equivalency factors for toxic compounds express the substances' "impact potentials." For the calculation of equivalency factors, considerations about fate and transport, exposure assessment and human toxicity have been considered.

A frequently used indicator for evaluating human health effects of a functional unit is human toxicity potential (HTP) (Hertwich et al., 2001; Guinée et al., 1996). HTP is a site-generic impact potential that is easy to apply; however, it has a limited environmental relevance because it is based on a multimedia environmental fate model that assumes uniformly mixed environmental compartments. In other words, it represents the behavior of chemicals in a uniform world model environment.

Two HTP methods developed by the Center of Environmental Sciences at Leiden University (CML) (Heijungs et al., 1992) and within the Danish Environmental

Design of Industrial Products (EDIP) (Hauschild and Wenzel, 1998) will be considered in the case study (MSWI). 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 method is dimensionless. The HTP for every pollutant "*p*" (HTP_p) is calculated using the human toxicity factor (HTF_p) for every pollutant and the mass of every pollutant (M_p) and shown in the following expression:

$$HTP_p = HTF_p \cdot M_p \tag{3.1}$$

The HTF_p is expressed in units of m^3/kg in the EDIP method (Hauschild and 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 as seen in the next expression (expression of overall HTP for the functional unit):

$$HTP = \sum HTP_p \tag{3.2}$$

Table 3.4 shows the HTP for the pollutants considered in the case study (MSWI). It should be mentioned that ozone, nitrate and sulfate are not considered in these HTFs due to the unavailability of the mass of these substances in the life-cycle inventory because they are not directly emitted but formed during dispersion into the atmosphere. Particulate matter with apparent diameter lower than 10 μm (PM_{10}) is also not included because no HTF is available.

TABLE 3.4
Human Toxicity Potential from the CML and EDIP Methods for Different Substances

Pollutant	CML (–/kg)	EDIP (m^3/kg)
As	4700	$9.5 \cdot 10^9$
Benzo(a)pyrene	17	$5.0 \cdot 10^{10}$
Cd	580	$1.1 \cdot 10^{11}$
Ni	0.014	$6.7 \cdot 10^7$
NO_x	0.78	$2.0 \cdot 10^6$
SO_2	1.2	$1.3 \cdot 10^6$

Sources: CML — Heijungs, R. et al., Environmental life-cycle assessment of products — guide and backgrounds, technical report, CML, University of Leiden, The Netherlands, 1992; EDIP — Hauschild, M. and Wenzel, H., *Environmental Assessment of Products — Scientific Background*, Vol. 2, Chapman & Hall, London, 1998.

3.3.5 Eco-Toxicity

Eco-toxic substances are those toxic to organisms in a manner that affects the functioning and structure of the ecosystem in which the organism lives and, as result, affects the health of the ecosystems. They are characterized by their persistence (low degradability) in the environment and their ability to bioaccumulate in organisms. Substances such as toxic heavy metals (Cd, Pb, Hg), persistent organic compounds (dioxins and furans, PCDD/Fs, polycyclic aromatic hydrocarbons, PCBs, etc.), and organic substances (PVC, etc.) that are emitted into the environment can accumulate in organisms and cause different types of damage. The target system is not one organism as in human toxicity, but a variety of organisms (fauna and flora, entire ecosystems). This makes the assessment even more complex.

In contrast to other environmental impact categories in the LCIA, the impacts of these types of substances are not based in one individual mechanism but in a large number, such as genotoxicity, inhibition of specific enzymes, etc.

3.3.6 Photo-Oxidant Formation

Human activities can increase air concentrations of photo-oxidant substances that can affect the health of living organisms and human beings. These substances can arise via photochemical oxidation of volatile organic compounds (VOCs) and carbon monoxide (CO) emitted by human activities in the troposphere.

The photo-oxidants include a large number of unstable substances formed when VOCs react with different oxygen compounds and oxides of nitrogen (NO_x). The most important oxygen compounds are hydroxyl radicals, OH·. Among the most important photo-oxidants are ozone and peroxyacetyl nitrate (PAN). The transformation of VOCs and CO to ozone requires, apart from the reactive forms of oxygen, sunlight and NO_x, which have a catalytic effect. The potential contribution to photochemical ozone formation is described by its maximum incremental reactivity (MIR) in the American literature and by its photochemical ozone creation potential (PCP) in Europe.

3.3.7 Acidification

Combustion processes contribute greatly to the air emission of contaminants as NO_x and SO_2. In contact with water these oxides are converted to acids (nitric acid, HNO_3) and sulfuric acid (H_2SO_4). Once deposited (by dry and wet deposition), these chemicals may lead to exceeding the acid buffer capacity of the soil and water, generating degradation of terrestrial and aquatic ecosystems. The presence of NH_3 (emitted primarily from agricultural soil) increases the potential uptake of SO_2 in drops of water in clouds and rains by the formation of $(NH_4)_2SO_4$, and thus affects the forest in which SO_2 is deposited.

The principal effect of acidification of the environment is the loss of health especially among conifers in many forests. The acidification of lakes can lead to dead fish. On the other hand, metals, surface coatings and mineral building materials exposed to air conditions are attacked by the air and acid rain, leading to patrimonial and economic loss of historic monuments.

3.3.8 Nutrification

Emission of salt nutrients by human activities involves a big impact in the environment. The eutrophication process in lakes, watercourses, and open coastal waters is due to excessive quantities of nutrient salts emitted by man and consequently results in increased production of planktonic algae and aquatic plants, which leads to a reduction in the quality of water. The process of decomposition of dead algae consumes important oxygen and causes with a loss of water quality. Agriculture has been identified as the most significant source of nitrogen loading. Wastewater treatment plants and fish farming are the most predominant causes of phosphorus emissions.

3.4 AREAS OF PROTECTION

The set of category indicators resulting from the life-cycle inventory configures and defines the environmental diagnosis associated with product manufacture or any other activity. The impact indicators are associated with environmental damages corresponding to areas of protection (AoP) or sectors of the environment to be protected.

In the first report of the Second SETAC Working Group on Life-Cycle Impact Assessment (Udo de Haes et al., 1999), an AoP was defined as a class of category endpoints. In ISO 14042 three of these classes are mentioned: human health, natural environment and natural resources. Another term used is the expressive "safeguard subject" introduced by Steen and Ryding (1992). It is important to note that these two terms convey the same message: they relate to the category endpoints as physical elements, not as societal values. Thus, following this terminology, the human right to life or economic welfare cannot be an AoP or a safeguard subject; neither can 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 3.5 gives an

TABLE 3.5
Assignment of Societal Values to AoP

Societal values	Human/manmade	Natural
Intrinsic values	Human health Manmade environment (landscapes, monuments, works of art)	Natural environment (biodiversity and natural landscapes)
Functional values	Manmade environment (materials, buildings, crops, livestock)	Natural environment (natural resources) Natural environment (life support functions)

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overview of the AoPs with underlying societal values, as presented by Udo de Haes and Lindeijer (2001). Because the AoPs are the basis for the determination of relevant endpoints, their definition implies value choices. Thus, there is no one scientifically correct way to define a set of AoPs (Udo de Haes et al., 2002).

Udo de Haes and Lindeijer (2001) propose to differentiate among the sub-AoPs' life support functions, natural resources and biodiversity, and natural landscapes within the AoP natural environment. Life support functions concern the major regulating functions of the natural environment, which enable life on Earth (human and nonhuman). These particularly include the regulation of the Earth's climate, hydrological cycles, soil fertility and the bio–geo–chemical cycles. Like manmade environments (materials, buildings, crops, livestock) and natural resources, the life support functions are of functional value for society. From a value perspective, these are fundamentally of another nature than those of AoPs with intrinsic value to society, particularly those connected with human health, biodiversity and natural landscapes, works of art, monuments and manmade landscapes. An overview of the classification of AoPs according to societal values is presented in Figure 3.4.

3.5 MIDPOINT AND ENDPOINT INDICATORS

The terms midpoint and endpoint refer to the level within the environmental mechanism at which the respective effects are characterized. In general it is assumed that an indicator defined closer to the environmental intervention will result in more certain modeling and that an indictor further away from the environmental intervention will provide environmentally more relevant information (i.e., more directly linked to society's concerns and the areas of protection). Although midpoints and endpoints can be overlapped in same cases, midpoint indicators are used to measure a substance's potency of effect, which in most cases is characterized by using a threshold, and does not take into account the severity of the expected impact. Figure 3.5 shows a schematic illustration of the definition of midpoint and endpoint levels (Olsen et al., 2001).

According to Udo de Haes and Lindeijer (2001), historically, the midpoint approaches have set the scene in LCIA; some prominent examples include 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 (2002).

Since the middle of the 1990s the endpoint approach has been set on the agenda (Udo de Haes and Lindeijer, 2001). 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 or ozone depletion, as well as other categories traditionally addressed using midpoint category indicators. The endpoint approach already has a longer history, particularly in the EPS (environmental priority strategy) approach from Steen and Ryding (Steen and Ryding, 1992; Steen, 1999); however, it has received a strong impetus from the

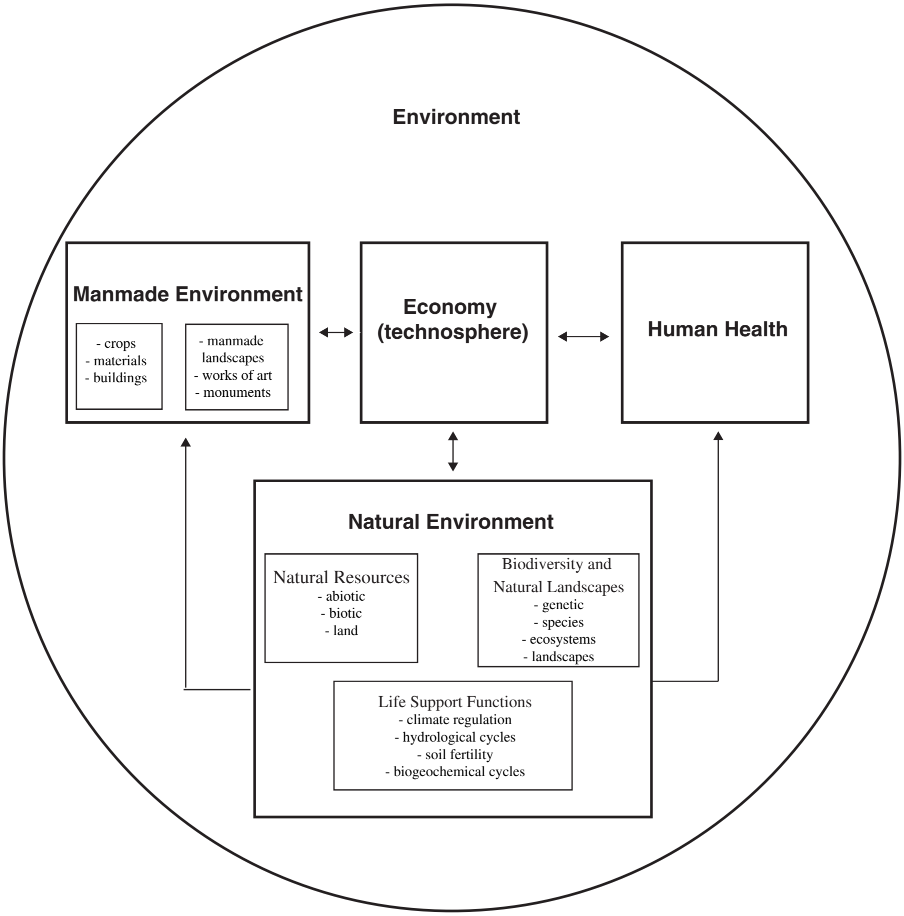


FIGURE 3.4 Classification of AoPs according to societal values. Arrows pointing both ways express interactions between economy and AoPs. Other arrows indicate main relationships between AoPs. (Reprinted with permission from Udo de Haes, H.A. and Lindeijer, E., in *Towards Best Available Practice in Life-Cycle Impact Assessment*, Udo de Haes et al., Eds., ©2001 SETAC Press, Pensacola, FL.)

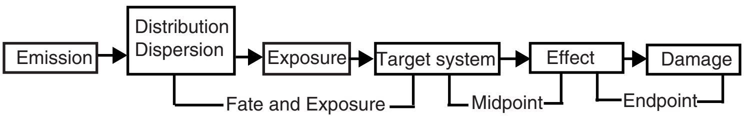


FIGURE 3.5 Schematic illustration of the definition of midpoint and endpoint levels. (Olsen, S.I. et al., Reprinted with permission from *Environ. Impact Assess. Rev.*, 21, 385–404, ©2001 with permission from Elsevier.)

eco-indicator 99 approach (Goedkoop and Spriensma, 1999). In Japan, impact assessment models are currently developed according to this approach (Itsubo and Inaba, 2000), which starts from the main values in society, connected with areas of protection. From these values and connected endpoints the modeling goes back to the emissions and resource consumption (Udo de Haes and Lindeijer, 2001).

Figure 3.6 shows the steps that can be involved if a practitioner wishes to take an LCA study from the inventory stage to valued scores via midpoints and endpoints in the impact assessment. Not all possible environmental loads can be considered in the inventory because data are not available for all of them. Based on the inventory table, two different routes to arrive at valued scores, representing the routes taken when using midpoint and endpoint approaches, are presented (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 impacts are expressed in the form of endpoints by relating the midpoints to endpoints or by modeling effects directly from the inventory to the endpoints. Then several endpoints can be aggregated to a valued score if the selected weighting scheme allows it.

At the moment, the availability of reliable data and sufficiently robust models to support endpoint modeling remains quite limited. Uncertainties may be very high beyond well-characterized midpoints. As a result, a misleading sense of accuracy and improvement over the midpoint indicators can be obtained. One of the biggest differences between midpoint and endpoint approaches is the way in which the environmental relevance of category indicators is taken into account. For midpoint approaches, the environmental relevance is presented as a qualitative relationship, while endpoint modeling can facilitate more informed and structured weighting (Bare et al., 2000; UNEP DTIE, 2003).

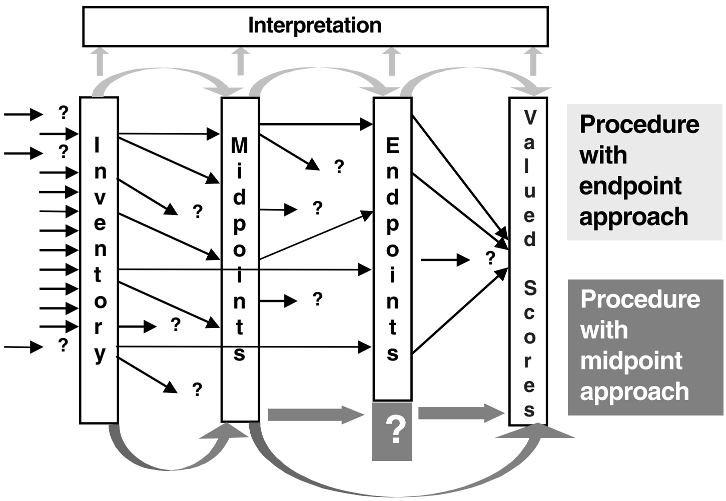


FIGURE 3.6 Some basic differences between the midpoint (lower row of swinging arrows) and the endpoint approach (upper row of swinging arrows). (From Bare, J.C. et al., *Int. J. LCA*, 5, 319–326, 2000.With permission)

3.6 WEIGHTING: SINGLE INDEX APPROACHES

3.6.1 INTRODUCTION

Weighting (in ISO terminology) or valuation (in SETAC workgroup terminology) is the phase of LCIA that involves formalized ranking, weighting and, possibly, aggregation of the indicator results into a final score across impact categories. Weighting or valuation inherently uses values and subjectivity to derive, respectively, a rank order and then weighting factors with values supporting the aggregation into a final score. Three types of weighting along similar lines are used:

- Monetary methods, such as mediation costs, willingness to pay, etc.
- Sustainability and target methods, such as in the distance-to-target procedure
- Social and expert methods

The results of an LCIA in the impact categories explained earlier can be difficult to interpret in certain cases because they may be contradictory. In these cases it would be helpful to have one single score.

The prioritization of impact categories often depends on political targets or business strategies. Weighting is necessary to obtain a single index of environmental performance of a functional unit. However, the weighting across impact categories is the most critical and controversial step in LCIA, i.e., a quantitative comparison of the seriousness of the different resource consumption or impact potentials of the product, aimed at covering and possible aggregating indicator results across impact categories.

The weighting methods in LCIA to obtain a single index can be distinguished and classified according to five types of concepts (Udo de Haes, 1996). Table 3.6 presents a description of these concepts, indicating their advantages and disadvantages. In this frame, no simple truth can decide what works best.

Examples for the proxy approach are the sustainable process index (SPI; Sage, 1993) and the material-intensity per-service unit (MIPS) (Schmidt-Bleek, 1994). MIPS is a measure of the environmental impact intensities of infrastructures, goods, and services. Materials and fuels are aggregated by mass and energy content. Important cases for the distance-to-target methods are eco-scarcity (Braunschweig et al., 1994), eco-indicator 95 (Goedkoop, 1995) and EDIP (Hauschild and Wenzel, 1998). Eco-scarcity is a Swiss method that has also been adapted by Chalmers University of Technology to suit Swedish conditions. Its units are ECO points per gram of emission or per MJ of energy. Panel approaches have been used, for instance, by the German EPA (Schmitz et al., 1995) and in the eco-indicator 99 weighting step (Goedkoop and Speedesma, 1999). A similar approach, the multicriteria evaluation (MCE), has been proposed for LCA by Powell and Pidgeon (1994). The abatement technology concept has been used in the method developed by the Tellus Institute (1992). It consists of an evaluation of internal environmental costs by means of the most adequate technology to fulfill the legal requirements. Monetization has been used as a weighting scheme in some of the damage-oriented methods like environ-

TABLE 3.6
Comparison of Concepts for Weighting 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.	Parameter is only a bad approximation of 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	No accepted reference value for 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	Result depends on composition of the panel and/or selected individuals
IV Abatement technology	Efforts to reduce pollution by technological means as reference	Internal	The efforts can be expressed by costs that are known	Internalized costs do not correspond to external costs
V Monetization	Expression of environmental damages in monetary values	External	Attempt to estimate the actual damage costs	External costs can only be estimated

mental priority strategies (EPS) (Steen, 1999) and in the uniform world model (UWM; Rabl et al., 1998).

In addition to the weighting scheme used, single index approaches can be differentiated according to whether impact potentials are the basis for the weighting. For instance, this is the case for eco-indicator 95 and EDIP, but not for Tellus and eco-scarcity, in which directly weighting factors are applied.

In the ongoing methodology development of LCIA, panel methods are increasingly important; a tendency also exists to reflect the emission–effect relation more accurately. In turn, proxy indicators "energy" and "mass displacement" (as a measure of energy and resource intensity) and monetization methods based on damage or abatement cost are also acquiring relatively increasing importance.

Next, the eco-indicator 95 will be further explained as a single index method.

3.6.2 Eco-Indicator 95 as Example of a Single Index Approach

Eco-indicators are numbers that express the total environmental load of a product or process. The eco-indicator 95 (Goedkoop, 1995) is one of the weighting methods based on the "distance to target" in the same way as the similarly structured EDIP method (Hauschild and Wenzel, 1998). The steps to achieve a weighting are:

1. Determine the relevant effects caused by a process or product.
2. Determine the extent of the effect; this is the normalization value. Divide the effect by the normalization value. This step determines the contribution of the product to the total effect. This is done because it is not the effect that is relevant but rather the degree to which the effect contributes to the total problem. An important advantage of the normalization stage is that all the contributions are dimensionless.
3. Multiply the result by the ratio between the current effect and the target value for that effect. The ratio, also termed the reduction factor, may be seen as a measure of the seriousness of the effect.
4. Multiply the effect by a so-called subjective weighting factor to link fatalities, health and ecosystem impairment.

An overview of the principle of eco-indicator 95 is given in Figure 3.7. The problem, of course, lies in determining the weighting factors — the subjective damage assessment phase. The eco-indicator 95 uses the so-called distance-to-target principle to determine weighting factors. The underlying premise is that a correlation exists between the seriousness of an effect and the distance between the current and target levels. Thus, if acidification must be reduced by a factor of 10 in order to achieve a sustainable society and smog by a factor of 5, acidification is regarded as twice as serious. The reduction factor is the weighting factor.

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 which factor the problem must be reduced to reach an acceptable level. Table 3.7 lists the weighting factors and the criteria applied.

3.7 DAMAGE-ORIENTED METHODS

3.7.1 Introduction

All damage-oriented methods 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, for example, this means not as HTP but as cancer cases. In order to illustrate the theory behind these damage-oriented methods, the eco-indicator 99 methodology (Goedkoop and Spriensma, 1999) and the uniform world model (Rabl et al., 1998) are introduced. Another method based on the same principles has been developed by Steen (1999).

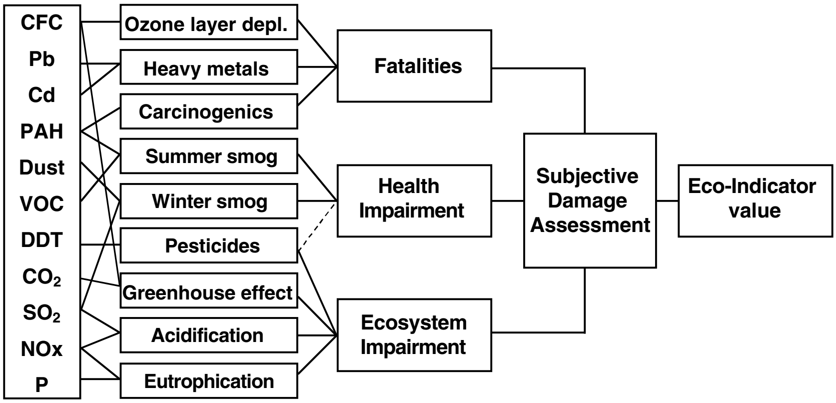


FIGURE 3.7 Overview of the structure of eco-indicator 95. (From Goedkoop, M.J., Eco-indicator 95 — final report, NOH report 9523, Pré Consultants, Amersfoort, The Netherlands, 1995. With permission.)

TABLE 3.7
Weighting Factors in Eco-Indicator 95

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 among asthma patients and the elderly, prevention of agricultural damage
Winter smog	5	Occurrence of smog periods, health complaints, particularly among 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 impacts people (see airborne)
Carcinogenic substances	10	Probability of 1 fatality per year per million people

Source: Goedkoop, M.J., Eco-indicator 95 — final report, NOH report 9523, Pré Consultants, Amersfoort, The Netherlands, 1995. With permission.

The described approaches use particular weighting methods, especially to evaluate the damage to human health. Eco-indicator 99 applies the cultural theory and disability adjusted life years (DALY) concept, using estimates of the number of years lived disabled (YLD) and years of life lost (YOLL), while the Uniform World Model (UWM) is based on monetization of environmental damages.

3.7.2 Cultural Theory

Hofstetter (1998) proposes using the sociocultural viability theory (Thompson et al., 1990), called cultural theory, to deal with the problem of modeling subjectivity. Based on this theory, Goedkoop and Spriensma (1999) distinguish five extreme value systems, which are illustrated in Figure 3.8. The most important characteristics of the five extreme archetypes can be summarized in the following way:

1. **Individualists** are 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** have a strong link to the group, but a weak link to their grid. No internal role differentiation exists in this environment and relations between group members are often ambiguous; conflicts can occur easily.
3. **Hierarchists** have a strong link to group and grid. In this environment people control others and are subject to control by others. This hierarchy creates a high degree of stability in the group.

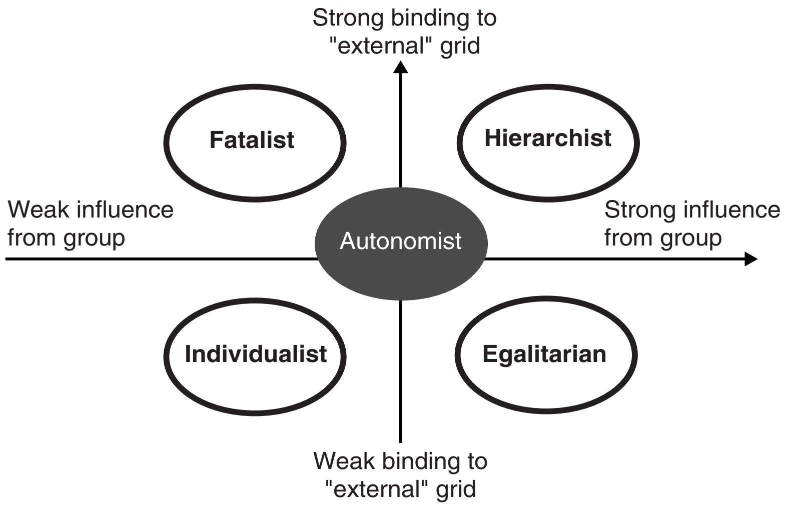


FIGURE 3.8 The grid-group dependency of the five extreme archetypes distinguished in cultural theory. The autonomist has no fixed position in this figure because it does not have social relations and should be seen as floating over the archetypes. (From Goedkoop, M. and Spriensma, R., The eco-indicator-99. A damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands, 1999. With permission.)

TABLE 3.8
Attitudes Corresponding to the Three Cultural Perspectives Used in the Eco-Indicator 99

Archetype	Time perspective	Manageability	Required level of evidence
Hierarchist	Balance between short and long term	Proper policy can avoid many problems	Inclusion based on consensus
Individualist	Short term	Technology can avoid many problems	Only proven effects
Egalitarian	Very long term	Problems can lead to catastrophe	All possible effects

Source: Goedkoop, M. and Spriensma, R., The eco-indicator-99. A damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands, 1999. With permission.

4. **Fatalists** have a strong link to grid, but not to group. Although these people act as individuals they are usually controlled by others who influence their conception of destiny.
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 as to modeling choices that must be made. Therefore, they are relevant for decision-making (Table 3.8). The last two archetypes cannot be used. The fatalist tends to have no opinion on such preferences because he is guided by what others say and the autonomist cannot be captured in any way because he thinks independently.

Only the hierarchist, egalitarian and individualist perspectives are relevant for decision-making and can be defined as the default scenarios, which are proposed as extreme cases if no other scenarios based on more specific information are available (Weidema et al., 2002).

The real value of sociocultural viability theory is that a wide range of basic attitudes and assumptions can be predicted for the three remaining extreme archetypes: hierarchist, individualist and egalitarian. (Figure 3.8 specifies some of the many different characteristics per archetype.) Therefore, the eco-indicator 99 methodology uses these three perspectives to facilitate analysis of the relative contribution of the different damage category indicators to one endpoint.

3.7.3 The DALY Concept for Human Health Impact

The DALY (developed by Murray and Lopez, 1996, for the World Health Organization [WHO] and World Bank) aggregates health effects leading to death or illness. Health effects leading to death are described using the years of life lost (YOLL) indicator, which 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 WHO. These statistics can show 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 (see Chapter 4), it can be calculated how many years of life are lost due to the concentration increase of a certain pollutant.

The DALY concept includes not only 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 decreased quality of life and for pain and suffering. Cough, asthma or hospitalizations due to different pollutants refer to this indicator. The morbidity health effects are expressed in years of disability (YLD). Value choices must be made to weight the pain or 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 1 year of suffering is supposed to be as severe as half a year of premature death. The DALY indicator is, then, the result of the addition of both indicators, with DW the relative disability weight and L representing the duration of the disability:

$$DALY = YOLL + YLD = YOLL + DW \cdot L \tag{3.3}$$

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

The value of YOLL or YLD does not depend only on the pollutant and the type of disease. Because 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 and a year of life lost at the age of 60 are not equally appreciated in every socioeconomic perspective, according to the cultural theory. For instance, one cultural theory discounts years of life lost in the same way that discounting is done in finances. Therefore, a year of life lost in the future is worth less than a year of life lost today. Another cultural perspective judges every YOLL to be equally important independently of the age when it occurs (Hofstetter, 1998). 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.

3.7.4 Monetization of Environmental Damages

Lately, acceptance of the approach of valuing health and environmental impacts in monetary units for policy-oriented decision support, which is based on the theory of neoclassical welfare economics, has been growing. In the U.S., cost benefit analysis (CBA) is mandatory for evaluation of various environmental policy measures. In Europe, use of CBA to justify new equipment regulation has also been increasing. The consideration of health and environmental impacts within a CBA requires quantification of health and environmental impacts as far as possible on the endpoint level to facilitate a subsequent valuation.

If a company or public administration must choose between one technological solution and another, money is a very important parameter. The cost benefit analysis has been developed to support long-term decisions from a societal point of view, in contrast to a company perspective. In particular, the field of application includes the evaluation of regulatory measures with a huge influence on the environment and the selection of general public environmental strategies. The CBA intends to convert the cost and benefits of regulatory measures, public environmental strategies, etc. to monetary units (Nas, 1996). The basic principle behind this purpose 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 the effects of the analyzed plans on society economically. Because these effects can be environmental damages, they refer to effects on the environment. Methods for their monetization allow estimating external environmental costs or externalities. They are called external because they are not considered in conventional accounting methods (Dasgupta and Pearce, 1972).

The CBA facilitates efficient management of resources for the whole society. When the results indicate that, as a consequence of the project, negative effects to third parts such as atmospheric pollution or generation of dangerous wastes dominate, the public administration intervene. Some of the interventions the government can undertake to neutralize the negative effects are to establish emission thresholds or taxes related to activities that provoke the damages. The CBA methodology consists of four phases (Nas, 1996):

- Identification of the relevant costs and benefits
- Assignment 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 appropriate, adoption of necessary interventions by the public administration

Figure 3.9 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 and environmental. The costs with a lot of visibility are the direct ones of the producer 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; these are the costs related to ownership after buying the product.

The indirect costs of the producer do not have much visibility; they consist of pollution abatement costs, actions to reduce the accident risk at the working place

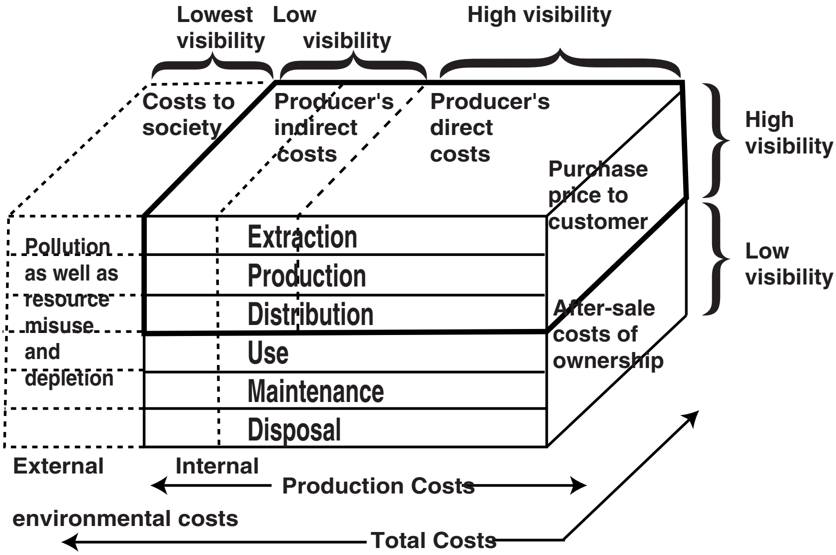


FIGURE 3.9 Types of production and environmental costs and their visibility.

and other measures not directly necessary to manufacturing the product. The first part of the environmental costs, the internal environmental costs or abatements costs, belongs to the producer's indirect costs in the first half of the life-cycle. 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 — and have very little 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 to, at least, under the legal threshold, e.g., the installation and maintenance of gas filters.
- External environmental costs or externalities are emissions and other environmental loads caused to society, e.g., increase of asthma cases; to obtain them the monetization of environmental damage estimates is necessary.

The conversion of environmental damages in external costs is called monetization. With the external environmental costs or externalities at one's disposal, it is possible to internalize these costs and calculate the total cost of a product. Theoretically, this is the price a product needs to be consistent with the market. In a figurative sense, it could be considered the amount that must be paid to maintain the planet in equilibrium, apart from the amount paid to the producer. More practically, it means that, with the monetization, environmental damages can be introduced into the equations of economic balances and that monetization gives support to solving the allocation problems of public funds for the protection of life.

However, lacking a common reference for comparison of different impact endpoints inevitably involves a value judgment. Monetization is just one option; therefore, the following critical points must be mentioned against the monetization 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; they are considered insufficient for the prescription of environmental policies.

It is out of the scope of this book to take the part of a particular point of view. The decision-maker must make the choice. Monetization methods will be briefly presented according to the state of the art; other weighting schemes exist and some of them are presented in this chapter.

In principle, two fundamental concepts exist in the science of environmental economics for the monetization 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 individuals consider 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.

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 1/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 a reference group. The focus of scientists evaluating the statistical life can be distinguished between the WTP approach and the human capital concept in which a salary not received essentially assesses the statistical life. 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, the YOLL principle has been established. It is possible to estimate YOLLs based on VSL if data on the age of the reference group affected by environmental damages are available.

Another important point for the monetization of environmental damages is the discount rate. Discounting is the practice of giving a lower numerical value to benefits in the future than to those in the present. This fact has many consequences when applied to the monetization 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 3.4, which means, for example, that at a 10% discount rate (r), 100 U.S.$

today is comparable to 121 U.S.$ in 2 years' time. Different stakeholders have broadly discussed the question of the most adequate discount rate.

$$Xt = X_0 \cdot (1 + r)^t \tag{3.4}$$

The accurate economic evaluation of environmental damages depends not only on the monetization method chosen and the discount rate used, but also on the question of how far economic damage values can be transferred from one place to another once they have been determined. For example, it is difficult to decide which modifications should be done in order to use results of U.S. studies in the EU.

In the case study in this book, monetization 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, which facilitates the valuation of an important part of the impacts. It allows obtaining an under-borderline of the total environmental cost, although other types of costs exist.
- The WTP method tries to answer the question of how much one is prepared to pay to reduce emissions. This method is considered an adequate measure of preference. For example, with certain decisions, such as buying a car with or without an airbag, individuals give a price to their lives.
- Discounting corresponds to weighting on the level of intergenerational equity, which means that the interests of future generations must be taken into account. However, because 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. 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 valuation of a VSL on the basis of:

$$VSL = \Sigma\ WTP/\Delta p \tag{3.5}$$

where VSL = value of statistical life, WTP = willingness to pay, and 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., 1997).

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; thus, 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). Because little empirical evidence on the WTP for LYLs exists, the ExternE study has developed a theoretical framework to calculate the VLYL from the VSL. Assuming for simplicity 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 based on this assumption, a VLYL corresponds to approximately 0.1 million U.S.$. In principle, a discount rate of 3% is applied throughout the case study of this book. Based on the uncertainty analysis in Chapter 5, this book will try to compare the uncertainties due to this valuation step with other sources of uncertainties in environmental impact analysis.

3.7.5 Eco-Indicator 99 as Approach using Cultural Theory and DALY

Eco-indicator 95 was based on the distance-to-target approach; however, this method has been criticized because it offers no clear-cut objective way to define sustainable target levels. Thus, the subjectivity of the weighting factors used contributed to the development of a new damage-oriented approach: Eco-Indicator 99 (Goedkoop and Spriensma, 1999).

To calculate the eco-indicator score, three steps are necessary:

1. Inventory of all relevant emissions, resources extraction and land-use in all processes that form the life-cycle of a product, which is the standard procedure in life-cycle assessment as described in Chapter 2
2. Calculation of the damages these flows cause to human health, ecosystem quality and resources
3. Weighting of these three damage categories

To simplify the weighting procedure, damage categories were identified, and as a result new damage models were developed that link inventory results into three damage categories: damage to 1) human health, 2) ecosystem quality, and 3) resources. A brief description of these three damages follows. Figure 3.10 gives an overview of the eco-indicator 99 method.

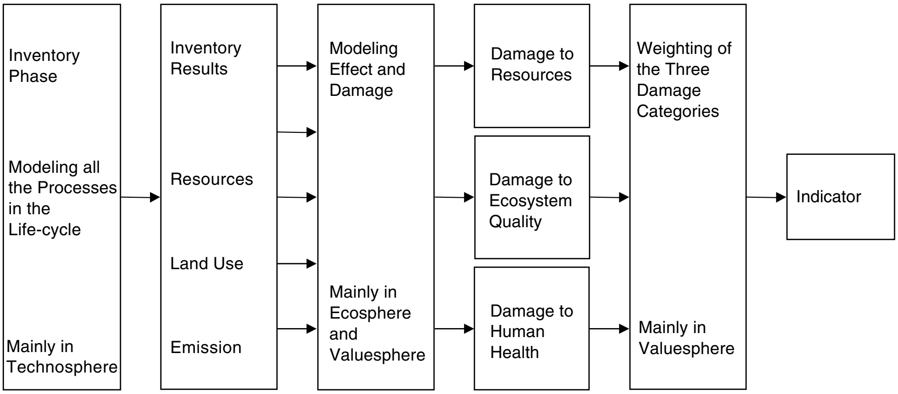


FIGURE 3.10 Overview of the eco-indicator 99 method. The term "sphere" is used to indicate that the method integrates different fields of science and technology. (From Goedkoop, M. and Spriensma, R., The eco-indicator-99. A damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands, 1999. With permission.)

3.7.5.1 Damage to Human Health

Damage models were developed for respiratory and carcinogenic effects, the effects of climatic change, ozone layer depletion and ionizing radiation. In these models for human health four steps are used:

- Fate analysis links an emission to a temporary change in concentration.
- Exposure analysis links concentration changes to a dose.
- Effect analysis links the dose to a number of health effects, such as occurrences and types of cancers or respiratory effects.
- Damage analysis links health effects to DALYs for humans, using estimates of the number of YLD and YOLL.

3.7.5.2 Damage to Ecosystem Quality

The entire damage category consists of ecotoxicity and acidification/eutrophication.

Ecotoxicity is expressed as the percentage of all species present in the environment living under toxic stress. The potentially affected fraction (PAF) is used (Van de Meent and Klepper, 1997) as an indicator and corresponds to the fraction of a species exposed to a concentration equal to or higher than the no-observed-effect concentration (NOEC). It is a measure for toxic stress and, in fact, is not a real damage.

Acidification and **eutrophication** are treated as one category. To evaluate the damage to target species in natural areas, the probability of occurrence (POO; Wiertz et al., 1992) is used. The eco-indicator 99 translates this concept to potentially disappeared fraction (PDF) = 1 – POO. Local damage on occupied or transformed areas and regional damage on ecosystems are taken into account. For land use, the PDF is used as indicator and all species are considered target species. Damages to ecosystem quality are expressed as percentage of species disappeared in a certain area due to environmental load (PDF). The PDF 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). Finally, the damages in potentially affected fraction (PAF) expressed in m^2yr of the different compartments can be added up, resulting in the total damage (Hamers et al. 1996). Table 3.9 shows an example of a calculation procedure given for an emission to air and the resulting damage in natural soil. The damages in $PAFm^2yr$ of the different compartments can be added up, resulting in the total damage in Europe.

3.7.4.3 Damage to Resources

With respect to damage category resources, the eco-indicator methodology only models mineral resources and fossil fuels. Chapman and Roberts (1983) developed an assessment procedure for the seriousness of resource depletion based on the energy needed to extract a mineral in relation to the concentration. Until now, no accepted unit to express damages to resources has been found.

For minerals, geostatistical models are used to analyze 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. In this case the "decrease" of a concentration as a result of an extraction is modeled.

TABLE 3.9
PDF Calculation for Emissions to Air and Resulting Damage in Natural Soil for 1 kg Pollutant Emissions in Europe

Calculation step	Calculation procedure	Result
Emission to air in Europe	10,000 kg/d standard flow	1.0×10^{-6} kg/m²/yr
Concentration increase (ΔC) in natural soil	EUSES	6.96×10^{-7} mg/l
No effect concentration (NOEC terrestrial)	Geometric mean NOECs	1.04 mg/l
Hazard unit (HU) increase	$\Delta HU = \Delta C/NOEC$	6.69×10^{-7}
PAF/HU at Combi-PAF = 24% (European average)	Slope factor = 0.593.(PAF/ΔH)	
PAF increase in natural soil for 10,000 kg/d in Europe	$\Delta PAF = \Delta HU \cdot 0.593$	4.13×10^{-7}
PAF increase in natural soil for 1 kg/yr in Europe	$\Delta PAF/(10{,}000 \cdot 365)$	1.130×10^{-13}
$PAFm^2\,yr$ in natural soil (2.16×10^6 km²)	(1.13×10^{-13})·surface area natural soil	0.244 $PAFm^2$.yr

Source: Goedkoop, M. and Spriensma, R., The eco-indicator 99. A damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands, 1999. With permission.

For fossil fuels, surplus energy is based on future use of nonconventional resources, especially oil shale and tar sands. In this case, the model for the surplus energy is constructed by means of descriptions of the typical characteristics of the fossil resources and with data on the increased extraction energy for nonconventional resources.

3.7.5.4 Weighting in the Eco-Indicator 99 Method

With respect to weighting step, different schemes for the evaluation of environmental damages have been developed. The most fundamental problem in damage estimations is 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. For example, in the case of the YLD and YOLL (previously seen), because value choices are necessary for weighting, they strongly depend on the attitude of the person carrying out the weighting step. Moreover, a YOLL at the age of 20 and a YOLL at the age of 60 are not equally appreciated in every socioeconomic perspective. Therefore, the DALY concept is linked to the Cultural Theory earlier described. Moreover, a panel approach is used as another weighting scheme.

3.7.5.5 The Panel Approach and Graphical Representation

Because weighting should represent the views of society or groups of stakeholders, the panel approach and the revealed preference were used by eco-indicator 99. The procedure was developed by Mettier (1998), based on previous experiences with panel and Delphi methods in LCA (Udo de Haes, 1996), and consists of a five-part questionnaire to be answered by the panel:

1. An introduction containing a brief description of the purpose, the outline and intended application of the eco-indicator 99 methodology, and a description of the damage categories (human health, ecosystems health and resources)
2. Ranking of the three damage categories (in order of decreasing importance)
3. Assigning weights
4. Linkages to cultural perspectives
5. Background questions (age, sex, etc.)

The results given by the panel can be represented on a triangle graphic (Figure 3.11).

3.7.5 Uniform World Model

Based on the Impact Pathway Analysis (IPA) studies on a European level in the ExternE Project (see in Chapter 4 for more details), Rabl et al. (1998) compared the results of detailed site-specific calculations for more than 50 electric power stations

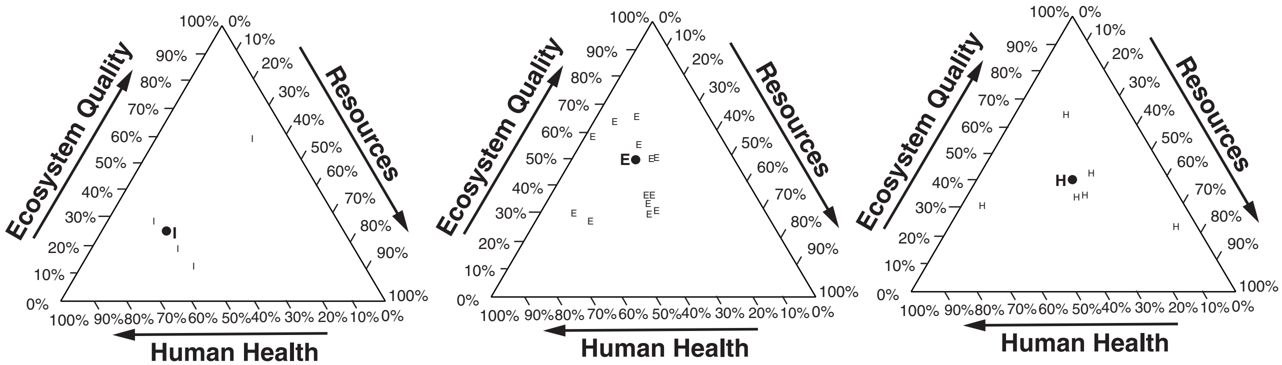


FIGURE 3.11 Triangle graphic for weighting per the three cultural perspectives: individualist = I; egalitarian = E; hierarchist = H. (From Goedkoop, M. and Spriensma, R. (1999). The Eco-Indicator 99. A damage-oriented method of life-cycle impact analysis, Pré Consultants, Amersfoort, The Netherlands. With permission.)

and MSWIs all over Europe and introduced the Uniform World Model (UWM) with the Expressions (3.5) and (3.6) for:

a) primary pollutants

$$D = D_{uni} = \frac{f_{CR} \cdot \rho_{uni}}{k_{uni}} \cdot Q \tag{3.5}$$

where: $D = D_{uni}$ =(uniform) damage [cases/a]
f_{CR} = slope of concentration-response function (dose-response or exposure-response) [cases/persons*a*μg/m^3]
ρ_{uni} = uniform receptor density [1.05E-04persons/m^2]
κ_{uni} = uniform removal velocity [m/s]
Q = emission [μg/s]

b) secondary pollutants

$$D_{2uni} = D_{uni} = \frac{f_{CR2} \cdot \rho_{uni}}{k_{2uni,eff}} \cdot Q_1 \tag{3.6}$$

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/m^3]
$k_{2uni,eff}$ = effective uniform removal velocity for secondary pollutant[m/s]
Q_1 = emission of primary pollutant [μg/s]

The slope of functions states the incremental number of cases (e.g. hospitalisations per concentration increment). Table 3.10 shows typical removal velocity values as obtained in different IPA studies.

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 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 for a first estimate within an order of magnitude of damage estimates expressed as external costs, monetized according to the guidelines of the European Commission (1995). Table 3.10 presents the results computer with the UWM. The multipliers indicate how much the costs can change with site (rural and urban) and stack conditions (height, temperature and exhaust velocity).

TABLE 3.10
Typical Removal Velocity Values for Different Pollutants

Primary pollutants	**$K_{2uni,eff}$ (m/s)**
$NO_2 \rightarrow$ nitrates	0.008
$SO_2 \rightarrow$ sulphates	0.019
Secondary pollutants	**k_{uni} (m/s)**
PM10	0.01
SO_2	0.01
CO	0.001
Heavy metals	0.01
PCDD/Fs	0.01

Source: Data taken from Rabl, A. et al., *Waste Manage. Res*., 16(4), 368–388. 1998.

3.8 SOPHISTICATION IN LIFE-CYCLE IMPACT ASSESSMENT

Sophistication in LCIA has been an important topic for scientific discussion (Bare et al., 1999; UNEP DTIE, 2003). 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, depending 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).

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

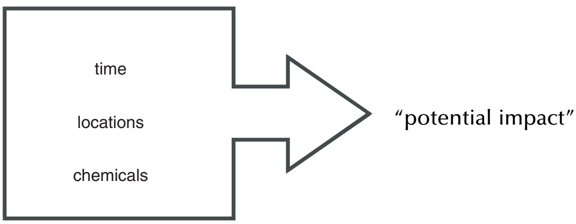


TABLE 3.11
European Health Damage Costs Calculated with the Uniform World Model

Pollutant	Cost (mU.S.$/kg emitted pollutant)	Multiplier for site (rural ↔ urban)	Multiplier for stack emissions (height 250 ↔ 0 m, T, v[a])
CO	2.07	?	?
NO_x via Nit	1.69×10^4	≈ 0.7 ↔ 1.5	≈ 1.0
SO_2 tot	1.22×10^4	≈ 0.7 ↔ 1.5	≈ 1.0
PM 10	1.36×10^4	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
As	1.50×10^5	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Cd	1.83×10^5	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Cr	1.23×10^5	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
Ni	2.53×10^3	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0
PCDD/Fs	1.63×10^{10}	≈ 0.3 ↔ 3	≈ 0.6 ↔ 2.0

m = 10^{-3}
[a] t= temperature; v = velocity
Source: Data taken from Rabl et al. (1998).

All this 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 characterization modeling to include not only the effects of the substances but also their fates. Another issue concerns a possible differentiation in space and time. Studies can include impact models that use data 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 modeling. More sophisticated possibilities arise that take background levels of substances into account and make use of nonlinear dose–response functions. An important question here is whether these are real science-based thresholds, or whether these thresholds are always of political origin. A further question relates to the role and practicality of including uncertainty analysis. Although sensitivity analysis is increasingly included in LCA studies, this is not yet the case for uncertainty analysis. Finally, there are the questions 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 (Bare et al. 1999; UNEP DITE, 2003).

The important issue of deciding the appropriate level of sophistication is typically not addressed in LCA. Often, 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 (Bare et al. 1999; UNEP DTIE, 2003):

- Study objective
- Inventory data and availability of accompanying parameters
- Depth of knowledge and comprehension in each impact category
- Quality and availability of modeling data
- Uncertainty and/or sensitivity analysis
- Level of financial resources

3.9 INTERPRETATION

To conclude the LCIA step, the practitioner must carry out analysis and interpretation of its results in order to evaluate the environmental performance of the product or activity under investigation. The actual assessment of the environmental profile of the product takes place during the evaluation. The nature of the assessment is determined by the goal step of the study. Usually, this will be a comparative assessment. Other examples include providing information about the environmental performance of the product regarding some function, product regulation by government agencies, benchmarking and comparing a product with one or more possible alternatives of its redesign.

The interpretation is an independent step when the goal of the LCA is to find options to improve a product. During the improvement analysis, environmental LCA-based product information is used to make recommendations about the optimization of its manufacturing (including actions of processess or product design) or changes concerning its use by the consumer, e.g., washing at low temperature.

In any case, some priorities need to be established in order to guide the work of the practitioner. In this frame, questions like "What is more important at this moment?" or "What comes first: dealing with the greenhouse effect or with photo-oxidation formation?" or "In terms of LCI, should the first action be to reduce the CO_2 emissions or the COD (chemical oxygen demand) generation?" define the type of evaluation to be carried out during the interpretation step.

The LCIA generates an environmental profile of the product consisting of a certain number of impact potentials that help to compare product alternatives. It depends on the specific case if it will then be possible to draw a conclusion without further weighting. In principle, this is only possible when all of the impact potentials of a product alternative are better than those of the other product (Heijungs et al., 1992).

However, in many cases, one product alternative will present a better environmental performance for some impact potentials but worse on others. In cases like this, the impact potentials will have to be rated in order to make an assessment. Usually, two methods can be used for this: qualitative multicriteria analysis and quantitative multicriteria analysis. As presented by Heijungs et al (1992), both methods include methodological as well as procedural aspects. The procedural aspects are largely concerned with issues such as who will undertake the evaluation and what information is provided to those concerned.

In the qualitative method a panel rates the better and poorer impact potentials (see eco-indicator 99 example in Section 3.7.5). The advantage of this method is that all involved parties can express their points of view, furnishing a multidisciplinary perspective to data interpretation. A clear disadvantage is the loss of uniformity inherent to the method: when two different persons assess a set of two environmental profiles, their results can be highly different.

A quantitative multicriteria assessment is based on weighting factors established by the explicit weighting of the impact potentials.

In the event, an important point about LCA application must be remarked no matter which interpretation method is selected. This methodology is a powerful instrument of support regarding the evaluation of environmental and human health impacts; however, in many cases its results can be useful without an appreciation of the reliability and validity of the information. In this framework, a quantitative sensitivity analysis must be performed to assess the effect of the key assumptions on the final results, to check the data whose quality is suspected or unknown, to show if the study results are highly dependent on particular sets of inputs, and to evaluate life-cycle effects of changes being considered (Consoli et al., 1993).

Conclusions should only be drawn on study results with consideration of the data variability and resulting variability of the findings. Chapter 5 will discuss this subject more thoroughly and present alternative methods to carry out a qualitative sensitivity analysis.

3.10 EXAMPLE: COMPARISON OF PET AND GLASS FOR MINERAL WATER BOTTLES

This example addresses the question of what is better from an environmental point of view: consuming mineral water in nonreturnable small plastic bottles made of PET or in returnable glass bottles. For the sake of simplicity, as parameter for the comparison, only the greenhouse effect (GWP) is to be considered. The basis of the calculation is 1 L of mineral water consumed in small bottles. Calculations must be made taking into account the life-cycle of the two types of bottles and the environmental load of water and bottle transportation. The following assumptions are considered:

- In the case of the plastic bottles, the impact related to bottle manufacturing is not taken into account. The same holds for transportation because the bottles are manufactured in the bottling plant. Empty glass bottles are delivered from the glass factory in 16-t trucks.
- The bottled water is delivered from the bottling plant to the wholesaler in 16-t trucks and from the wholesaler to the retail trader by van. The glass bottles are returned by van to the wholesaler and from the wholesaler to the bottle manufacturer in 16-t trucks.
- The impact of cleaning the bottles is not considered.

For the calculations the following data must be used:

	PET	Glass
Bottle weight (g)	20	237
Bottle capacity (L)	0.33	0.25
Number of uses	1	20
Distance from bottling plant to wholesaler (km)	50	50
Distance from wholesaler to retail trader (km)	20	20
Distance from bottle manufacturer to bottling plant (km)	—	100

Regarding GWP, the following EL must be considered:

		Production		Van <3.5 t	Transport
EL	Units	PET (1 kg)	Glass (1 kg)	(1 tkm)	16-t Truck (tkm)
CO_2	kg	3.45	9.68×10^{-1}	1.54	3.46×10^{-1}
CH_4	kg	1.17×10^{-2}	2.32×10^{-3}	2.61×10^{-3}	5.34×10^{-4}

tkm equivalent to a mass of 1 t (1000 kg) transported 1 km.

For the calculation of the greenhouse effect, consider the following impact factors:

Compound	Factor
CO_2	1
CH_4	62

Solution:

Weight of bottling material associated with 1 L of water consumption:
PET: 20/0.33/1000 = 0.0606 kg
Glass: 237/0.25/20/1000 = 0.0474 kg
Transport TP-PET (tkm) assigned to 1 L of water consumption in PET bottles
1/transport in **16-t truck**
Only one trip from bottling plant to the wholesaler
$\mathbf{(TP\text{-}PET)_{16t}} = ((1000 + 60.61)/10^6).50 = \mathbf{0.053\ tkm}$
2/transport in **Van <3.5 t**
One trip from wholesaler to the retail trader
$\mathbf{(TP\text{-}PET)_{van}} = ((1000 + 60.61)/10^6).20 = \mathbf{0.0212\ tkm}$
Transport TP-glass (tkm) assigned to 1 L of water consumption in glass bottles
1/transport in **16-t truck**
1a/one trip of empty bottles from glass manufacturer to bottling plant
$(TP\text{-}glass)_{16t,a} = (47.4/10^6).100 = 0.00474$ tkm
1b/one trip of full bottles from bottling plant to the wholesaler
$(TP\text{-}glass)_{16t,b} = ((1000+(237/0.25))/10^6).50 = 0.0974$ tkm
1c/one return trip of empty bottles from wholesaler to the bottling plant
$(TP\text{-}glass)_{16t,c} = ((237/0.25)/10^6).50 = 0.0474$ tkm
Total $\mathbf{(TP\text{-}glass)_{16t}} = 0.00474 + 0.0974 + 0.0474 = \mathbf{0.150\ tkm}$
2/transport in **Van < 3.5 t**
2a/one trip from wholesaler to the retail trader
$(TP\text{-}glass)_{Van,a} = ((1000+(237/0.25))/10^6)*20 = 0.0390$ tkm
2b/one return trip of empty bottles from retail trader to the wholesaler
$(TP\text{-}glass)_{Van,b} = ((237/0.25)/10^6)*20 = 0.0190$ tkm
Total $\mathbf{(TP\text{-}glass)_{van}} = 0.0390 + 0.0190 = \mathbf{0.058\ tkm}$
The respective amounts (kg) of CO_2 and CH_4 assigned to the consumption of 1 L of mineral water bottled in PET or glass are calculated by multiplying

the corresponding mass of bottling material (kg) and transport intensity (tkm) by the EL per unit of material and transport previously given as data:

EL	PET	PET bottle Transport	Total	Glass	Glass bottle Transport	Total
CO_2 (kg)	2.09×10^{-1}	5.11×10^{-2}	**2.60×10^{-1}**	4.59×10^{-2}	1.41×10^{-1}	**1.87×10^{-1}**
CH_4 (kg)	7.09×10^{-4}	8.37×10^{-5}	**7.93×10^{-4}**	1.10×10^{-4}	2.31×10^{-4}	**3.41×10^{-4}**

The contribution in percentage is presented as follows:

EL	PET	PET bottle Transport	Total	Glass	Glass bottle Transport	Total
CO_2 (kg)	80.4	19.6	100.0	47.6	52.4	100.0
CH_4 (kg)	89.4	10.6	100.0	57.1	42.9	100.0

The amount of equivalent CO_2 is calculated by multiplying the mass of methane (CH_4) by the corresponding factor and adding it to the mass of CO_2.

CO_2 Equivalent	PET	Glass
CO_2	0.260	0.187
CH_4	0.049	0.021
CO_2 TOTAL kg	**0.309**	**0.208**

Conclusion: Regarding the greenhouse effect, it is preferable to consume mineral water in glass bottles, in consideration of the assumptions made and the data provided.

3.11 CASE STUDY: APPLICATION OF LCIA METHODS IN THE MSWI PROCESS CHAIN LCA

The results of the application of the Eco-Indicator 95 method to the MSWI process chain LCA (see Chapter 2) are shown in Table 3.12. Scenario 2, the current operation of the incineration plant after the installation of an advanced gas treatment system,

TABLE 3.12
Differences in Impact Categories According to Eco-Indicator 95

	GWP	ODP	POCP	NP	AP	Pb equiv.	PAH equiv.	SO_2 equiv.	Eco-Ind. 95
Difference[a] (%)	–10.3	–18.5	–19.5	–7.5	65.4	64.9	63.0	38.6	59.9

[a](Scenario 1 – Scenario 2)/Scenario 1

is associated with a higher global warming potential (GWP) and nutrification potential (NP) than Scenario 1, i.e., the former operation, because it has a higher CO_2 and NO_x emission per produced TJ due to additional energy consumption for the advanced gas treatment system. Scenario 2 also has a higher ozone depletion potential (ODP) and photochemical ozone creation potential (POCP) caused by the higher contribution of the transport.

Scenario 2 is more favorable than Situation 1 in cases of acidification potential (AP) and winter smog (SO_2 equivalent) due to the reduction of HCl, SO_2 and dust. This scenario is also favorable for heavy metals (Pb equivalent) and carcinogenic substances (PAH equivalent) because they are 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 impacts reduced by the advanced gas treatment system (mainly acidification and heavy metals). Therefore, it can be concluded that the installation of an advanced gas treatment decreases stack emissions and the related inventory and impact assessment data, but increases the majority of the other environmental loads considered because of 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.

Furthermore, the results obtained with Eco-Indicator 95 methods are comparable with results obtained from other LCIA methods: MIPS (Schmidt-Bleek, 1994), EPS (Steen and Ryding, 1992) and the method of the Tellus Institute (1992). Figure 3.12 shows overall results for the difference between Scenario 1 without and Scenario 2 with an advanced gas system.

It can be observed that selected methods corresponding to different weighting approaches do not deliver results with the same tendency. Two methods show an

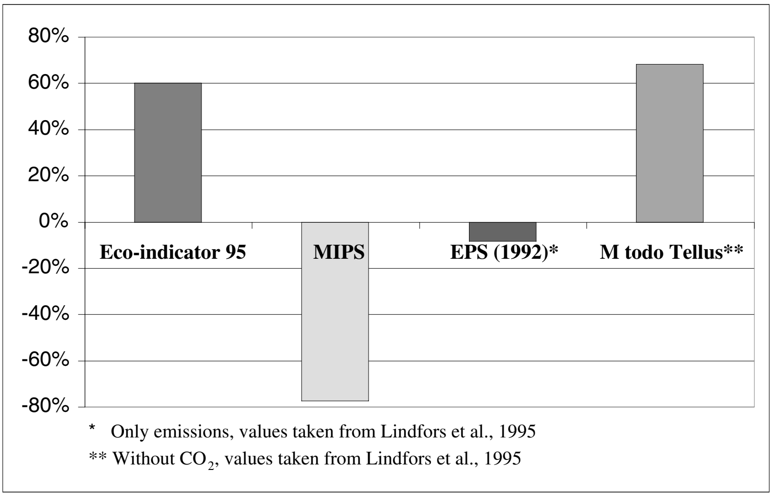


FIGURE 3.12 Comparison of different single-index LCIA methods.

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 only on a value to measure the environmental performance. However, above all, this kind of comparison is difficult because the number and types of environmental loads considered in each method vary significantly.

According to EU environmental policy, the installation of advance gas treatment systems is obligatory in order 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 results obtained, more than 96% of the total is caused by CO_2. In the same way as explained for the Eco-Indicator 95 results, the contribution of the heavy metals decreases, while the values for NO_x and CO_2 increase because they are not eliminated and the overall energy efficiency declines.

3.12 QUESTIONS AND EXERCISES

1. Build a cause-effect diagram associating inventory results with impact categories, category indicators, and category endpoints for each case, starting from a life-cycle inventory for: CO_2, SO_2, CH_4, N_2O, NO_2, and Pb.
2. Associate possible midpoint and/or endpoint indicators with the following impact categories: stratospheric ozone depletion; acidification; eco-toxicity; and human toxicity.
3. Explain the main differences between midpoint and endpoint indicators in LCIA. Try to explain these differences by using some examples.
4. Summarize the main advantages and disadvantages of using endpoint approaches in LCA.
5. Estimate the GWP of the air emissions, including direct impacts to the environment (from 1,1,1-TCA) and indirect impacts from fossil fuel-based energy use (CO_2, N_2O), and the percentage distribution for each chemical. Use the following data for air emission (20,000 kg 1,1,1-TCA/h).

Chemical	m_i (kg/h)	GWP
TCA	13	100
CO_2	10	1
N_2O	0.18	310

Discuss the result comparing the effect resulting from the direct impacts and those from indirect impacts due to the fossil fuel–based energy used. Considering the possibility of a renewable energy use (biomass-based fuels), what could be concluded regarding the effects to the global warming of the actual process?

6. Describe the scheme of the eco-indicator 95 structure of LCA for eutrophication and carcinogenics.
7. The application of eco-indicator 95 has been sometimes criticized because of its failure to be an objective assessment tool at the weighting phase

level. Explain briefly the main faults of eco-indicator 95 and the advantages introduced by eco-indicator 99.

8. Summarize the weighting methods proposed for use in LCIA and briefly describe their advantages and disadvantages.
9. How many times should the glass bottle be recycled to generate the same amount of equivalent CO_2 as the plastic bottle? Answer: 7 times.
10. Compare plastic sheets to paper from cellulose pulp.

According to NASA, the ozone's layer hole above Antarctica has been increasing. If, on September 19, 1998, it was measured 27.2×10^6 km^2 and, 2 years later, on September 3, this surface had increased to 28.3×10^6 km^2, determine what would have a worse effect on the ozone's layer: using a 5-g paper pulp sheet or a 5-g low density polyethylene (LDPE) sheet. The comparison must be based on the measurement of the ozone depletion potential (ODP) expressed as kilograms of CFC-11 equivalent. The needed environmental loads (ELs), in this case emissions of chlorfluorinated compounds (CFCs) to the air for paper and polyethylene production and the corresponding impact factors, are given in the following table:

EL	Unit	Paper (EL/kg)	LDPE (EL/kg)
Halon 1301	kg	7.15×10^{-8}	6.85×10^{-7}
CFC-11	kg	2.37×10^{-9}	4.06×10^{-9}
CFC-114	kg	6.24×10^{-8}	1.07×10^{-7}
CFC-12	kg	5.08×10^{-10}	8.73×10^{-10}
CFC-13	kg	3.19×10^{-10}	5.48×10^{-10}
HCFC-22	kg	5.66×10^{-10}	9.60×10^{-10}

Impact factors:

Compound	Unit	Factor
Halon-1301	kg	1.6×10^1
CFC-11	kg	1
CFC-114	kg	8.0×10^{-1}
CFC-12	kg	1
CFC-13	kg	1
HCFC-22	kg	5.5×10^{-2}

11. Compare the use of two different plastics: PP and PVC. A company endeavors to compare the environmental impact due to the industrial use of two polymers: polypropylene (PP) and polyvinyl chloride (PVC) as raw material for manufacturing piping. It considers it necessary to take into account two variables: the raw material and its transport. Two alternatives are supposed:
    1. Buying PP piping from a distributor who receives the product by train from a factory 1000 km away and uses a van to deliver the product to a consumer who is 200 km from the distributor
    2. Buying PVC tubes from a distributor who acquires the product by a 16-t truck from a factory 700 km away and uses a van to deliver the product to a consumer who is 50 km from the distributor

To compare the environmental behavior of each product, the company makes use of three parameters: kilograms of consumed oil, GWP measured as kilograms of equivalent CO_2, and acidification (acid potential, AP) measured as kilograms of equivalent SO_2.

Determine for each alternative the value of these parameters related to environmental impact caused by the use of 1 kg of polymer and determine which of these kinds of plastics would be better used from an environmental point of view. The necessary ELs and impact factor data are given in the following table.

EL	Unit	PP (1 kg)	PVC (1 kg)	Van (tkm)	16-t Truck (tkm)	Train (tkm)
Oil	kg	1.82	1.07	4.71×10^{-1}	$1{,}06 \times 10^{-1}$	8.73×10^{-3}
CO_2	kg	3.11	3.22	1.54	3.46×10^{-1}	5.41×10^{-2}
SO_2	kg	2.22×10^{-2}	1.92×10^{-2}	3.39×10^{-3}	7.3×10^{-4}	2.15×10^{-4}
NO_2	kg	4.44×10^{-2}	6.70×10^{-3}	8.72×10^{-2}	3.52×10^{-3}	2.89×10^{-4}
CH_4	kg	1.07×10^{-2}	8.05×10^{-3}	2.61×10^{-3}	5.34×0^{-4}	1.06×10^{-4}
HCl	kg	1.65×10^{-4}	3.26×10^{-4}	3.48×10^{-5}	6.05×0^{-6}	5.43×10^{-3}

tkm equivalent to a mass of 1 t (1000 kg) transported 1 km.

Environmental impact factors:

GWP		AP	
Component	Factor	Component	Factor
CO_2	1	SO_x	1
CH_4	62	NO_x	0.7
		HCl	0.88

REFERENCES

Bare, J.C., Hofstetter, P., Pennington, D.W., and Udo de Haes, H.A. (2000), Midpoints versus endpoints — the sacrifices and benefits, *Int. J. LCA*, 5, 319–326.

Bare, J.C., Pennington, D.W., and Udo de Haes, H.A. (1999), Life-cycle impact assessment sophistication — international workshop, *Int. J. LCA*, 4(5), 299–306.

Braunschweig, A., Foerster, R., Hofstetter, P., and Müller-Wenk, R. (1994), Evaluation und Weiterentwicklung von Bewertungsmethoden fuer Oekobilanzen — erste Ergebnisse. IÖW-Diskussionsbeitraege Nr. 19, IÖW-HSG, St. Gallen, Switzerland.

Chapman, P.F. and Roberts, F. (1983), Metal resources and energy, *Butterworth's Monographs in Materials,* ISBN 0408108029.

Consoli, F. et al. (1993), Guidelines for life-cycle assessment: a code of practice, Society of Environmental Toxicology and Chemistry, SETAC.

Dasgupta, A.K. and Pearce, D.W. (1972), *Cost-Benefit Analysis — Theory and Practice,* Macmillan Press, Cambridge.

EC (European Commission) (1995), *ExternE – Externalities of Energy*, 6 vols., EUR 16520-162525. DG XII Science, Research and Development, Brussels.

Fava, J., Consoli, F., Denison, R., Dickson, K., Mohin, T., and Vigon, B., (1993), *A Conceptual Framework for Life-Cycle Impact Assessment*, SETAC Press, Pensacola, FL.

Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G., Hischier, R., Martin A. (ETH Zürich), Dones, R., and Gantner, U. (PSI Villigen) (1996), *Ökoinventare von Energiesystemen – Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz*. 3rd ed., ETH Zürich: Gruppe Energie-Stoffe-Umwelt, PSI Villigen: Sektion Ganzheitliche Systemanalysen, 1996.

Goedkoop, M. and Spriensma, R. (1999), The eco-indicator 99. A damage-oriented method for life-cycle impact assessment, Pré Consultants, Amersfoort, The Netherlands.

Goedkoop, M.J. (1995), Eco-indicator 95 — final report, NOH report 9523, Pré Consultants, Amersfoort, The Netherlands.

Guinée, J., Heijungs, R., van Oers, L., Van de Meent, D., Vermeire, T., and Rikken, M. (1996), LCA impact assessment of toxic releases — generic modeling of fate, exposure, and effect for ecosystems and human beings with data for about 100 chemicals, RIVM report number 1996/21. Bilthoven, The Netherlands.

Hamers, T., Aldenbergc T., and van de Meent, D., (1996), Definition report indicator effects toxic substances (I_{tox}), RIVM report number: 60712001.

Hauschild, M. and Wenzel, H. (1998), *Environmental Assessment of Products — Scientific Background*, Vol. 2, Chapman & Hall, London.

Heijungs, R., Guinée, J.B., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., and Wegener-Sleeswijk, A. (1992), Environmental life-cycle assessment of products — guide and backgrounds, technical report, CML, University of Leiden, The Netherlands.

Hertwich, E.G., Mateles, S.F., Pease, W.S., and McKone, T.E. (2001), Human toxicity potentials for life-cycle assessment and toxic release inventory risk screening, *Environ. Toxicol. Chem.*, 20, 928–939.

Hofstetter, P. (1998), *Perspectives in Life-Cycle Impact Assessment — a Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere*, Kluwer Academic Publishers, London.

Hogan, L.M., Beal, R.T., and Hunt, R.G. (1996), The threshold inventory interpretation methodology, *Int. J. LCA*, 1, 159–167.

Houghton, J.T., Meira Filho, L.G., Bruce, J., Lee, H., Callander, B.A., Haites, E., Harris, N., and Maskell, K. (1995), *Climate Change 1994 — Radioactive Forcing of Climate Change and an Evaluation of the IPCC IS92 Emissions Scenarios*, Cambridge University Press, London.

ISO 14042 (2000), Environmental management — life-cycle assessment — life-cycle impact assessment, technical standard, International Standards Organization, Geneva, Switzerland.

ISO/TR 14047 (2002), Illustrative examples on how to apply ISO 14042 — life-cycle assessment — life-cycle impact assessment, committee draft 1, International Standards Organization, Geneva, Switzerland.

Itsubo, N. and Inaba, A. (2000), Definition of safeguard subjects for damage oriented methodology in Japan, presented at the 4th Eco-Balance Conference, Tsukuba, Japan.

Krewitt, W., Holland, M., Truckenmüller, A., Heck, T., and Friedrich, R. (1999), Comparing costs and environmental benefits of strategies to combat acidification and ozone in Europe, *Environ. Econ. Policy Stud.,* 2, 249–266.

Lindfors, L.-G., Christiansen, K., Virtanen, Y., Juntilla, V., Janssen, O.J., Rønning, A., Ekvall, T., and Finnveden, G. (1995), Nordic guidelines on life-cycle assessment, *Nord* 1995: 20, Nordic Council of Ministers, Copenhagen.

Mayerhofer, P., Krewitt, W., and Friedrich, R. (1997), Extension of the Accounting Framework, Final report. ExternE Core Project. IER, Universität Stuttgart, Germany.

Mettier, T. (1998), Der Vergleich von Schutzguetern — Ausgewaehlte Resultate einer Panel-Befragung, in Hofstetter, P., Mettier, T., and Tietje, O., Eds., *Ansaetze zum Vergleich von Umweltschaeden, Nachbearbeitung des 9.* Diskussionsforums Oekobilanzen vom 4. ETZ Zuerich.

Murray, C.J.L. and Lopez, A.D. (1996), *The Global Burden of Disease,* vol. 1 of *Global Burden of Disease and Injury Series*, WHO, Harvard School of Public Health, World Bank. Harvard University Press, Boston.

Nas, T.F. (1996), *Cost-Benefit Analysis — Theory and Application*, SAGE Publications, New York.

Olsen, S.I., Christensen, F.M., Hauschild, M., Pedersen, F., Larsen, H.F., Toerslov, J. (2001), Life-cycle impact assessment and risk assessment of chemicals — a methodological comparison, *Environ. Impact Assess. Rev.*, 21, 385–404.

Owens, J.W., Amey, E., Barnthouse, E., Consoli, F., Coulon, R., Fava, J., Humphreys, K., Haaf, B., Hunt, B., Laibson, L., Mones, E., Noesen, S., Norris, G., Todd, J.A., Vigon, B., Weitz, K., and Young, J. (1997), *Life-Cycle Impact Assessment — State of the Art,* SETAC Press, Pensacola, FL.

Potting, J. (2001), Possibilities and limitations of spatial resolved impact characterization in LCA, presented at the P&G European Technical Center OMNIITOX-Project Workshop, Brussels.

Powell, J.C. and Pidgeon, S. (1994), Valuation within LCA: a multicriteria approach, in SETAC-Europe, Eds., *Integrating Impact Assessment into LCA*, SETAC publication, Brussels.

Rabl, A., Spadaro, J.V., and McGavran, P.D. (1998), Health risks of air pollution from incinerators — a perspective, *Waste Manage. Res.*, 16(4), 368–388.

Sage, S. (1993), Industrielle Abfallvermeidung und deren Bewertung am Beispiel der Leiterplattenherstellung. Ph.D. thesis. Institute for Process Engineering, TU Graz, Austria.

Schmidt-Bleek, F.B. (1994), *Wieviel Umwelt braucht der Mensch? MIPS das Mass für ökologisches Wirtschaften*, Birkhauser Verlag, Berlin.

Schmitz, S., Oels, H.J., and Tiedemann, A. (1994), Ökobilanz für Getränkeverpackungen, UBA-Texte 52/95, Umweltbundesamt, Berlin.

Solomon, S. and Albritton, D.L. (1992), Time-dependent ozone depletion potentials for short- and long-term forecasts, *Nature*, 357, 33–37.

Steen, B. (1999), Systematic approach to environmental priority strategies in product development (EPS), version 2000 — models and data of the default method, Chalmers University of Technology, Göteborg, Sweden.

Steen, B. and Ryding, S.O. (1992), The EPS enviro-accounting method, report for Federation of Swedish Industries, Swedish Environmental Research Institute, Göteborg.

Tellus Institute (1992), The Tellus Packaging Study — inventory of material and energy use and air and water emisions from the production of packing materials, technical report, Tellus Institute, Boston.

Thompson, M., Ellis, R., and Wildavsky, A. (1990), *Cultural Theory*, Westview Print, Boulder, CO.

Udo de Haes, H., Jolliet, O., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E., Hofstetter, P., Klöpffer, W., Krewitt, W., Lindeijer, E., Mueller-Wenk, R., Olson, S., Pennington, D., Potting, J., and Steen, B. (2002), *Towards Best Available Practice in Life-Cycle Impact Assessment*, SETAC-Europe publication, Brussels.

Udo de Haes, H.A and Lindeijer, E. (2001), The conceptual structure of life-cycle impact assessment, final draft for the Second Working Group on Impact Assessment of SETAC-Europe (WIA-2), in Udo de Haes et al. (2002), *Towards Best Available Practice in Life-Cycle Impact Assessment*, SETAC-Europe publication, Brussels.

Udo de Haes, H.A. (1996), *Towards a Methodology for Life-Cycle Impact Assessment*, SETAC-Europe publication, Brussels.

Udo de Haes, H.A., Joillet, O., Finnveden, G., Hauschild, M., Krewitt, W., Mueller-Wenk, R. (1999), Best available practice regarding categories and category indicators in life-cycle impact assessment, background document for the Second Working Group on LCIA of SETAC-Europe, *Int. J. LCA*, 4, 66–74, 167–174.

UNEP DTIE (2003), Evaluation of environmental impact in life-cycle assessment, UNEP publication, Paris.

Van de Meent, D. and Klepper, O. (1997), Mapping the potential affected fraction (PAF) of species as an indicator of generic toxic stress, RIVM report number: 607504001.

Weidema, B.P., Ekvall, T., Pesonen, H.L., Rebitzer, G., and Sonnemann, G.W. (2002), *Scenario Development in LCA. Report of the SETAC-Europe LCA Working Group "Scenario Development in LCA*, SETAC Press, Pensacola, FL.

Wenzel, H., Hauschild, M., and Alting, L. (1997), *Environmental Assessment of Products.* Vol. 1, *Methodology, Tools and Case Studies in Product Development*, Chapman & Hall, London.

Wiertz, J., van Dijk, B, and Latour, J.B. (1992), MOVE. Vegetatie-module; de kans op voorkomen van 700 plantensoorten als functie van voch, pH, nutrienten en zout. RIVM report number: 711901006, Bilthoven, The Netherlands.

4 Environmental Risk Assessment

With the contribution of Montserrat Meneses

4.1 INTRODUCTION

This chapter deals with the typical concept of risk assessment as a fundamental basis of environmental management. A general introduction of the risk concept is given in the first part of the chapter and hazard identification is explained. Exposure assessment and risk characterization for human health and ecological exposure are described; also included is a description of environmental monitoring, as well as general information about the fate and exposure models. Special emphasis is given to the study of the dose–response and exposure–response functions and the human health risk characterization for carcinogenic and noncarcinogenic risk. Some examples are introduced and different practical problems related to them are given at the end of the chapter.

Another approach is also explained: the impact pathway analysis (IPA), which is shown as an alternative way of analysis in cases for which risk assessment (RA) results have to be converted into damage estimations.

IPA is used to assess the impacts produced by some of the processes of the life-cycle of a process or service, with a higher level of detail than that obtained from a conventional LCA. Because the impact caused by a pollutant emitted at a specific site depends on site-specific parameters such as population density, meteorological data, etc., the tools used by risk assessment, such as fate, transport and exposure assessment, are included in the analysis. In this manner, risk assessment and LCA can be linked.

IPA is presented in this chapter as an application of environmental risk assessment (ERA) but with a huge potential for use within the life-cycle impact assessment (LCIA). The IPA will be thus taken up again in later chapters for different applications, development of methodology and examples.

4.2 RISK ASSESSMENT

The term “risk” has different meanings depending on different contexts. For a layperson it embodies the concepts of severity and probability of outcome. For example, people do not consider death by asteroid impact very risky, primarily because the likelihood of such an occurrence is perceived to be very small. Similarly, death from an accident or a fall at home is not appreciated as a significant risk because these do not normally connote a lethal injury and their severity seems to be within an individual's control. Death and injury from attack by strangers is widely feared as a high risk because of the apparent frequency of such occurrences as reported by the news media. Risk implies not only some adverse result, but also uncertainty. Risk changes as information becomes more specific — a golfer has greater risk of death by lightning than the population as a whole, whether this is perceived as likely or not. The risk from an injury at home or being struck by lightning can be calculated because these events actually happen. In contrast, assessment of risk attributable to low levels of environmental contaminants is an uncertainty exercise.

People use the term risk in everyday language to mean "chance of disaster." When used in the process of risk assessment it has specific definitions; the most commonly accepted is "the combination of the probability, or frequency, of occurrence of a defined hazard and the magnitude of the consequence of the occurrence" (Royal Society, 1992). On the other hand, hazard can be defined as "the potential to cause harm" and also as "a property or situation that in particular circumstances could lead to harm" (Royal Society, 1992).

The risk assessment is applied in a wide range of professions and academic subjects. Engineers "risk assess" bridges to determine the probability and effect of failure of components; social welfare workers "risk assess" their clients to evaluate the likelihood of the recurrence of antisocial behavior. Risk assessment has become a commonly used approach in examining environmental problems. It is used to examine risks of very different natures.

Environmental contamination problems are complex issues with worldwide implications. Risks to human and ecological health as a result of toxic materials or their introduction into the environment are a matter of great interest to modern society. The effective management of environmental contamination problems has therefore become an important environmental aim that will remain a growing social issue for the next years.

The foundations for risk assessment methodologies have traditionally been based on the examination of effects to human health, but much more emphasis is now placed on all types of environmental damage. In comparison to human health risk assessment, which is a relatively new field, risk assessment for ecological effects is very much in its infancy and the field is constantly developing.

ERA consists of evaluating the probability that adverse effects on the environment or human health occur or may occur as a consequence of exposure to physical, chemical or biological agents. Evaluation of environmental risk requires knowledge of adverse effects that might be caused by exposure to chemical substances or materials, as well as of the intensity and duration necessary to produce adverse effects on the environment, including the population.

Risk assessment is a tool used to organize, structure and compile scientific information in order to help identify existing hazardous situations, anticipate potential problems, establish priorities and provide a basis for regulatory controls and/or corrective actions. It can also be used to determine and measure the effectiveness of corrective measures or remedial actions. A key underlying principle of risk assessment is that some risks are tolerable — a reasonable and even sensible view, considering the fact that nothing is wholly safe per se. In fact, whereas large amounts of toxic substances may be of major concern, simply detecting a hazardous chemical in the environment should not necessarily be a cause for alarm. The intrinsic knowledge of the physical–chemical properties of pollutants, biodegradability, potential of bioaccumulation or potential effects of the chemical substances is necessary for the evaluation of environmental risk. Moreover, it is necessary to carry out a detailed evaluation of the emission sources, as well as the fate, transport and distribution in the different media. Due to all this, the analysis of environmental samples in the laboratory and the application of mathematical models are vital (EC, 1996).

4.3 FRAMEWORK OF ENVIRONMENTAL RISK ASSESSMENT

ERA is a formal mathematical tool used to evaluate potential hazards introduced by pollutant emissions in human health and the environment. This risk assessment process entails a sequence of actions outlined below:

1. **Hazard identification:** identification of the adverse effect that a substance has an inherent capacity to cause
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. **Dose–response assessment:** estimation of the relationship between dose, or level of exposure to a substance, and the incidence and severity of an effect
4. **Risk characterization:** 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

Figure 4.1 shows a framework for human and ecological ERA. The EU has provided a technical guidance document on ERA in support of 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). The U.S. Environmental Protection Agency has produced different risk assessment guidelines: Guidelines for Ecological Risk Assessment (EPA/630/R-95/002F. FR 63(93) 26846–26924), the Guideline for Exposure Assessment (EPA/600Z-92/001. FR 57: 22888–22938) and the Proposed Guideline for Carcinogenic Risk Assessment (EPA/600/P-92/003C. FR 61(79) 17960–18011).

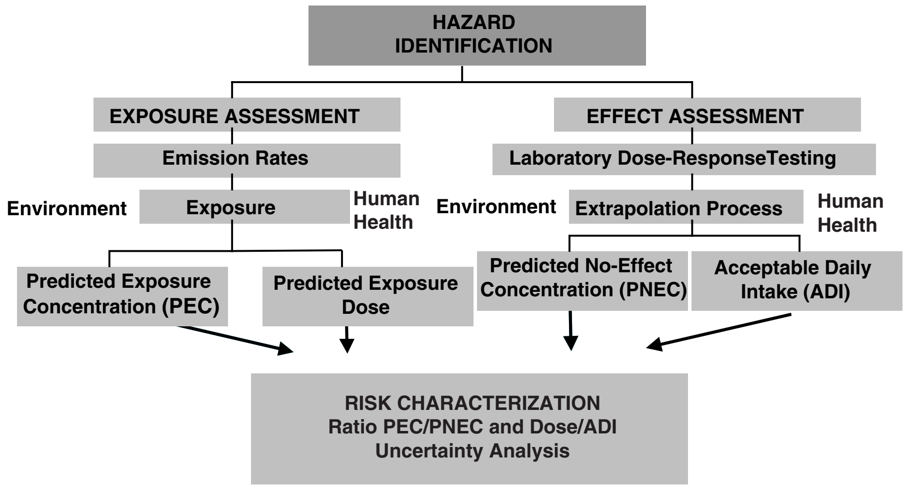


FIGURE 4.1 Framework of environmental risk assessment. (Adapted from Fairman, R. et al., *Environmental Risk Assessment — Approaches, Experiences and Information Sources*, European Environment Agency, Copenhagen, 1998.)

4.4 HAZARD IDENTIFICATION

The first step in an ERA in cases for human health and environment is to determine whether exposure of humans and ecosystems to chemicals is likely to have any adverse effects.

4.4.1 Human Health

The human health hazard identification involves an evaluation of whether a pollutant can cause an adverse health effect in humans. The process is a qualitative risk assessment that examines the potential for exposure and the nature of the adverse effect expected. The information used in hazard identification includes human, animal and mechanistic evidence; therefore, the risk assessor must evaluate the quality of the evidence, the severity of the effects, and whether the mechanisms of toxicity in animals are relevant to humans. The result is a scientific judgment of whether a particular adverse health effect in humans is caused by a chemical or process at certain concentrations. This is the work of toxicologists and epidemiologists, who study the nature of the adverse effects caused by toxic agent and the probability of their occurrence.

4.4.2 Ecosystems

The design of an ecological risk assessment program for an environmental contamination problem typically involves a process to define the common elements of populations and ecosystems clearly; this then forms a basis for the development of a logical framework that can be used for risk characterization. First, the development of an ecological risk assessment includes the identification of one or several ecological assessment endpoints — a very important point because the different types of ecosystems have unique combinations of physical, chemical, and biological characteristics and thus may respond to contamination in unique ways. The physical and chemical structure of an ecosystem determines how contaminants affect its resident species and biological interactions may determine where and how the contaminants are distributed in the environment and which species are exposed to particular concentrations. The following ecosystems are normally studied in an ERA:

- Terrestrial ecosystems are classified depending on the vegetation types that dominate the plant community and terrestrial animals.
- Wetlands are areas in which topography and hydrography create a zone of transition between terrestrial and aquatic environments.
- In freshwater ecosystems the dynamics of water temperature and movement of water can affect the availability and toxicity of contaminants.
- Marine ecosystems are of primary importance because of their vast size and critical ecological functions.
- Estuaries support a multitude of diverse communities and are important breeding grounds for numerous fish, shellfish and bird species.

Assessment endpoints, mentioned in Chapter 3 with regard to life-cycle impact assessment, are explicit expressions of the actual environmental value to be protected. The main criteria used in the selection of assessment endpoints include their ecological relevance, their susceptibility to the stressor, and whether they represent management goals (to include a representation of societal values). Ecological resources are considered susceptible when they are sensitive to human-induced stressors to which they are exposed. Delayed effects and multiple stressor exposures add complexity to evaluations of susceptibility. Conceptual models need to reflect these factors. If a species is unlikely to be exposed to the stressor of concern, it is inappropriate as an assessment endpoint.

To evaluate every species that may be present at a locale affected by an environmental contamination problem is not feasible. Therefore, the selected target of indicator species will normally be chosen in an ERA study. Then, by using reasonably conservative assumptions in the overall assessment, it is rationalized that adequate protection of selected indicator species will enable protection for all other environmental species as well.

A guiding criterion for the selection of ERA target species considers if they are:

- Threatened, endangered, rare or of special concern
- Valuable for several purposes of interest to human populations (i.e., of economic and societal value)
- Critical to the structure and function of the particular ecosystem that they inhabit
- Indicators of important changes in the ecosystem
- Of relevance for species at the site and its vicinity

4.5 EXPOSURE ASSESSMENT

Exposure assessment is the determination of the concentration/doses to which human populations or environmental compartments are or may be exposed. An exposure assessment is designed to estimate the magnitude of actual and potential receptor exposures to environmental contaminants, as well as the frequency and duration of these exposures, the nature and size of the populations potentially at risk (i.e., the risk group), and the pathways by which the risk group are or may be exposed.

The following steps must be taken in a typical exposure analysis for an environmental contamination problem:

- Determination of the concentrations of the chemicals of concern in each medium to which potential receptors are or may be exposed
- Estimation of the intakes of the chemicals of concern, using the appropriate case-specific exposure parameter values

The exposure assumptions election can be very difficult and is one of the critical elements of an ERA. Efforts have been made to standardize the process of exposure assessment, but the best approach remains to tailor the exposure assessment to the particular characteristics of the study. For instance, risk experts should visit the study area if possible and contact relevant agencies and individuals to assemble information regarding the habits and activities of local populations.

4.5.1 Exposure Assessment Data

One of the most important steps in the ERA process is the determination of potential exposure. Exposure estimation involves combining predicted concentrations for target chemicals with certain assumptions about the environmental fate of these chemicals and activity patterns of the receptors. Subsequently, the results of the exposure assessment include toxicity and epidemiologist information to provide a quantitative estimate of risk. Therefore the exposure assessment is based on representative monitoring data and/or on model calculations. Appropriate information on substances with analogous use and exposure patterns or analogous properties will be taken into account when available. However, the availability of representative and reliable monitoring data or the amount and detail of the information necessary to derive realistic exposure levels by modeling will vary.

4.5.1.1 Environmental Monitoring

Whenever possible, high-quality and relevant measured exposure data should be used in risk characterization. Measured exposure data may be available for existing substances, but are unavailable for new substances. The latter may be obtained from industry monitoring programs, particularly for occupational exposure, or other monitoring studies.

As a first step, the available data must be assessed with regard to their reliability. The confidence in measured exposure concentrations is determined by the adequacy of techniques, strategies and quality standards applied for sampling analysis and protocol.

Second, whether the data are representative must be established. The type, location, duration and frequency of sampling should be evaluated. The selected representative measured data need to be allocated to specific exposure scenarios to allow meaningful exposure assessment.

The types of environmental monitoring can be classified as follows:

- Biological monitoring allows actual measurement of exposure and accurate assessment of likely health outcomes. It involves analyzing human biological samples (i.e., blood, urine, hair, nails, or breast milk) for the presence of target chemicals.
- Environmental monitoring allows actual measurement of exposure and accurate assessment of likely ecological outcomes. It involves analyzing environmental samples (i.e., air, grass, soils, fish or shellfish) for the presence of target chemicals.

Monitoring is useful in assessing occupational exposures to airborne chemicals because workplaces typically involve exposure to a single or only a few chemicals at relatively high concentrations (in contrast to typical environmental concentrations) and exposure activity is well known. Although monitoring is a useful method, some disadvantages can be found. The main advantages and disadvantages of environmental monitoring are:

Advantages:

1. Defines environmental exposure accurately and precisely
2. Identifies associated health effects in a good way
3. Improves the determination of susceptibility to target pollutants

Disadvantages:

1. Biomarkers integrate all routes and sources of exposure; thus, it is impossible to distinguish whether the exposure is due to the chemicals in air, water, or food.
2. A distinction between variations in the exposed populations, such as health status and individual lifestyle, cannot be made.
3. The timing of sample collection in relation to exposure that is critical to the successful measurement of a biomarker cannot be considered.

The best marker would be one that was chemical specific, measured well in trace quantities, measurable in easily sampled biological media or by noninvasive techniques (i.e., blood, urine, hair or nails), and well correlated with a previous exposure. For instance, a good biomarker to assess the municipal solid waste incinerator (MSWI) emissions of our case study would ideally be associated with a chemical unique to the emissions, easily monitored in the stack, and associated only with inhalation exposure. Inorganic tracer chemicals for MSWI emissions include antimony, arsenic, beryllium, cadmium, chromium, lead, mercury, nickel and tin. Organic tracer chemicals include benzo(a)pyrene, polychlorinated biphenyls and dioxins. None of these chemicals is good for biomonitoring because each one exists naturally in the environment, so exposure may occur naturally via air, water, soil, and food. Although the organic compounds are not naturally occurring, they are inadvertently produced as an impurity in the manufacture of many chemicals or as a byproduct of many combustion processes and thus considered to be ubiquitous in the environment.

4.5.1.2 Fate and Exposure Models

Examination of the total exposure is a comprehensive evaluation that necessitates modeling. The models or the mathematical calculations for the exposure scenarios provide a means of calculating exposure levels. A chemical's final distribution in the media and its respective concentrations are the result of numerous highly complex and interacting processes that are not easy to estimate. Fate and transport models have been developed to estimate pollutant transport among and transportation within multiple environmental media.

Once a chemical has been emitted to a medium (air, water or soil), it is distributed in the environment. The distribution is not normally restricted to one environmental compartment, but is partitioned among different compartments. Therefore, it may cause one or more types of environmental impacts. Also, it can enter the food chain and become a risk to human beings. This ability depends on the specific physical, chemical or biological (toxicological) properties of the compound and of the properties and characteristics of the medium to which the emission is released. On the other hand, chemicals can suffer different processes (i.e., degradation, biodegradation, metabolization, transformation, dissociation, hydrolytic process, etc.) in the environment. In this way, fate and transport models can help resolve how a chemical will be distributed in the different media and which transformation may suffer.

Two different modeling approaches exist: 1) multimedia fate and exposure modeling and 2) specific single-medium models. Integrated multimedia fate and exposure models represent the distribution of a chemical among different compartments and the transfer of chemicals through various exposure routes to a species of interest. For human toxicity, the models calculate a potential dose, which is indicative of the level of impact expected. For ecological toxicity, the models calculate environmental compartment concentrations or potential doses for animals at different levels of the food chain.

The examination of the total exposure is a comprehensive evaluation most accurately carried out for micropollutants (organics and heavy metals) by the use of multimedia modeling. For macropollutants (SO_2, NO_x and particles) single-medium models are generally applied. Table 4.1 consists of selected models that may be applied to some aspects of risk assessment and environmental management problems. The choice of one particular model over another will generally be specific to the problem.

Environmental fate models determine the concentration in different compartments (air, surface water, sediments) through the solution of mass balance equations describing the release, transformation, and intercompartmental distribution of a pollutant. Exposure pathway models calculate the exposure of an organism via a stated pathway resulting from a given environmental concentration. These factors take into account transfer factors, uptake rates such as the rate of inhalation, partitioning and bioconcentration factors, and environmental concentrations.

TABLE 4.1
Environmental Models Applicable to Risk Assessment and Environmental Management Problems

Model	Description	Special features
	Single-medium models	
FAIR (framework to assess international regimes)	An interactive model to explore options for differentiation of future commitments in international climate policy making	A decision-support tool, allows interactively evaluating the implications of different approaches and criteria for international burden sharing because it enable users to relate burden-sharing schemes to global climate protection targets
TAF (racking and analysis framework)	Integrated modeling framework developed to guide U.S. regulatory policies on emissions of precursors to acid rain	Evaluates the status of implementation, effectiveness, costs and benefits of the acid-deposition control program; determine whether additional reductions are necessary to prevent adverse ecological effects
ISCST/ISCLT (industrial source complex short-/long- term model)	An air dispersion model that calculates annual ground-level concentrations and deposition values associated with point and area sources of air emissions	Used for predicting short-/long-term air concentrations; provides mechanisms to account for pollutant removal by physical or chemical processes
RAINS (regional acidification information and simulation)	Scenario-generating device allowing visualization of future impacts of current actions/inaction and to design a transition strategy toward long-term environmental goals	Brings information about costs of control, emissions, atmospheric transport and ecological impacts in a multipollutant/multieffect framework
Modflow (modular flow model)	Modular three-dimensional ground-water flow model	Simulates steady and nonsteady flow in an irregularly shaped flow system in which aquifer layers can be confined, unconfined or both

-- *continued*

TABLE 4.1 (continued)
Environmental Models Applicable to Risk Assessment and Environmental Management Problems

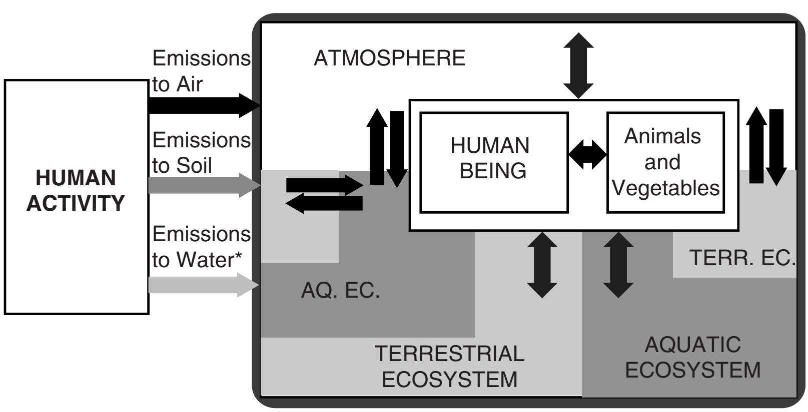
Model	Description	Special features
	Single models	
DREAM (Danish Rimpuff and Eulerian accidental release model)	Used for studying transport, dispersion and deposition of air pollution due to a single but strong source, such as an accident	Comprehensive three-dimensional tracer model based on a combination of a Lagrangian short-scale puff model (describe the transport, dispersion and deposition in the area near the source) and a Eulerian long-range (describe the long-term transport calculations)
TRIM (total risk integrated methodology)	Modeling system for assessing air pollutants and finding needs of the hazardous and critical air pollutant programs	Time series modeling system with multimedia capabilities for assessing human health and ecological risks for hazardous and criteria air pollutants
AQUATOX (simulation model for aquatic ecosystems)	Predicts the fate of different pollutants, such as nutrients and organic chemicals, and their effects on the ecosystem	Valuable tool for ecologists, water quality models and anyone involved in performing ecological risk assessments for aquatic ecosystems
	Multiple-media models	
CalTOX (California total exposure model)	A risk assessment model that mathematically relates the concentration of a chemical to the theoretical dose a person may receive	A multimedia, multiple pathway exposure, transport and transformation model
EUSES (European Union system for the evaluation of substances)	A multimedia transport and transformations model; estimation of emissions, distribution in the environment, exposure forecast of humans and environment, assessment of bioeffects and risk estimation for the environment and humans	End points are humans (consumers, workers, man exposed through environment) and environment (population of micro-organisms), aquatic, terrestrial and sediment ecosystems, populations of top predators; risk environmental and health human assessment of new and existing substances

MENDTOX (multimedia environmental distribution of toxics)	An environmental simulator designed to track the dynamic distribution of chemicals in the multimedia environment, in eight main compartments including: air, aerosol, soil, water, sediment, suspended solids, biota, and vegetation	Based on a detailed mechanistic description of intermedia transfer processes; incorporates theoretical and empirical descriptions of transport processes (gaseous, dissolved and particle phases) and the particle-size distribution and their dependence on intermedia transfer processes
SMCM (spatial multimedia compartmental model)	Describes the fate of organic chemicals in a conventional air, water, soil , and sediment system under steady-/unsteady-state condition	Based on estimation of the multimedia partitioning of organic pollutant in local environments; used to predict multimedia concentrations and to analyze the multimedia distribution of organic chemicals in the environment
SIMPLEBOX	A nested multimedia model that uses contaminant concentrations as input and computes steady-state and time-dependent (transient) concentrations as outputs	Terrestrial and aquatic fauna are not incorporated into the model; however contaminant concentrations in water bodies and sediments can be evaluated; uses contaminant concentrations as input and computes steady-state and time-dependent (transient) concentrations as outputs

The development of the ERA methodology was an important step forward with the inclusion of such models for pollutant fate calculations. Figure 4.2 shows a general overview of the exposure assessment by multimedia modeling for human health. This figure shows the influence of human activities on the environment and illustrates the connections of the different important impact pathways within the total exposure. Thus, this figure describes the main goal of multimedia modeling: to identify the different pathways of a pollutant fate and to calculate the total exposure due to the different sources. Starting with emission from the plant, the distribution of the pollutants goes over the air, soil, ground water and surface water in the other bordering compartments. Through these paths the pollutants can enter the food chain of human beings and animals. With the direct consumption of agricultural products, i.e., plants or animals, humans stand under the direct and indirect pollutant influence (see Figure 4.2).

Numerous model classification systems with different complexities exist in practice; these are broadly categorized as analytical or numerical models, depending on the degree of mathematical sophistication involved in their formulation. Analytical models are models with simplifying underlying assumptions, often sufficient and appropriate for well-defined systems for which extensive data are available and/or for which the limiting assumptions are valid. Whereas analytical models may be enough for some situations, numerical models (with more stringent underlying assumptions) may be required for more complex configurations and systems.

The procedure of deriving an exposure level by applying model calculations must be transparent and the input data or default values used for the calculations should be documented. Nowadays a large number of different models are available to describe an exposure situation, so the choice of the most appropriate model for the specific substances and scenario should be made and explained. The choice of which model will be used for specific applications depends on numerous factors;



*Emissions to superficial and underground water.

FIGURE 4.2 Cause–effect chain for ecosystem and human health as basis for exposure assessment by multimedia modeling.

choosing a more complex model over a simple one will not necessarily ensure a better solution in all situations. In fact, because a model is a mathematical representation of a complex system, usually some degree of mathematical simplification must be made about the system being modeled. Due to the complexity of natural systems, it is usually not possible to obtain all the input parameters, so data limitations must be weighted appropriately when choosing a model. Here ERA is confronted with the same problem as that of the LCA described previously because both are methods based on system analysis.

Ultimately, the type of model selected will be dependent on the overall goal for the assessment, complexity of the problem, type of contaminants of concern, and nature of the impacted and threatened media, as well as the type of corrective actions considered in the investigation.

4.5.2 Human Health Exposure Assessment

After estimation of the increasing concentrations of pollutants in air, water, soil, and food in the affected regions (by means of a monitorization, multicompartmental or single model), an exposure assessment must be carried out. The exposure assessment phase of human health risk assessment is the estimation of the rates at which chemicals are absorbed by potential receptors. Because most potential receptors can be exposed to chemicals from a variety of sources and/or in different environmental media, an evaluation of the relative contributions of each medium and/or source to total pollutant intake could be critical in a multipathway exposure analysis. In fact, the accuracy of exposure characterization could be a main determinant for the validity of a risk assessment.

Humans may be exposed to substances in their workplaces (occupational exposure), due to use of consumer products (consumer exposure) and indirectly via the environment. Different types of individuals and ages may be required to characterize the population at greatest risk. Also, frequent exposure to adults and children must be considered separately because adulthood provides the longest period of exposure and childhood accentuates some exposure routes (such as the incidental ingestion of soil) and potential sensitivities due to higher ratios of intake to body weight.

In a first step of the exposure assessment, the probability of an exposure of the population to the substances under consideration must be evaluated. Exposure levels and concentrations for each exposed population need to be evaluated based on available measured data and/or modeling.

A contaminant can enter the body using any of three pathways: ingestion, inhalation or by contact with the skin (dermal or other exterior surfaces such as eyes). Once in the body it can be absorbed and distributed to various organs and systems. The toxic may then be stored, for example in fat, as in the case of DDT, or it may be eliminated from the body by transformation into something else and/or by excretion. The biotransformation process usually provides metabolites more readily eliminated from the body than the original chemicals; however, metabolism can also convert chemicals into more toxic forms.

On the other hand, environmental exposure to chemicals can be direct (as a result of exposure to the media where the emission directly takes place) or indirect (as a result of exposure to a media in which the pollutants arrive by transport for another media where the emission takes place). Thus, all derived exposure levels should be representative of the exposure situation they describe. The duration and frequency of exposure, routes of exposure, human habits and practices, as well as technological processes, need to be considered. Furthermore, the spatial scale of exposure (e.g., personal, local, regional levels) must be taken into account.

The quantitative process of estimating exposure is straightforward. With the exception of the inhalation pathway, exposure is normally estimated as the rate of pollutant contact per unit of body weight:

$$Dose \equiv \frac{Concentration \cdot Contact\ Rate \cdot Frequency}{Body\ weight} \tag{4.1}$$

where *Dose* is the rate of exposure, *Concentration* is the level of pollutant in a particular environmental media, *Contact rate* is the amount (per time) of the media contacted, *Frequency* is a measure of how often (and over what period) exposure occurs, and *Body weight* is the weight of the individual.

For some exposure routes, the individual term of doses may include multiple parameters. For example, in estimating dermal pollutant intake during swimming, the contact rate is calculated as the product of (1) the surface area of the skin, (2) a chemically specific permeability, and (3) the density of water.

Exposure parameters are generally selected as a mix of typical and high-end values to afford an overall conservative bias. Although situation-specific values are always preferable, they are seldom available and often impractical to develop. Default values have been established for many parameters and some conventions have been yielded. For example, an average adult body weight of 70 kg is routinely used in dose calculations. Moreover, exposure profiles are subject to considerable discretion; the difficulty of exposure assessment is to choose a combination of assumptions that satisfies the aim of the assessment and is appropriate for the populations of interest. Implications of parameter variability and uncertainty are difficult to test with deterministic methods; probabilistic techniques such as those described in Chapter 5 can directly incorporate these aspects.

Risk assessments contain numerous uncertainties that are typically compensated for by conservative assumptions designed to bias risk estimates high. Recently, the philosophy has shifted toward the use of less conservatism. Most risk assessments conducted in the late 1980s were centered on extreme situations such as a maximally exposed individual (MEI). An MEI was built to receive (in theory) a level of exposure not likely to be exceeded by any person, a level that would be extremely improbable. More recent guidance, however, has recommended the use of reasonable maximum exposure (RME) scenarios that attempt to work out plausible, high-end exposure estimates. In reality, the difference between MEI and RME scenarios may be one of semantics because concepts such as plausible, maximum and high-end are too often subjective. Psychologically, however, the shift from MEI to RME implies a movement from the unlikely to the plausible and assigns a greater sense of realism to the risk estimates.

4.5.3 Ecological Exposure Assessment

Ecosystems may be exposed to chemical substances during all stages of their life-cycle — from production to disposal or recovery. For each environmental compartment potentially exposed, the exposure concentrations should be evaluated. The objective of the ecological exposure assessment is to estimate the concentration to which an environmental compartment is or may be exposed.

A chemical may be released into the environment and is then subject to physical dispersal into the air, water, soil, or sediment. The chemical may then be transported spatially and into the biota and perhaps be chemically or otherwise modified or transformed and degraded by abiotic processes (such as photolysis, hydrolysis, etc.) and/or by microorganisms present in the environment. The resulting transformation may have different environmental behavior patterns and toxicological properties from those of the chemical. Nonetheless, it is the nature of exposure scenarios to determine the potential for any adverse impacts. The amount of a target species' exposure to environmental contamination is based on the maximum plausible exposure concentrations of the chemicals in the affected environmental matrices. The total daily exposure (in mg/kg-day) of target species can be calculated by summing the amounts of constituents ingested and absorbed from all sources (e.g., soil, vegetation, surface water, fish tissue, and other target species) as well as those absorbed through inhalation and dermal contacts.

The process for the environmental risk assessment of a substance is based on the comparison of the concentration in the environmental compartment (predicted environment concentration, PEC) with a concentration below which unacceptable effects on organisms will most likely not occur (predicted no effect concentration, PNEC) as shown in Section 4.6.2. Therefore the aim of exposure assessment for the environment is the evaluation of PEC. It can be derived from available monitoring data and/or model calculations.

Analytical processes used to estimate receptor exposure to chemicals in various contaminated media (such as a wildlife or a game species' daily chemical exposure and the resulting body burden) are similar to those discussed under human health risk assessment.

4.6 DOSE–RESPONSE AND EXPOSURE–RESPONSE FUNCTIONS

Some experts argue that it is unnecessary to determine first if a chemical is hazardous. Their philosophy stems from the first definition of toxicology, given by Paracelsus (1493–1541) over 450 years ago: "All substances are poisons; there is none which is not a poison. The right dose differences a poison and a remedy." In other words, all chemicals have the potential to be hazardous, depending on the dose; therefore, exposure assessment could potentially take the place of hazard identification.

Dose–response and exposure–response evaluation, the third component of the risk assessment process, involves the characterization of the relationship between the dose administered or received and the incidence or severity of an adverse effect in the exposed population or ecosystem. Characterizing the dose–response relationship includes understanding the importance of the intensity of exposure, the concentration vs. time relationship, whether a chemical has a threshold level, and the shape of the dose–response curve. In the determination of a dose response and exposure response, the following aspects need to be considered: the metabolism of a chemical at different doses, its persistence over time, and an estimate of the similarities in disposition of a chemical between humans and animals.

The dose–response and exposure–response functions are based on toxicological dose-oriented and epidemiological exposure-oriented studies. Figure 4.3 proposes to illustrate the difference between the toxicological and the epidemiological approaches. Although the toxicological approach is based on bioassays or animal tests that allow determining dose–response functions, the epidemiological approach uses empirical studies in which correlations are established between exposure situations and observed human effects. In this way exposure–response functions, which allow an estimation of human effects depending on the exposure concentration, are calculated. Epidemiological studies focus more on macropollutants responsible for respiratory effects such as SO_2, NO_x and particles because they usually act together and it is quite difficult to conduct laboratory assays. 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 foundations to obtain toxicological information for micropollutants, i.e., heavy metals and the huge number of organic compounds like PCDD/Fs or PAH

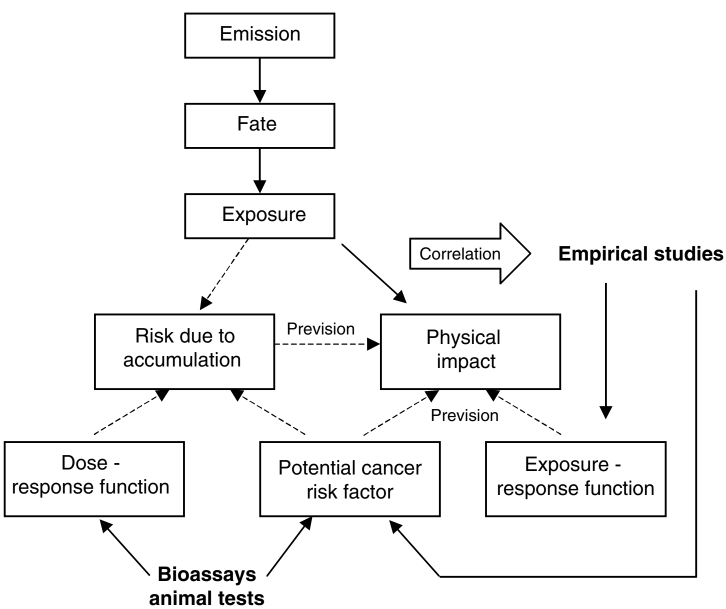


FIGURE 4.3 Comparison of the toxicological approach by bioassays and animal tests and the epidemiological approach using empirical studies to determine damage factors for human health impacts.

(polyaromatic hydrocarbons). Potential cancer risk factors are determined by both approaches and both types of damage functions are important sources of uncertainties — especially the question of extrapolation to lower doses and the correlations made implying insecurity.

4.6.1 Human Health Dose–Response and Exposure Response

Government agencies are charged with the protection of the public health and ecology; however, the U.S. Environmental Protection Agency (USEPA), the Agency for Toxic Substances and Disease Registry (ATSDR), and the U.S. Food and Drug Administration (FDA) must have a reference, or comparison value, upon which to base an evaluation of potential health threats posed by any substances or chemicals. The bases, or starting points, for such estimations may have different names or acronyms but they represent more or less the same thing. The risk of carcinogenic pollutants is estimated with a factor of potential cancer; however, the risk of noncarcinogenic pollutants is characterized by a reference dose. These values for noncarcinogenic endpoints are called oral reference doses (RfDs) and inhalation reference concentrations (RfCs) by the USEPA, acceptable daily intakes (ADIs) by the FDA, and oral and inhalation minimal risk levels (MRLs) by ATSDR (Table 4.2).

TABLE 4.2
Oral RfDs/ADIs/MRLs for Noncarcinogenic Effects of Some Different Chemicals

Regulated substance	RfDs (mg/kg-day)	ADIs (mg/kg-day)	MRLs (mg/kg-day)
Acetone	0.100	0.1	Oral int. 2
Cadmium	0.0005	5.10^{-4}	Oral chr. 0.0002
Chloroform	0.010	0.01	Oral acute 0.3 Oral int. 0.1 Oral chr. 0.01
1,1-Dichloroethylene	0.009	0.009	Oral chr. 0.009
Cis-1,2-Dichloroethylene	0.010	0.01	Oral acute 1 Oral int. 0.3
Methylene chloride	0.060	0.06	Oral acute 0.2 Oral chr. 0.06
Tetrachloroethylene	0.010	0.01	Oral acute 0.2 Oral chr. 0.06
Toluene	0.200	0.3	Oral acute 0.8 Oral int. 0.02
1,1,1-Trichloroethane	0.035	0.09	Inhalation acute 2 ppm[a] Inhalation int. 0.7 ppm
Xylene	2.000	2.0	Oral int. 0.2

[a]Just available inhalation effects.

Note: Oral RfDs and ADIs refer to chronic effects. MRL values refer to intense, acute and chronic effects.

RfDs, RfCs, MRLs and ADIs are very often construed as rigid threshold limits, above which toxicity is likely to occur. The truth, however, is that these values actually represent levels of a potential toxicant that are highly unlikely to represent any threat to human health over a particular or specified duration of daily exposures. The more frequently these levels are exceeded and the greater the excess, the more likely that some toxic manifestation will occur. Most definitely these guidance or reference values are not threshold values for the onset of toxicology in any exposed population. Health guidance values must be considered in the context of their intended role as mere screening or trigger values, as which they serve as tools for assisting in the determination of whether further evaluation of a given potential exposure scenario is warranted.

For a general environmental health risk assessment of an unspecified emitted pollutant it is necessary to divide the population into different groups depending on their sensibility, for instance, babies, children, adults, adults above 65 years of age, people with asthma, etc. A division is especially important for the assessment of noncarcinogenic pollutants with the effect of chronic illness. In the case of carcinogenic pollutants, it is generally sufficient to divide the population into two groups of adults and children (until a certain age) in order to consider the different physical conditions (e.g., breathing, surface of the body, etc.) and the different life-styles (e.g., children playing outside). For substances that induce a carcinogenic response, it is always conservatively assumed that exposure to any amount of the carcinogen will create some likelihood of cancer; that is, a plot of response vs. dose is required to go through the origin. Therefore, for noncarcinogenic responses, it is usually assumed that a threshold dose exists, below which no response will occur. As a result of these two assumptions, the dose–response curves and the methods used to apply them are quite different for carcinogenic and noncarcinogenic effects, as suggested in Figure 4.4 (Masters, 1991). Realize that the same chemical may be capable of causing both kinds of responses.

4.6.1.1 Toxicological Information: Carcinogenic Effect

A controversial point in the discussion of carcinogenic effects is the estimation of the dose–response functions for different pollutants. Animal tests administer various doses to observe toxic effect; however, because the doses in these tests are higher than those existing in the environment, the discussion deals with the possibility of an extrapolation from the higher concentration in the animal test to the lower concentration in the environment and the possible effects. It is necessary to take into account that, even with extremely large numbers of animals in a bioassay, the lowest risks that can be measured are usually to a small percentage. Because regulators attempt to control human risk to several orders of magnitude below that, no actual animal data will be anywhere near the range of highest interest.

Different mathematical models to extrapolate to the lowest level exist, but there is still a lot of uncertainty. With the main aim of protecting human health, the USEPA chooses the safer way to estimate the risk with an additional correction factor, which means an overestimation of risk (Olsen et al., 2000).

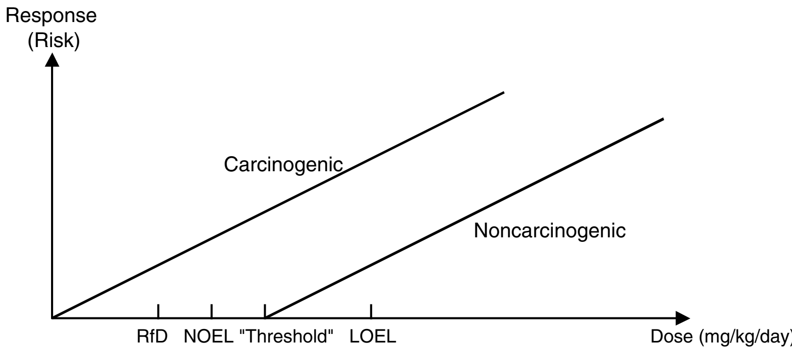


FIGURE 4.4 Dose–response functions for carcinogenic and noncarcinogenic pollutants. This is a schematic presentation; generally these curves are not linear.

4.6.1.2 Toxicological Information: Noncarcinogenic Effect

A threshold exists for noncarcinogenic substances; that is, any exposure below the threshold would be expected to show no increase in adverse effects above natural background rates. As for carcinogenic substances, in order to get knowledge of this threshold, it is necessary to conduct animal tests by changing the dose. The minimal dose at which a special effect appears is called LOEL (lowest observed effect level). The maximum dose without any special effect to the tested animal represents the NOEL (no observable effect level) (Olsen et al., 2000).

The RfD is defined as an estimation of the daily exposure of a human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime (Barnes and Dourson, 1988). However, it should not be concluded that all doses below a reference dose, or concentration, are acceptable. Despite this caution, most noncancer 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.

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 subchronic to chronic exposure durations, and to account for the quality and breadth of the database. A factor of 10 is usually the default value for the uncertainty factors. Values below 10 are sometimes used when sufficient data and justifications are available. LOEL is only used in the absence of NOEL, and an additional adjustment factor is required in this case to compensate for the lack of a NOEL estimate. For instance, the use of a large number of animals in a study may enhance NOEL certainty. An important point is that the factors may include an inconsistent margin of safety across different chemicals, contributing to varying RfDs, in terms of their conservatism.

Altogether, the values of the RfD represent a dose of approximately 1000 times the value below which it is not possible to observe an adverse effect (NOEL) in animals.

TABLE 4.3
Example of RfD Values and Potential Factors

	RfD (mg/day/kg)	Oral cancer factor (kg/day/mg)	Inhalation cancer factor (kg/day/mg)
As	3.0×10^{-4}	1.75	50
Cd	5.0×10^{-4}	—	6.3
Cr	1.0	—	42
Ni	2.0×10^{-2}	—	1.19
Pb	6.0×10^{-3}	—	—
Hg	3.0×10^{-4}	—	—
Sn	6.0×10^{-1}	—	—
Zn	3.0×10^{-1}	—	—
PCDD/Fs	4.0×10^{-9}	1.56×10^{5}	1.16×10^{5}

Source: IRIS (1996), EPA Integrated Risk Information System, available online at: http://www.epa.gov/ngispgm3/iris/.

The RfD has the unit milligram of a daily intake of the toxic element, which is absorbed in the body, divided by body weight (Table 4.3) (Olsen et al., 2000).

Baird et al. (1996) analyzed the RfDs reported in the USEPA'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 percentiles and 3% were above the 15th percentile. Such a conservatively biased approach is consistent with the objectives of many risk-screening assessments (Krewitt et al., 2002).

In carcinogenic and noncarcinogenic effects, it can be expected that some of the available data for existing substances have been derived from studies conducted *in vitro* — the basic (and perhaps additional) studies on genotoxicity, for example. *In vitro* studies may yield data on, for instance, metabolism and/or mechanism of actions (including studies in cell cultures from different species), dermal absorption (which may also be for different species) and various aspects of toxicity (e.g., test for cytotoxicity in different types of cells, macromolecule bonding studies, tests using embryo culture systems, and sperm mobility tests).

4.6.1.3 Epidemiological Information

Some data can be obtained from clinical studies of humans who have inadvertently been exposed to a suspected toxicant; thus, a source of information relating exposure to risk is obtained from epidemiological studies. Epidemiology is the study of the incidence rate of diseases in real populations. When attempting to find correlations between elevated rates of incidence of a particular disease in certain groups of people and some measure of their exposure to various environmental factors, an epidemiologist tries to show in a quantitative way the relationship between exposure and risk. Such data should be used to complement animal data, clinical data and scientific analyses of the characteristics of the substances in question.

From studies of former smog pollution episodes (e.g., London smog in the 1950s), it is known that very high ambient pollution concentration is associated with adverse health effects on the same day or on subsequent days. In the last 20 years numerous well-conducted epidemiological* as well as experimental** studies have confirmed this correlation between exposure to pollutants and the occurrence of health and environmental damages. Hence, they allow establishing a direct link between ambient concentration of certain pollutants and effects on different receptor endpoints, as indicated in Pilkington et al. (1997) and Beer and Ricci (1999). Under the assumption of linearity in incremental damage with incremental exposure, slope factors (SF) could be defined. Figure 4.5 shows some possible forms of exposure–response (E–R) functions as they have been found. E–R functions exist for human health effects, damages to material and crops, and harm to ecosystems. Unfortunately, at the moment sufficient epidemiological data are not available to address human health effects caused by the majority of chemicals. Therefore, the epidemiological approach must be combined with bioassays. Moreover, it should be taken into account that epidemiological data are criticized for 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).

According to European Commission (1995 and 2000) the E-R functions of macro-pollutants for human health can be subdivided into the seriousness of their effects:

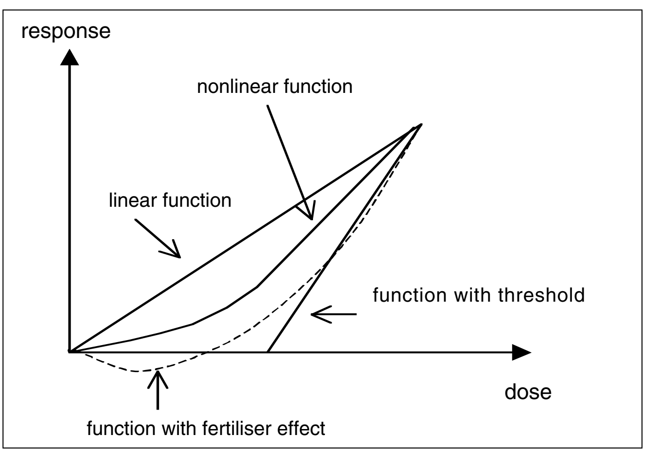


FIGURE 4.5 Possible forms of exposure–response functions. (Adapted from European-Commission-DGX11, *ExternE — Externalities of Energy*, ECSC-EC-EAEC, Brussels-Luxembourg, 1995.)

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* Epidemiological studies are statistical methods in which causal coherence between environmental pollutant concentration and the occurrence of cases of illness have been thoroughly investigated.

** Experimental studies are laboratory studies with animals and *in vitro*, whereby safety factors are used for the transfer of the obtained results from animal to human.

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.

Mortality

The effect of mortality can be expressed by fatal cases or years of life lost (YOLL), as previously explained in Chapter 3. However, since in recent years researchers have moved from studies based on fatal cases to YOLL (EC, 1995; EC 2000), YOLL is used as endpoint in this study. Table 4.4 illustrates the highly important E-R functions that use YOLL as endpoint.

For example the additional YOLL by sulphates are calculated in the following way:

0.0012 YOLL/(mg/m3) * 0.00082 mg/m3)/
functional unit * 423,000 pers.* 0.75 adults/pers. =
0.312 YOLL/ TWh

where Dc = 0,00082 ($\mu g/m^3$) 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)

Caution must be exercised in interpreting every epidemiological study because any number of confusing variables may lead to invalid conclusions. For example, a study may be biased because workers are compared with nonworkers (workers are usually healthier) or because relative rates of smoking have not been accounted for, or other variables that may be the actual causal agents may not even be hypothesized

TABLE 4.4
Mortality Functions Expressed in YOLL Due to Concentration Increments ($\mu g/m^3$) (IER, 1998)

Receptor	Category	Pollutant	Formula
Adults: Percentage of Population with Age above 18			
	'chronic' YOLL	PM10 / Nitrates	0.00072 * Dc * Pop. * adults
		Sulphates	0.0012 * Dc * Pop. * adults
	'acute' YOLL	SO_2	0.0719 * Dc * Pop. * b_m /100 * adults
		PM10 / Nitrates	0.04 * Dc * Pop. * b_m /100 * adults
		Sulphates	0.0677 * Dc * Pop. * b_m /100 * adults
		NO_x	0.0648 * Dc * Pop. * b_m /100 * adults

b_m: baseline mortality
Source: IER (1998)

in the study. As an example of the latter, consider an attempt to compare lung cancer rates in a city with high ambient air pollution levels with rates in a city with less pollution. Suppose the rates are higher in the more polluted city, even after accounting for smoking history, age distribution, and working background. To conclude that ambient air pollution is causing those differences may be totally invalid. Instead, different levels of radon may be in homes, for example, or differences that are causing the cancer variations may exist in other indoor air pollutants associated with the type of fuel used for cooking and heating.

4.6.1.4 Quantitative Structure Activity Relationship

When data do not exist or are limited for a given endpoint, the structure activity relationship (SAR) has been used lately by scientists. It should be noted that SAR techniques and methods, particularly for quantitative structure activity relationship (QSAR) techniques, have been created recently and are not well developed in relation to mammalian toxicology. The SARs, which are used for risk-assessment purposes, are usually more of the "expert judgment" type.

4.6.2 Ecosystems (Environment)

The procedure for the environmental risk assessment of substances consists in comparing the concentration in the environmental compartments (PEC) with the concentration below which unacceptable effects on organisms will most likely not occur (PNEC).

The PNEC values are usually determined on the basis of results from monospecies laboratory tests or of established concentrations from model ecosystem tests, taking into account adequate safety factors. A PNEC is regarded as a concentration below which an unacceptable effect will most likely not occur. In principle, the PNEC is calculated by dividing the lowest short-term L(E)C50 or long-term NOEC value by an appropriate assessment factor. The assessment factors reflect the degree of uncertainty in extrapolation from laboratory toxicity test data for a limited number of species to the "real" environment. Assessment factors applied for long-term tests are smaller because the uncertainty of the extrapolation from laboratory data to the natural environment is reduced. For this reason long-term data are preferred over short-term data.

Because aquatic organisms are exposed for a short period to compounds with an intermittent release patterns, short-term L(E)C50 values are used to derive a $PNEC_{water}$ for these compounds. For most compounds, data will probably not be present for sediment-dwelling organisms. Appropriate test systems are under development but standardized guidelines are not yet available. A method to compensate for this lack of toxicity data, known as the equilibrium partitioning method, is used to derive a $PNEC_{sed}$. Toxicity data are also scarce for the soil compartment. When such data are present, they will normally include only short-term studies. In cases in which data are missing, the equilibrium partitioning method can be used to derive a $PNEC_{soil}$. For the atmosphere, biotic and abiotic effects like acidification are addressed. Due to the lack of suitable data and unavailability of adequate methods to assess both types of effects, a provisional strategy is used.

The main function of risk assessment is the overall protection of the environment. However, certain assumptions are made concerning the aquatic environment that allow an uncertainty extrapolation to be made from single-species, short-term toxicity data to ecosystem effects. It is assumed that ecosystem sensitivity depends on the most sensitive species and that protecting ecosystem structure protects community function. These two assumptions have some important consequences. When the most sensitive species to the toxic effects of a chemical in the laboratory is established, extrapolation can subsequently be based on the data from that species. Furthermore, the functioning of any ecosystem in which that species exists is protected, provided the structure is not sufficiently distorted to cause an imbalance. It is accepted that protection of the most sensitive species should protect the structure.

With regard to the assessment of impacts that affect ecosystems, E-R or damage functions have been developed for acidic deposition on natural and semi-natural terrestrial ecosystems. That means it accounts for the impact corresponding to the sub-area of protection (see Chapter 3) 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 analyzed by taking into account predefined background conditions. The relative exceedance factor fRE 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 (4.2)). The REW ecosystem area indicator is expressed in km^2 and is obtained by the multiplication of f_{RE} by the ecosystem exceeded area A_{EE} where the critical load is exceeded by the pollutant (see Expression (4.3)) (IER, 1998).

$$f_{RE} = \Delta c / C_{CL} \tag{4.2}$$

$$REW = f_{RE} * A_{EE} \tag{4.3}$$

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

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

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

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

Moreover, damage functions exist for environmental damage 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. Damages to material refers to surface damage, especially in buildings, bridges and cars, due to acidic deposition. However, not included are cultural damages, e.g., to ancient cathedrals, due to their intrinsic value as mankind's patrimony.

4.7 RISK CHARACTERIZATION AND CONSEQUENCE DETERMINATION

The last stage of the risk assessment process, risk characterization, involves a prediction of the probability and severity of health and ecological impact in the exposed population and environmental damages. That is, the information from the dose–response evaluation (What human dose or PEC is necessary to cause an effect?) is combined with the information from the exposure assessment (What human dose or PEC is the population or ecosystem receiving?) to produce an estimate of the likelihood of observing an effect in the population or ecosystem being studied. An adequate characterization of risks from hazards associated with environmental contamination problems allows risk management and corrective action decisions to be better focused. To the extent feasible, risk characterization should include the distribution of risk among the target populations.

4.7.1 Human Health Risk

Human health risk characterization is the estimation of whether adverse effects are likely to affect humans who are exposed to certain substances. This process includes the comparison of the hazard and dose–response information, usually derived from animal experiments or *in vitro* test systems, with data on human exposure levels, as has been explained before. For notified new substances, risk characterization should take account of each adverse effect for which the substances have been assigned a hazard classification, together with any other effect of possible concern. Workers, consumers and humans exposed indirectly via the environment must be considered.

The health risk to exposed populations from exposure to environmental pollutants is characterized by the calculation of noncarcinogenic hazard quotient and indices and/or carcinogenic risks. These parameters can then be compared with benchmark criteria or standards in order to arrive at risk decisions about an environmental pollution problem.

4.7.1.1 Carcinogenic Risk to Human Health

Cancer risk expresses the likelihood of suffering cancer due to a definite daily intake of a pollutant. In this way, carcinogen risk is defined by the incremental probability of an individual developing cancer over a lifetime as a result of exposure to carcinogenic substances. It can be described by a dose–response estimate. Cancer risk is nonthreshold; this means that even the lowest doses have a small, or finite, probability of generating a carcinogenic response. Although risk decreases with the dose, it does not become zero until the dose becomes zero. For the characterization of cancer risk, the specific exposure is compared with a corresponding health benchmark for the relevant contaminant. The results correspond to the probability that cancer occurs, e.g., a value of 10^{-6} means that the probability exists that one person in a million may get cancer due to the study exposition.

In this way, cancer risk is estimated as excess risk (ER). ER does not express total cancer. It is an incremental risk due to exposure to the considered pollutant. In general, risks associated with the inhalation and noninhalation pathways (oral pathways) may be considered separately, and therefore different human health benchmarks are required. They can be estimated in accordance with the following generic relationships:

$$\text{Inhalation risk} = \text{air ground-level concentration (GLC)} \ (\mu g/m^3) \times \text{unit risk} \ (m^3/\mu g)$$

$$\text{Noninhalation risk} = \text{dose (mg/kg-day)} \times \text{potency slope} \ ((\text{mg/kg-day})^{-1}) \quad (4.4)$$

Therefore, different human health benchmarks are required. Inhalation risk of carcinogenic substances is assessed using the inhalation unit risk factor, while the noninhalation risk is the oral cancer slope factor. The latter (cancer slope factors) do not represent a safe exposure level, but relate the exposure with the probability of causing carcinogenic effects (Gold et al., 1995).

A total pathway risk can be calculated by summing the cancer risk estimates to each pathway. However since cancer risk describes the probability of developing cancer over a lifetime, the entire duration of exposure must be considered for risk assessment.

4.7.1.2 Noncarcinogenic Risk to Human Health

Depending on the exposure level, adverse health effects other than cancer can be associated with all chemical substances. Therefore, a noncancer risk characterization is always a dose–response analysis that determines whether the actual human exposure exceeds a defined exposure level. This critical exposure level represents the threshold below which adverse effects are assumed to be unlikely and it is determined in a toxicity assessment.

The human health benchmarks most widely used are the RfD for oral exposure and the RfC for inhalation of contaminants. RfD and RfC are lower-bound estimates of the NOAEL (no observed adverse effects level) of a pollutant, expressed for the different types of human exposure. These human health benchmarks are established for chronic exposure and do not account for acute toxicity of a pollutant.

Noncancer risks are expressed by the hazard quotient (HQ), which relates the exposure to the RfD and RfC, respectively (U.S. Environmental Protection Agency, 1989). HQ refers only to the potential for some individuals to be affected and cannot address the absolute level of risk. If HQ is higher than 1, this does not necessarily indicate a potential health risk. In consequence, noncarcinogenic quantitative estimates only identify the exposure level below which adverse effects are unlikely but say nothing about incremental risk for higher exposure. Although cancer risks are expressed as an increased probability of the occurrence of carcinogenic effects due to additional exposure, noncancer risks are assessed for the total exposure to a pollutant.

4.7.2 Ecological Risk

Having conducted the exposure assessment and the dose (concentration)–response (effect) for all environmental compartments, risk characterization is carried out by comparing the PEC with the PNEC. This is done separately for each of the protection goals identified before, for instance: aquatic ecosystem, terrestrial ecosystem, atmosphere, top predator, microorganisms in sewage treatment plants, etc.

For the risk characterization of the aquatic and terrestrial ecosystems a direct comparison of the PEC and PNEC values must be carried out. If the PEC/PNEC ratio is greater than one, the substance is "of concern" and further action must be taken.

For the air compartment, only a qualitative assessment of abiotic effects is carried out. If there are indicators that one or more of these effects occur for a given substance (for example, for ozone depletion substances), expert knowledge and consulting such as that provided by the responsible body in the United Nations Environment Program (UNEP) will be necessary.

4.8 IMPACT PATHWAY ANALYSIS (IPA)

Impact pathway analysis (IPA) has been lately introduced as a simplified way to assess the environmental fate and exposure of emissions to air; it allows the expression of effects in physical impact parameters, such as cancer cases and restricted activity days, that can be evaluated in monetary terms. IPA takes into account damages on a regional level due to pollutants with a long residence time and last used exposure–response functions based on epidemiological studies and also to dose–response functions based on toxicological tests.

IPA is also known as a damage function or "bottom-up" approach that traces the passage of pollutants from the place where they are emitted to the endpoint, i.e., the receptor that is affected by them. The approach provides a logical and transparent way of quantifying environmental damages, i.e., externalities. This methodology was developed through the ExternE-Project (Externalities of Energy) in a collaborative study between the European Commission and the U.S. Department of Energy.

The difference between IPA and the earlier used "top-down" damage assessment methodologies (Hohmeyer, 1992; Friedrich and Voss, 1993) is that in IPA 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 principal steps are illustrated in Figure 4.6 and described as follows:

- **Activity and emission:** characterization 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 affected regions via atmospheric dispersion models and chemical reactions (e.g., SO_2 transport and transformation into sulfates)
- **Receptor response and physical impact:** characterization 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 sulfates)
- **Monetization and cost accounting:** economic valuation of the mentioned impacts, determination of external costs that have not been internalized by governmental regulations (e.g., multiplication of the monetary value with the asthma incidents gives the damage costs).

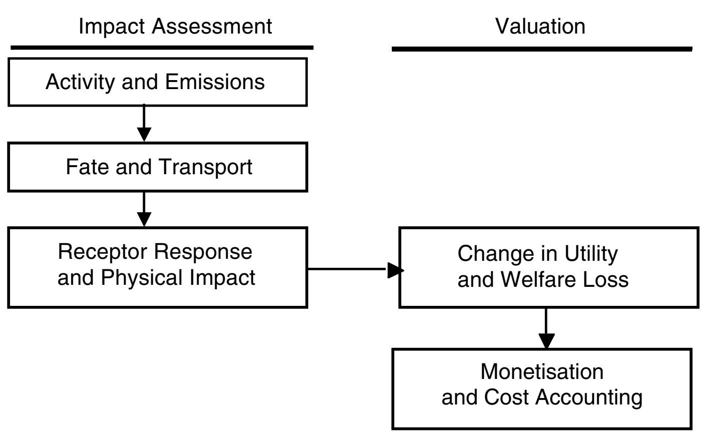

FIGURE 4.6 Illustration of the main steps of the IPA. (From European-Commission-DGXII, *ExternE — Externalities of Energy*, ECSC-EC-EAEC, Brussels-Luxembourg, 1995.)

Although the IPA is a complex approach, its application is quite easy thanks to the support of 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, further details are given about this model.

EcoSense stems from the experiences learned in the ExternE project (EC, 1995; EC, 2000) to support the assessment of priority impacts resulting from 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 modeling. Version 3.0 of EcoSense covers 13 pollutants, including the "classical" pollutants SO_2, NO_x, particulate matter and CO, and photochemical ozone creation as well as some of the most important heavy metals and hydrocarbons, but does not include impacts from radioactive nuclides.

In view of increased 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 harmonized European-wide database supporting the assessment of environmental impacts from air pollution. In the 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 realized that country-specific data sources and grid systems were hardly compatible when the analysis had to be on a European scale. Thus, 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 assessment of external costs to guarantee a harmonized and standardized 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 standardized 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
- To support easy modification of assumptions for sensitivity analysis

Because 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 easy integration of new scientific findings into the system. As a consequence, all the calculation modules (except for the integrated ISCST model) were designed so that they were model interpreters rather than models. Model specifications, 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 easy modification of model parameters; at the same time the model should not necessarily appear as a black box because the user can trace back what the system is actually doing.

Figure 4.7 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 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 (IER, 1998).

4.8.1 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 modeling including, e.g., emission factors, flue gas characteristics, stack geometry and the geographic coordinates of the site.

4.8.2 Reference Environment Database

The reference environment database is the core element of the EcoSense database, providing data on the distribution of receptors and meteorology, as well as a European-wide emission inventory. All geographical information is organized using the EUROGRID coordinate system, which defines equal-area projection grid cells of 10,000 and 100 km^2 (Bonnefous and Despres 1989), covering all EU and European

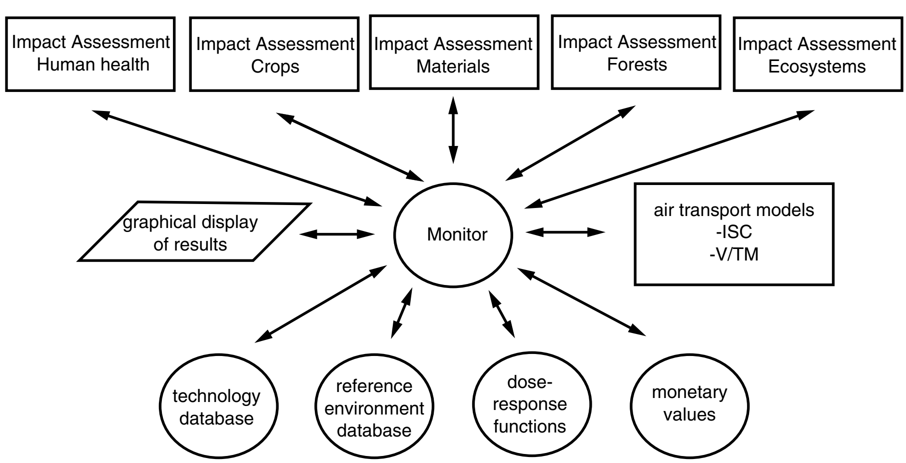

FIGURE 4.7 Structure of the EcoSense model. (From European-Commission-DGXII, *ExternE — Externalities of Energy*, ECSC-EC-EAEC, Brussels-Luxembourg, 1995.)

non-EU countries. Data on population distribution and crop production are taken from the EUROSTAT REGIO database and, 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. Meteorological data (precipitation, wind speed and wind direction) and a European-wide emission inventory for SO_2, NOx and NH_3 from EMEP (Sandnes and Styve, 1992), transferred to the EUROGRID-format, are also included for the long-range pollutant transport model.

4.8.3 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. Examples are given in Section 4.6.

4.8.4 Monetary Values

The database provides monetary values for most of the impact categories following the recommendations of the ExternE economic valuation task group according to guidelines from the European Commission, explained in Chapter 3.

4.8.5 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 end point as well as a more automated analysis including a range of prespecified impact categories.

4.8.6 Presentation of Results

Input data as well as intermediate results can be presented on several steps of the IPA in 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 program.

4.8.7 Air Quality Models

A special feature of EcoSense is the fact that air quality models are included. Apart from the local-scale ISCST-2 or -3 model, for which a set of site-specific meteorological data must be added by the user, a long-range pollutant transport model is included; both models have also been applied separately in this study for the calculation of site-dependent impact factors.

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. For these reasons, the computation of ambient air concentrations of primary pollutants on a local scale is done with a model that 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 Gaussian plume model. The concentration distribution from a continuous release into the atmosphere is assumed to have a Gaussian shape (see Expression (4.5)):

$$c(x,y,z)=\frac{Q}{u2\pi\sigma_y\sigma_z}\cdot\exp\left[-\frac{y^2}{2\sigma_y^2}\right]\cdot\left(\exp\left[-\frac{(z-h)^2}{2\sigma_z^2}\right]+\exp\left[-\frac{(z+h)^2}{2\sigma_z^2}\right]\right) \quad (4.5)$$

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
σy = 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 idealized 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 4.8). 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 justified for a statistical evaluation of a long period, where mutual changes in wind direction cancel out each other, rather than for an evaluation of short episodes.

In this study the Industrial Source Complex Short Term model, version 2 (ISCST-2) of the U.S. EPA (1992) and version 3 (ISCST-3) of U.S. EPA (1995a) in the form of BEEST (Beeline, 1998) have been applied. The model calculates hourly concentration values of gases and particulate matter for 1 year at the center of each specified grid. Effects of chemical transformation are neglected. Annual mean values are obtained by temporal averaging of the hourly model results. Currently U.S. EPA has proposed the establishment of a new regulatory dispersion model AERMOD (U.S. EPA, 2002).

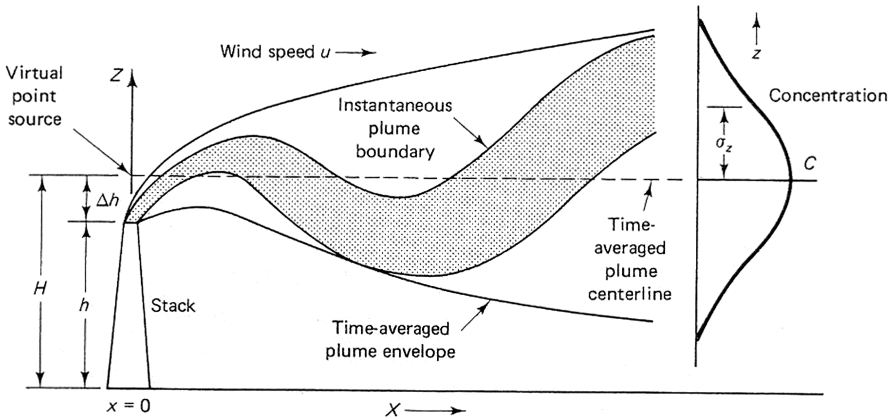

FIGURE 4.8 Gaussian plume shapte (H mixing layer height, h – stack height, H – plume height assuming flat terrain, H* – plume height above terrain).

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.

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 that the pollutants have vertically been mixed throughout the height of the mixing layer of the atmosphere and that 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 (WTM) used in EcoSense to estimate the concentration and deposition of acid species on a regional scale was originally developed at Harwell Laboratory by Derwent and Nodop (1986) for atmospheric nitrogen species, and extended to include sulfur species by Derwent et al. (1988). The model is a receptor-oriented 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 h from the receptor point. In addition to dealing with primary pollutants, the WTM is also able to calculate concentrations of secondary sulfate and nitrate aerosols formed from emissions of SO_2 and NO_x, respectively. The chemical reaction schemes implemented in the model are shown in Figure 4.9.

As we have seen, IPA and ERA sharer many similarities. In Figure 4.10 the steps of both methods are compared.

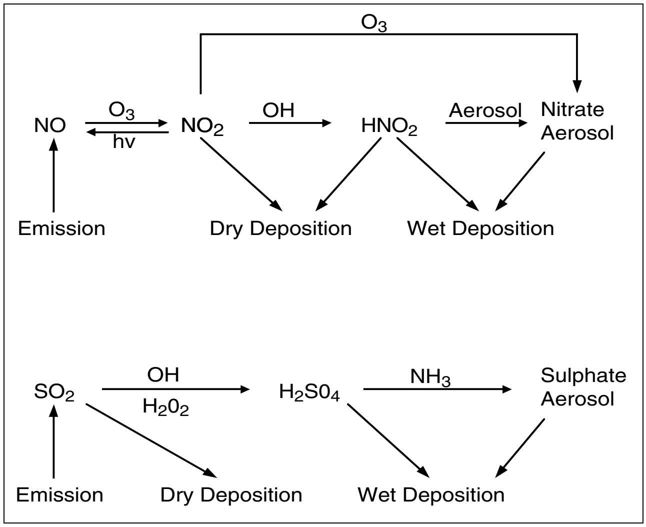

FIGURE 4.9 Chemical scheme in WTM. (From European-Commission-DGXII, *ExternE — Externalities of Energy*, ECSC-EC-EAEC, Brussels-Luxembourg, 1995.)

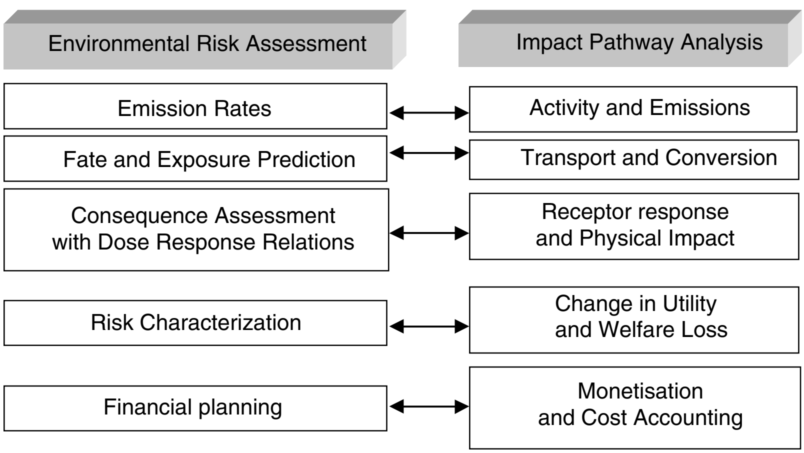

FIGURE 4.10 Steps of an IPA compared to those in a conventional ERA.

4.9 THE ROLE OF RISK ASSESSMENT IN ENVIRONMENTAL MANAGEMENT DECISIONS

The risk assessment proposes to give the risk management team the best possible evaluation of all available scientific data, in order to arrive at justifiable and defensible decisions on a wide range of issues. In relation to the types of risk management actions necessary to an environmental contamination problem or hazardous situation, decisions could be made based on the results of a risk assessment. In fact, risk-based decision-making will generally result in the design of better environmental management programs because risk assessment can produce more efficient and consistent risk reduction policies; at the same time, it can also be used as a screening tool for setting priorities.

Risk management based on the risk assessment result is the rpocess of deciding what has to be done. Given the estimates of risk established, political and social judgment is required to decide which risk is acceptable; therefore an acceptable risk must be defined. On the other hand, risk assessment has several specific applications that could affect the types of decisions to be made in relation to environmental management programs. Further general discussion of some of the more prominent applications scenarios is offered below.

4.9.1 Environmental Impact Assessments

Environmental impact assessment (EIA), as described in Chapter 1, is the analysis of the likely environmental consequences associated with a determined human activity. The main goal of an EIA is to ensure that environmental aspects are incorporated into planning for decision-making on and implementation of development activities. Typically, an identification of factors contributing the most to overall risks of exposures to the environmental hazards of concern may be included. It may also incorporate an analysis of baseline risks, as well as a consistent process to document potential public health and environmental threats from the relevant activity. EIAs are designed to prevent or minimize the adverse impacts of a development activity while maximizing its beneficial effects.

As an example, quantitative risk assessment is often undertaken as part of the siting process for newly proposed facilities. In many cases it is carried out as a regulatory requirement for EIAs or, sometimes, is used for operating facilities to evaluate implications from design changes or changes in exposure parameters. In fact, the risk assessment of stack emissions from MSWI facilities seems to be one of the most important applications for this type of evaluation. This becomes necessary because MSWIs typically release various potentially toxic compounds — some of which escape pollution control equipment and enter the outside air. These chemicals may include metals (e.g., arsenic, cadmium, and mercury), organic compounds such as polychlorinated dibenzo(p)-dioxins and furans (PCDD/Fs) (Zemba et al., 1996).

4.9.2 Air Toxics

An air pathway exposure assessment (APEA) is a systematic approach to address air pollutant emissions from a variety of sources. This approach includes the application of modeling and monitoring methods to estimate pollutant emission rates and their concentrations in air. The goal of an APEA is to evaluate the extent of actual or potential receptor exposures to air pollutants. This includes emission quantification, modeling of environmental transport and fate by means of a determination of the effects of the atmospheric processes on pollutant fate and transport, identification of potential exposure routes and populations potentially at risk from exposures, and the estimation for various exposure periods (short- and long term).

The air pathways methods of analysis in relation to air pollutant emission find several specific applications in situations such as the estimation of VOC emissions rates from landforms — sludge, landfills, etc. (see, e.g., Mackay and Leinonen, 1975; Mackay and Yeun, 1983; Thibodeaux and Hwang, 1982) and particulate inhalation exposure from fugitive dust (see, e.g., CAPCOA, 1990; U.S. Environmental Protection Agency, 1989).

4.9.3 Potential Risks of Contaminant Vapors into Building

The migration of subsurface contaminant vapors into buildings can become a very important source of human exposure via the inhalation route. Although the degree of dilution in the indoor air of a building is generally far less than the situation outdoors, contaminant vapors entering or infiltrating into a building structure may represent a significant risk to occupants of the building. An application could be a risk characterization scenario involving exposure of populations to vapor emissions from cracked concrete foundations or floors that can be determined for this situation, in order to adopt responsible risk management and/or mitigating measures.

4.9.4 Environmental Assessment and Characterization

Environmental assessments are invariably a primary activity in the general processes involved in the management of environmental assessment. The main goal of environmental assessments is to determine the nature and extent of potential impacts from the release or threat of emission of hazardous substances.

First, a representative sampling from the potentially contaminated media must be collected, along with historical data and sufficient details about the likely environmental contaminants. Sampling programs can be designed to search for specific chemical pollutants that become indicator parameters for sample analysis. For instance, a multimedia approach to environmental characterization can be adopted for most environmental pollution problems so that the significance of appropriate field sampling and analysis procedures increases. The activities included are expected to provide high-quality environmental data needed to support possible corrective action response decisions. To accomplish this, samples are collected and analyzed for the pollutants of concern. Proper protocols in field sampling and laboratory analysis procedures are used to help minimize uncertainties associated with data collection and evaluation activities.

The results from these activities will be a complete analysis of the pollutants detected in the environment. Risk assessment techniques and environmental characterization are typically applied to provide the development of effectual environmental characterization programs. Thus, the information obtained is used to evaluate current and potential future risks to human health and the environment. In addition to information about the nature and magnitude of potential risks associated with potential environmental contamination problems, risk assessment also affords a basis for judging the need for mitigating actions. On this basis, corrective actions are developed and implemented with the main goal of protecting public health and the environment.

4.9.5 Corrective Action

A variety of corrective action strategies may be applied in case of contaminating processes to restore sites into healthier and more ecologically sound conditions. The processes involved will generally incorporate a consideration of the complex interactions among the environment, regulatory policies, and the technical feasibility of remedial methodologies. A clear understanding of the fate and behavior of the pollutants in the environment is essential for developing successful corrective action response programs, and also to ensure that the problem is not exacerbated.

The design of corrective action response programs for contaminated site problems includes various formalized steps. In general, when the existing site information has been analyzed and a conceptual understanding of a site is obtained, then remedial action objectives should be defined for all impacted media at the contaminated site. Subsequently, alternative site restoration programs can be developed to support the requisite corrective action decision. Overall, risk assessment plays a very important role in the development of remedial action objectives for contaminated sites, the identification of feasible remedies that meet the remediation objectives, and the selection of an optimum remedial alternative.

Risk assessment has become particularly useful in determining the level of clean-up most appropriate for potentially contaminated sites. By utilizing methodologies that establish clean-up criteria based on risk assessment principles, corrective action programs can be conducted in a cost-effective and efficient manner.

These procedures can help determine whether a particular remedial alternative will pose unacceptable risks following implementation and to determine the specific remedial alternatives that will result in the least risk upon achieving the clean-up goals or remedial action objectives for the site. Consequently, risk assessment tools can be used as an aid in the process of selecting among remedial options for contaminated sites.

4.9.6 Ecological Risks

Often, especially in the past, only limited attention has been given to the ecosystems associated with contaminated sites, as well as to the protection of ecological resources during site remediation activities. Instead, much of the focus has been on the protection of human health and resources directly affecting public health and safety. In recent times, however, the ecological assessment of contaminated sites has gained considerable attention. This is the result of prevailing knowledge or awareness of the intricate interactions between ecological receptors and systems and contaminated site clean-up processes.

In fact, remedial actions can alter or destroy aquatic and terrestrial habitats; the consequences of ecosystem disturbances and other ecological effects must therefore be given adequate consideration during the corrective action response process. Thus, it is important to integrate ecological investigation results and general concerns into the overall site clean-up process.

4.9.7 Environmental Management Strategies

Environmental pollution problems have reached an important level in most societies globally because pollution through chemicals represents a significant portion of the overall problem of environmental protection. The effective management of environmental pollution problems has certainly become an important environmental priority that will remain a growing social concern for next future. This is mainly because of the numerous complexities and inherent uncertainties involved in the analysis of such problems.

Whatever the cause of an environmental pollution problem, the impacted media must be remedied. However, restoration or clean-up may not be economically or technically feasible. In this case, risk assessment and monitoring the situation, together with institutional control measures, may be acceptable risk management strategies in lieu of remediation.

Overall, a risk assessment will generally provide the decision maker with scientifically defensible procedures for determining whether a potential environmental pollution problem could represent a significant adverse health effect, and environmental pollution problems could represent a significant adverse candidate for mitigative actions. In fact, the use of health and environmental risk assessments in environmental management decisions in particular, and a corrective actions program in general, is becoming an important regulatory requirement in several places. For instance, a number of environmental regulations and laws in various jurisdictions increasingly require risk-based approaches in determining clean-up goals and related decision parameters.

4.10 EXAMPLE: COMPARISON OF TWO FATE AND EXPOSURE MODELS

The choice of which model must be used for an application is a big issue. Next we will present two different models, EUSES and CalTOX, and explain how differently they work. EUSES has been developed in Europe to assess the risk of new or existing chemicals produced and CalTOX was released by the California EPA as a tool to assist in health-risk assessment. They differ significantly in their functional properties and limits, necessary inputs and results. Figure 4.11 shows a screen shot of EUSES depicting the typical outline section and an opened data input window. Figure 4.12 shows a screen shot of the Excel spreadsheet with the input and output sections of CalTOX.

An important point is the input data (physical, chemical, toxicological and other properties of the contaminant) that the models need for calculation. Figure 4.13 depicts the amount of exclusive inputs of CalTOX and EUSES. The input data

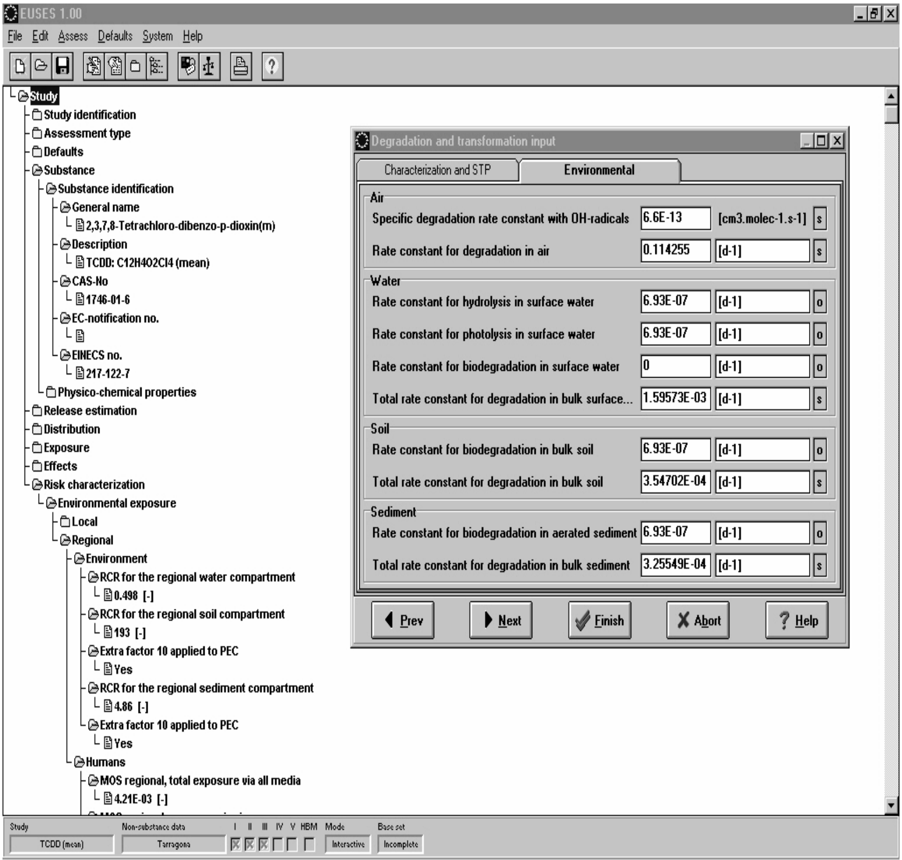

FIGURE 4.11 Screen shot of EUSES.

characterize the reference substances and the reference area in study. Output data will indicate which model could give the needed results, i.e., which model can provide a solution to the problem. Figure 4.14 shows a view of the results that the two models have in common, consisting of concentrations and some daily human doses. The risk is evaluated in a different way for the two models. The exclusive outputs of each model reflect the different approaches and areas of use for which it is designed. With 61 exclusive outcomes, EUSES offers almost twice as many results as CalTOX with its 34 exclusive outputs.

Further advantages and disadvantages of EUSES and CalTOX are presented. Beneficially, CalTOX as an Excel spreadsheet offers dynamic data exchange (DDE) and many data import and export formats enabling comfortable further processing (e.g., graphical visualization or uncertainty/sensitivity analysis) of its results. A good range of substances can be assessed with CalTOX, but the variety of output data as well as spatial and time flexibility and assessable endpoints are less comprehensive compared to those of EUSES. Perhaps the best resources of EUSES are its great amount of output data, assessable endpoints (environment and humans) and spatial

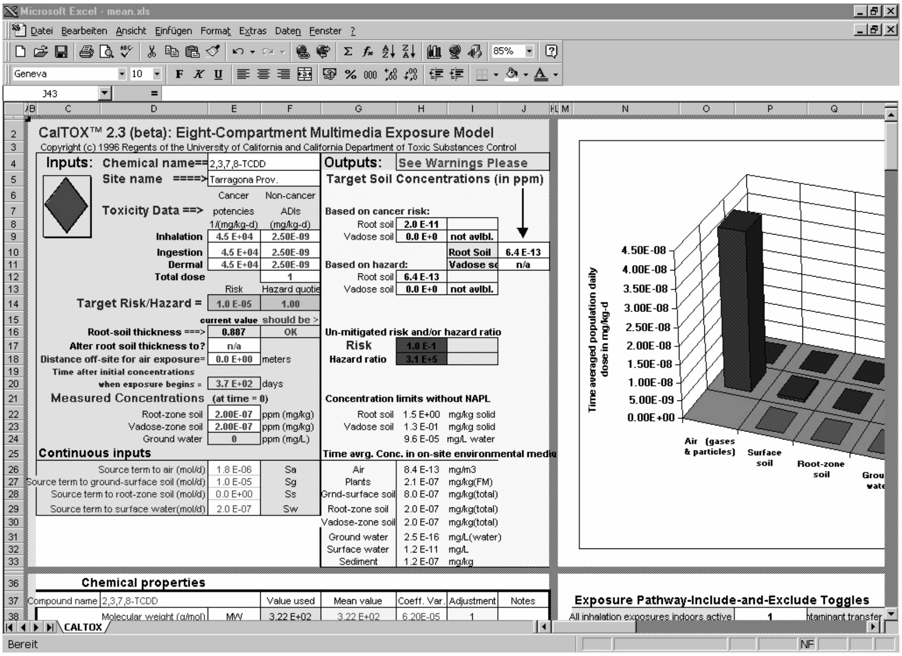

FIGURE 4.12 Screen shot of CalTOX.

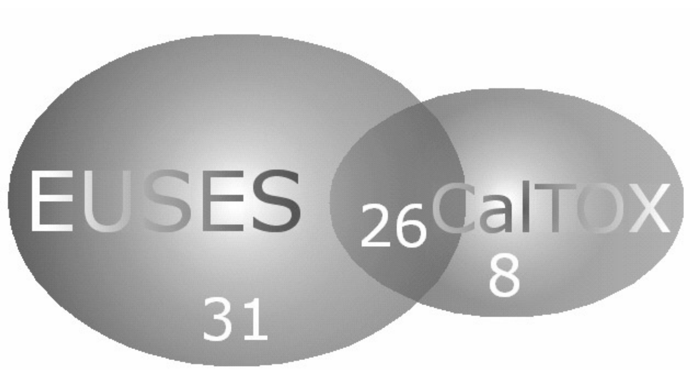

FIGURE 4.13 Amount of shared (intersectional set) and exclusive inputs of EUSES and CalTOX used in this study.

and time flexibility, as well as many estimation routines for missing data. The latter also make it very complex and opaque. Furthermore, any form of data exchange with other applications is very difficult, if not impossible, including uncertainty and sensibility analysis. With EUSES, not as many substances can be assessed as with CalTOX.

To make the choice of model easier, the user needs to know the desired results, the substances to be evaluated, and to which situation to apply the model. As the following comparison of the two models shows, each gives different results and fits

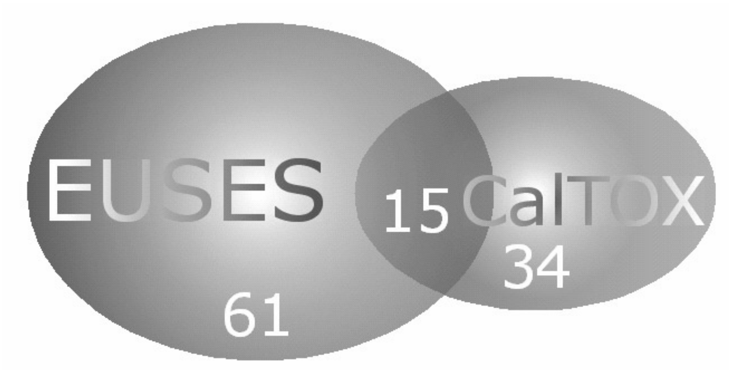

FIGURE 4.14 Amount of shared (intersectional set) and exclusive outputs calculated by EUSES and CalTOX.

well in different situations. Based on these two points, the choice of which model to use would be determined.

Advantages of CalTOX:

- Easy export/import of data/results for visualization or further processing
- Possibility of DDE to other applications
- Direct applicability of sensibility/uncertainty analysis
- Relatively well-structured input/output sections

Advantages of EUSES:

- Assessment for the environment and humans
- Implemented step-by-step-input of necessary data
- Control of range of data introduced by the user
- Possibility of introducing the data in different units
- Great amount of different output data
- Estimation routines for a lot of the necessary data (emissions, effects, etc.)
- Great amount of European default data
- Wide range of time and spatial scales
- Implemented life-cycle steps for individual assessments

Disadvantages of CalTOX

- Only risk assessment for humans
- No control of the range of the values introduced by the user
- Fixed units
- Default data are valid for California
- Limited applicability for short term, exposure and small areas/sites
- Limited output of risk values (only non/cancer risk for humans)

Disadvantages of EUSES

- Rather complex and difficult to understand
- Data exchange/export is difficult and does not work automatically
- Sensibility/uncertainty analysis are not applicable
- Graphical visualization of the results is not implemented

4.11 CASE STUDY: APPLICATIONAL ERA TO MSWI IN TARRAGONA, SPAIN

Consider the example of estimating risk by lifetime of a person living in the surroundings of the municipal solid waste incinerator (MSWI) under study. In this example, the methodology for estimating the distribution of daily PCDD/Fs intake for the population living near a MSWI is presented. First, risk assessment requires identification of the pathways through which people will be exposed to the potential chemicals of concern, in this case PCDD/Fs. The quantitative estimation of health noncancer and cancer risks due to a PCDD/F exposure was considered to be a combination of six pathways: ingestion of soil, ingestion of vegetation from the area, inhalation of resuspended soil particles, inhalation of air, dermal adsorption and through diet. These pathways were classified depending on whether they were due to direct deposition of the MSWI emissions or to an indirect exposure. Ingestion of soil, ingestion of edible vegetables from the area, dermal absorption, inhalation of resuspended particles, and air inhalation were considered pathways of direct exposure, and exposure through the diet a pathway of indirect exposure.

The concentrations of PCDD/Fs were determined in soil and vegetation samples collected near the MSWI in Tarragona, Spain (Schuhmacher et al., 1998a, b). Food samples, which were randomly obtained from local markets and supermarkets, were also analyzed for PCDD/Fs (Domingo et al., 1999).

We may begin by calculating the total amount of contaminant ingested via the six different exposure pathways.

1. **Ingestion of contaminated soil (Ings).** Humans ingest small amounts of soil indirectly (hand-to-mouth transfer) when they work outdoors or during home gardening. Although outdoor workers can be exposed during the whole year, most people have contact with soil only when they work in their gardens. Exposure to soil is a function of the pollutant concentrations in soil and the individual consumption rate. Average daily dose resulting from ingestion of contaminated soil is (Table 4.5):

$$\text{Ings} = \text{Sc}\cdot\text{SIR}\cdot\text{AFIs}$$

where Sc: PCDD/F concentrations in soil (ng/kg), SIR: soil ingestion rate (mg soil/day), and AFIS: fraction absorption ingestion of soil (unitless). SIR varies depending upon the age of the individual, amount of outdoor/indoor activity, frequency of hand-to-mouth contact and seasonal climate (U.S. Environmental Protection Agency 1990).

TABLE 4.5
Parameter Value for Direct Exposure for the Population Living in Area Surrounding an MSWI

Parameter	Symbol	Units	Value	References
Soil ingestion rate	SIR	mg/day	3.44	LaGrega et al. (1994)
Fraction absorption ingestion of soil	AFIS	unitless	40	Nessel et al. (1991)
Vegetable ingestion rate	VIR	g/day	99	Arija et al. (1996)
Fraction absorption ingestion of vegetables	AFIV	unitless	60	Nessel et al. (1991)
Fraction vegetables from the area	IA	unitless	5	Personal communication
Resuspended particles from the soil	RES	unitless	50	Hawley (1985)
Ventilation rate	Vr	m^3/day	11	Shin et al. (1998)
Fraction retained in the lung	RET	unitless	60	Nessel et al. (1991)
Particle concentration	Pa	$\mu g/m^3$	133	Personal communication
Fraction absorption inhalation	AFIn	unitless	100	Nessel et al. (1991)
Contact time soil–skin	CT	h/day	1.50	USEPA (1990)
Exposed skin surface area	SA	cm^2	1980	USEPA (1990)
Dermal absorption factor	Add	unitless	0.003	Katsumata and Kastenberg (1997)
Soil to skin adherence factor	AF	mg/cm^2	1.00	USEPA (1990)
PCDD/F concentrations in soil from the area	Sc	ng/kg	1.17	Schuhmacher et al. (1998)
PCDD/F concentrations in vegetables from the area	Vc	ng/kg	0.197	Schuhmacher et al. (1998)
PCDD/F concentrations in air	Ac	pg/m^3	0.07	Personal communication

2. **Ingestion of vegetables from the area (Ingv).** In general, a high fraction of consumed vegetables is not grown in the region where a person lives but is imported from other regions. Therefore, only a fraction of 5% of total vegetables ingested (equivalent to locally grown) was considered. The average daily intake of TCDD equivalents was estimated by multiplying the PCDD/F concentrations in vegetables by the daily amount of intake, by the fraction of vegetables from the area, and by the absorption factor (Table 4.5):

$$\text{Ingv} = \text{Vc}\cdot\text{VIR}\cdot\text{AFIV}\cdot\text{IA}$$

where Vc: PCDD/F concentrations in vegetables (ng/kg), VIR: vegetables ingestion rate (mg vegetables/day), AFIV: fraction absorption ingestion of vegetables (unitless), and IA: fraction vegetables from the area (unitless).

3. **Inhalation of resuspended particles of soil (Inhp).** Contaminants that deposit to the ground become aggregated with soil particles. Natural and mechanical disturbances, e.g., wind, construction and demolition of buildings, can lead to a resuspension of particles from soil into the atmosphere. Along with this resuspended dust, the adhering pollutants reach the atmosphere and are subsequently inhaled by people who live or work in the region. Inhalation exposure from emissions was calculated by assuming that individuals were exposed to contaminated air and that indoor air exposure was equal to outdoor exposure (Nessel et al., 1991) (Table 4.5):

$$\text{Inhp} = \text{Sc}\cdot\text{RES}\cdot\text{Vr}\cdot\text{RET}\cdot\text{Pa}\cdot\text{AFIn}$$

where Sc: PCDD/F concentrations in soil (ng/kg), RES: fraction of resuspended particles from soil (unitless), Vr: ventilation rate (m^3/day), RET: fraction retained in lungs (unitless), Pa: particle concentration ($\mu g/m^3$), and AFIn: fraction absorption inhalation (unitless).

4. **Air inhalation (Inh; Table 4.5).** The inhaled quantity of an airborne pollutant depends mainly on the atmospheric concentration and the individual inhalation rate. Vapor and particle-bound pollutants are taken up likewise. The total daily intake was related to the body weight in order to obtain a daily inhalation dose.

$$\text{Inh} = \text{Ac}\cdot\text{Vr}\cdot\text{AFIn}$$

where Ac: PCDD/F concentrations in air (pg/m^3), Vr: ventilation rate (m^3/day), AFIn: fraction absorption inhalation (unitless).

5. **Dermal absorption exposure (Ads).** Dermal absorption was assumed to occur only in case of direct skin contact to contaminated soil. People come in contact with soil when they work outdoors or during home gardening. Although outdoors workers are exposed during the whole year to contaminated soil, most people are affected only a limited period of the year because bad weather impedes the stay in their garden or they tend to other weekend activities. Daily dermal exposure was estimated by the following model (Table 4.5):

$$\text{Ads} = \text{Sc}\cdot\text{SA}\cdot\text{CT}\cdot\text{AF}\cdot\text{Add}$$

where Sc: PCDD/F concentrations in soil (ng/kg), SA: exposed skin surface area (cm^2), CT: contact time soil to skin (hr/day), AF: soil to skin adherence factor (mg/cm^2), Add: dermal absorption factor (unitless).

6. **Ingestion through diet.** Human daily PCDD/Fs intake from diet is calculated by multiplying the concentration of PCDD/Fs in each food group by the amount of food group consumed daily and by the absorption fraction (Nessel et al., 1991). Food groups were the following: meat, eggs, fish, milk, dairy products, oil, cereals, pulses, vegetables and fruits (Table 4.6):

TABLE 4.6
Parameter Value for Ingestion through Diet Exposure for the General Population of Tarragona, Spain

Parameter	Symbol	Units	Value	References
Intake of meat	Inme	g/day	180	Arija et al. (1996)
Intake of eggs	Ineg	g/day	30	Arija et al. (1996)
Intake of fish	Infi	g/day	53	Arija et al. (1996)
Intake of milk	Inmi	g/day	226	Arija et al. (1996)
Intake of dairy products	Indm	g/day	50	Arija et al. (1996)
Intake of oil	Inol	g/day	43	Arija et al. (1996)
Intake of cereals	Ince	g/day	210	Arija et al. (1996)
Intake of pulses	Inpu	g/day	14	Arija et al. (1996)
Intake of vegetables	Inve	g/day	99	Arija et al. (1996)
Intake of fruits	Infr	g/day	236	Arija et al. (1996)
PCDD/F concentrations in meat	Mec	ng/kg	0.12	Domingo et al. (1999)
PCDD/F concentrations in eggs	Egc	ng/kg	0.13	Domingo et al. (1999)
PCDD/F concentrations in fish	Fic	ng/kg	0.42	Domingo et al. (1999)
PCDD/F concentrations in milk	Mic	ng/kg	0.12	Domingo et al. (1999)
PCDD/F concentrations in dairy products	Dmc	ng/kg	0.04	Domingo et al. (1999)
PCDD/F concentrations in oils	Olc	ng/kg	0.56	Domingo et al. (1999)
PCDD/F concentrations in cereals	Cec	ng/kg	0.25	Domingo et al. (1999)
PCDD/F concentrations in pulses	Puc	ng/kg	0.19	Domingo et al. (1999)
PCDD/F concentrations in vegetables	Vec	ng/kg	0.14	Domingo et al. (1999)
PCDD/F concentrations in fruit	Frc	ng/kg	0.09	Domingo et al. (1999)
Fraction absorption ingestion of food	AFIF	unitless	60	Nessel et al. (1991)

$$\mathrm{Ingd} = \Sigma \mathrm{Fc} \cdot \mathrm{FIR} \cdot \mathrm{AFIF}$$

where Fc: PCDD/F concentration in a food group (ng/kg), FIR: food ingestion rate (mg food/day), and AFIF: fraction absorption ingestion food (unitless).

The addition of the PCDD/F amount through the different pathways gives the total dose. See Table 4.7.

Table 4.7 summarizes the value of the direct exposure to PCDD/Fs by the population living in the proximities of the MSWI of Tarragona, Spain. Inhalation of air from the area and ingestion of vegetables are the pathways that contribute more to the direct exposure. The other pathways of exposure — dermal absorption, soil ingestion and inhalation of resuspended particles — are, in fact, minimal. With regard to PCDD/F exposure through diet, Table 4.8 shows the daily intake of PCDD/F from different food groups and from total diet. The intake of cereals is the exposure daily intake that contributes more to the total exposure diet. Milk, fish, vegetables,

TABLE 4.7
Different Types of Direct PCDD/Fs Exposure[a] of Population Living in Proximity of the MSWI

Type of exposure	Mean
Soil ingestion	1.61×10^{-6}
Vegetable ingestion	5.85×10^{-4}
Inhalation of resuspended particles	5.14×10^{-7}
Inhalation of air	7.70×10^{-4}
Dermal absorption	1.04×10^{-9}
Total direct exposure	1.36×10^{-3}

[a]ng I-TEQ/day.

TABLE 4.8
Daily Intake of PCDD/Fs[a] from the Diet

Food group	Mean
Meat	12.96
Eggs	2.34
Fish	13.36
Milk	16.27
Dairy products	1.20
Fat	14.45
Cereals	31.50
Pulses	1.60
Vegetables	8.32
Fruits	12.74
Total diet	114.73

[a]pg I-TEQ/day.

fruits, meat and fat intake have a considerable contribution to the total exposure diet, and the other food groups have insignificant contribution. The results corresponding to the diet exposure are based on a Mediterranean diet. See Table 4.7 and Table 4.8.

Once the human daily intake of PCDD/Fs through the different exposure pathways has been calculated, the noncarcinogenic and carcinogenic risks can be calculated. To determine if the contaminant poses a noncancer risk to human health, daily intake is compared with the reference dose (RfD) for chronic exposure. The carcinogenic risk is calculated by multiplying the estimated dose by the carcinogenic potency factor for PCDD/Fs (Table 4.9).The predicted carcinogenic risk is an upper-bound estimate of the potential risk associated with exposure. Table 4.10 shows the noncarcinogenic and carcinogenic risks from direct, indirect (= diet) and total exposure.

TABLE 4.9
Parameter Values for PCDD/Fs

Parameter	Symbol	Units	Value	References
Body weight	BW	kg	67.52	—
Noncarcinogenic potency factor	NCP	pg/kg/day	1–4	Van Leeuwen et al. (2000)
Carcinogenic potency factor	CP	$(mg/kg/day)^{-1}$	34,000–56,000	Katsumata and Kastenberg (1997)

TABLE 4.10
Noncarcinogenic and Cancer Risks by Direct Diet, Direct Risk and Total Exposure

Noncarcinogenic		Carcinogenic	
Direct risk	8.04×10^{-3}	Direct risk	8.64×10^{-7}
Diet risk	6.79×10^{-1}	Diet risk	7.31×10^{-5}
Total exposure	6.87×10^{-1}	Total exposure	7.39×10^{-5}

The total cancer risk is 7.39×10^{-5}, which means that a person living in the surroundings of the MSWI has chance of less than one in a million of developing cancer during his or her lifetime. The total noncancer risk is 6.87×10^{-1}, which means that the population exposure does not exceed the threshold value. Neither the emissions from the MSWI nor the indirect exposure (diet) to PCDD/Fs in the Tarragona area would mean an additional noncarcinogenic risk for health to general population living in the area.

Another point is that, in both cases (noncarcinogenic and carcinogenic risk), the total risk is due mainly to diet. Consequently, under the present conditions, Tarragona's MSWI would not cause a substantial additional exposure to PCDD/Fs in the area under potential influence of the plant. See Table 4.9.

4.12 QUESTIONS AND EXERCISES

1. Indicate possible and ideal target/indicator species for an ERA in the following cases: a) dioxins emission of a MSWI; b) acid rain; c) indoor radiation. Explain their advantages and disadvantages.
2. Which aspects do you consider the main difficulties in the selection of target species of an ecosystem to overall risk assessment? How do you consider these problems can be noticeably improved?
3. Describe the main concept of risk assessment and sum up the main fields and boundaries of application as well as the main tools utilized in it.

4. Calculate an acceptable concentration (8-h time-weighted average) to prevent cancer effects in workers where there is working lifetime exposure to an airborne threshold toxicant. The pollutant has a potency factor of 0.002 $(mg/kg/day)^{-1}$, the absorption factor is estimated at 80%, and the exposure time is 5 days per week, 50 weeks per year over a 30-year period. The worker is assumed to breathe for 3.5 h per workday at the rate of 1.5 m^3/h and 3.5 h per workday at a moderate breathing rate of 1 m^3/h.
5. The reference dose (RfD) for arsenic is set at 3.0×10^{-4} mg/kg/day and the carcinogenic oral scope is 1.75 $(mg/kg/day)^{-1}$. Discuss which would be more stringent: oral concentration standard based on a carcinogenic risk or on RfD.
6. A 70-kg person is exposed to 1.33 ng/m^3 of cadmium in the air and also consumes an average of 53 g fish per day twice a week taken from a contaminated river with a cadmium concentration of 0.2 μg/g. The reference concentration is 2.0×10^{-2} $\mu g/m^3$, reference dose is 1.0×10^{-3} mg/kg/day and the inhalation unit factor $(mg/m^3)^{-1}$. Calculate his or her lifetime cancer and noncancer risks.
7. Suppose a 70-kg person consumes an average of 5.6 g fish per day with a concentration of DDT (oral potency factor = 0.34 $(mg/kg\text{-}day)^{-1}$) equal to 50 ppb (0.05 mg/L). Calculate the maximum lifetime cancer risk from this source.
8. Suppose that a 70-kg individual eats 7 g fish per day taken from a river contaminated by methylene chloride and that the bioconcentration factor (BCF) is 2 L of water per kilogram of fish. What concentration of methylene chloride (mg/L) in the river water would produce a lifetime risk of 6×10^{-7} to this individual? (Bioconcentration factor is a measure for the characterization of the accumulation of a chemical in an organism. It is defined as the concentration of a chemical in an organism — plants, microorganisms, animals — divided by the concentration in a reference compartment, e.g., food, surrounding water.)
9. The reference dose (RfD) for 1,1-dichloroethylene is set at 0.009 mg/kg/day and its oral potency factor is 0.58 $(mg/kg\text{-}day)^{-1}$. Discuss which would be more stringent: a dichloroethylene oral concentration standard based on a carcinogenic risk of 1×10^{-6} or a standard based on RfD.
10. Consider the issue of indoor air pollution caused by sidestream smoke emitting roughly 0.4 mg/cigarette of 1,3-butadiene. Calculate the average concentration of 1,3-butadiene producing a 1×10^{-6} lifetime cancer risk using standard values of inhalation. The inhalation potency factor of 1,3-butadiene is 6.1×10^{-1} $(mg/kg\text{-}day)^{-1}$.
11. Estimate and discuss what would be better regarding the cancer risk: drinking unchlorinated groundwater with 25 ppb of benzene, which has an oral potency factor of 2×10^{-2} $(mg/kg\text{-}day)^{-1}$, or switching to a surface water supply that, as a result of chlorination, has a chloroform concentration of 50 ppb (which has a potency factor of 6.1×10^{-3} $(mg(kg\text{-}day)^{-1})$.

REFERENCES

Arija, V., Salas, J., Fernández-Ballart, J., Cucó, G., and Marti-Hanneberg (1996), Consumo, hábitos alimentarios y estado nutricional de la población de Reus (IX). Evolución del consumo de alimentos de su participación en la ingestion de energia y nutrientes y de su relación con el nivel socioeconomico y cultural entre 1983 y 1993, *Med. Clin.,* 106, 174–179.

Baird, S.J.S., Cohen, J.T., Graham, J.D., Shlyakhter, A.I., and Evans, J.S. (1996), Noncancer risk assessment — a probabilistic alternative to current practice, *Hum. Ecol. Risk Assessment*, 2(1), 79–102.

Barnes, D.G. and Dourson, M.L. (1988), Reference dose (RfD) — descriptions and use in health risk assessments, *Regul. Toxicol. Pharmacol*., 8, 471-486.

Beeline (1998), Beest for Windows. User's Manual. Beeline Software Inc., Asheville, NC.

Beer, T. and Ricci, P.F. (1999), A quantitative risk assessment method based on population and exposure distributions using Australian air quality data, *Environ. Int*., 21, 887–898.

Bonnefous, S. and Despres, A. (1989), Evolution of the European database, IPSN/EURATOM-CEA Association, Fontenay-Aux-Roses, France.

CAPCOA (California Air Pollution Control Officers Association) (1990), Air Toxics "Hot Spots" Program, Risk Assessment Guidelines, Sacramento, CA.

Crettaz, P., Pennington, D., Brand, K., Rhomberg, L., and Jolliet, O. (2002), Assessing human health response in life-cycle assessment using ED10s and DALYs, Part 1: Cancer effects and Part 2: Noncancer effects, *Risk Anal*., 22(5), 931–963.

Derwent, R.G., Dollard, G.J., and Metcalfe, S.E (1988), On the nitrogen budget for the United Kingdom and northwest Europe, *Q. J. R. Meteorol. Soc*., 114, 1127–1152.

Derwent, R.G. and Nodop, K. (1986), Long-range transport and deposition of acidic nitrogen species in northwest Europe, *Nature*, 324:356–358.

Domingo, J.L., Schuhmacher, M., Granero, S., and Llobet, J.M. (1999), PCDDs and PCDFs in food samples from Catalonia, Spain. An assessment of dietary intake, Chemosphere, 38, 3517-3528.

EC (European Commission) (1995), *ExternE — Externalities of Energy*, 6 vol. EUR 16520-162525. DG XII Science, Research and Development, Brussels.

EC (European Commission) (1996), Technical Guidance Document (TGD) 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, Parts I–IV, Office for Official Publications of the European Communities, Luxembourg.

EC (European Commission) (2000), *Externalities of Fuel Cycles — Volume 7: Methodology Update*, Report of the JOULE research project ExternE (Externalities of Energy), EUR 19083. DG XII Science, Research and Development, Brussels.

Fairman, R., Mead, C.D., and Williams, W.P. (1998), *Environmental Risk Assessment — Approaches, Experiences and Information Sources*, European Environment Agency, Copenhagen.

Friedrich, R. and Voss, A. (1993), External costs of electricity generation, *Energy Policy*, 21, 114–122.

Gold, L.S., Manley, N.B., Slone, T.H., Garfinkel, G.B., Rohrbach, L., and Ames, B.N. (1993), The fifth plot of the Carcinogenic Potency Database: results of animal bioassays published in the general literature through 1988, *Environ. Health Perspect.,* 100, 65–135

Hawley, J.K. (1995), Assessment of health risk from exposure to contaminated soil, *Risk Anal.*, 5, 78–85.

Hohmeyer, O. (1992), Renewables and the full costs of energy, *Energy Policy*, 20, 365–375.

IER (Institut für Energiewirtschaft und Rationale Energieanwendung) (1998), EcoSense 2.1 Software, University of Stuttgart, Germany.

IRIS (1996), EPA Integrated Risk Information System, available online at: http://www.epa.gov/ngispgm3/iris/.

Katsumata, P.T. and Kastenberg, W.E. (1997), On the impact of future land use assumptions on risk analysis for superfund sites, *Air Waste Manage. Assoc.,* 47, 881–889.

Krewitt, W., Pennington, D.W., Olsen, S., Crettaz, P., and Jolliet, O. (2002), Indicators for human toxicity in Life Cycle Impact Assessment, in *Towards Best Available Methods for Life Cycle Impact Assessment,* Society of Environmental Toxicology and Chemistry (SETAC). ISBN 1-880661-64-6.

Krewitt, W., Trukenmueller, A., Mayerhofer, P., and Freidrich, R. (1995), EcoSense — an integrated tool for environmental impact analysis, in Kremers, H. and Pillmann, W., Eds., *Space and Time in Environmental Information Systems*, Umwelt-Informatik aktuell, Band 7. Metropolis–Verlag, Marburg, Germany.

LaGrega, M.D., Buckingham, P.L., and Evans, J.C. (1994), *Hazardous Waste Management,* McGraw-Hill, New York.

Mackay, D. and Yeun, A.T.K. (1983), Mass transfer coefficient correlations for volatilization of organic solutes form water, *Environ. Sci. Technol*., 17, 211–217.

Mackay, D. and Leinonen, P.J. (1975), Rate of evaporation of low-solubility contaminants from water bodies, *Environ. Sci. Technol*., 9, 1178–1180.

Masters, G. (1991), *Introduction to Environmental Engineering and Science*, Prentice Hall, Englewood Cliffs, NJ.

Mayerhofer, P., Krewitt, W., and Friedrich, R. (1997), Extension of the accounting framework, final report, ExternE Core Project, IER, Universität Stuttgart, Germany.

Morgan, M.D. and Henrion, M. (1990), *Uncertainty: A guide to Dealing with Uncertainty in Quantitative Risk and Policy Analysis*, Cambridge: Cambridge University Press.

Nessel, C.S., Butler, J.P., Post, G.B., Held, J.L., Gochfeld, M., and Gallo, M. (1991), Evaluation of the relative contribution of exposure routes in a health risk assessment of dioxin emissions from a municipal waste incinerator, *J. Exposure Anal. Env. Epidemiol.,* 1(3), 283–307.

Olsen, S.I., Krewitt, W., Cretaz, P., Pennington, D., and Effting, S. (2000), Indicators for human toxicity in life-cycle impact assessment, position paper for SETAC-Europe WIA-2 Task Group on Human Toxicity, intermediate draft.

Pilkington, A., Hurley, F., and Donnan, P. (1997), Health effects in the ExternE transport — assessment and exposure–response functions, technical report, Institute of Occupational Medicine, Edinburgh.

Rabl, A., Spadaro, J.V., and McGavran, P.D. (1998), Health risks of air pollution from incinerators — a perspective, *Waste Manage. Res*., 16(4), 368–388.

Royal Society (1992), *Risk Analysis, Perception and Management*, The Royal Society, London.

Sandnes, H. and Styve, H. (1992), Calculated Budgets for Airborne Acidifying Components in Europe 1985, 1987, 1988, 1989, 1990 and 1991. MSC-W Report 1/92. Technical Report no. 97, EMEP, The Norwegian Meteorological Institute, Oslo.

Schuhmacher, M., Domingo, J.L., Llobet, J.M., Müller, L., Sünderhauf, W., and Jager, J. (1998b), Baseline levels of PCDD/F in vegetation samples collected in the vicinity of a new hazardous waste incinerator in Catalonia, Spain, *Chemosphere*, 36(12), 2581–2591.

Schuhmacher, M., Granero, S., Xifró, A., Domingo, J.L., Rivera, J., and Eljarrat, E. (1998a), Levels of PCDD/Fs in soil samples in the vicinity of a municipal solid waste incinerator, *Chemosphere*, 37(9-12), 2127–2137.

Shin, D., Lee, J., Yang, J., and Yu, Y. (1998), Estimation of air emission for dioxin using a mathematical model in two large cities of Korea, *Organohalogen Compd.*, 36, 449–453.

Thibodeaux, L.J. and Hwang, S.T. (1982), Landfarming of petroleum wastes — modeling the air emission problem, *Environ. Prog.*, 1, 42–46.

U.S. EPA (1989), Estimating air emissions from petroleum UST cleanups, Office of Underground Storage Tanks, Washington, D.C.

U.S. EPA (1990), Methodology for assessing health risks associated with indirected exposure to combustor emissions, Office of Health and Environmental Assessment, Cincinnati, OH, EPA/600/6-90/003.

U.S. EPA (1995), Guidance for risk characterization, U.S. Environmental Protection Agency, Science Policy Council, Washington, D.C.

U.S. EPA (1992), User's Guide for the Industrial Source Complex (ISC2) — Dispersion Models Volumes I-III. EPA-450/4-92-008a, EPA-450/4-92-008b, EPA-450/4-92-008c. Triangle Park, NC.

U.S. EPA (1995), User's Guide for the Industrial Source Complex (ISC3) — Dispersion Models Volumes I-II. EPA-454/B-95-003a, EPA-454/B-95-003b. Triangle Park, NC.

U.S. EPA (2002), A Brief Description of the AERMOD. Available online at: http://www.epa.gov/scram001/tt26.htm#aermod

UN-ECE United Nations Economic Commission for Europe (1991), Protocol to the 1979 convention on long-range transboundary air pollution concerning the control of emissions of volatile organic compounds or their transboundary fluxes, report ECE/EB AIR/30, UN-ECE, Geneva, Switzerland.

van Leeuwen, F.X.R., and Younes, M. (1998), WHO revises the Tolerable Daily Intake (TDI) for dioxins. *Organohalogen Compd.,* 38, 295–298.

Zemba, S.G., Green, L.C., Crouch, E.A.C., and Lester, R.R. (1996), Quantitative risk assessment of stack emissions from municipal waste combustors, *J. Hazard. Mat.*, 47, 229–275.

5 Uncertainty Assessment by Monte Carlo Simulation*

With the contribution of Luiz Alexandre Kulay and Yolanda Pla

5.1 INTRODUCTION

The high uncertainty present in the implementation of life-cycle and environmental risk assessment studies introduces a crucial limitation when interpreting the environmental impact and damage estimations provided by these methodologies. Within this particular context this chapter presents a strategic procedure to better deal with an uncertainty assessment based on the stochastic model of Monte Carlo (MC) simulation. Initially, we will present an overview about the importance of implementing uncertainty assessments in studies of life-cycle assessment (LCA) and impact pathway analysis (IPA). Then, the basic statistical concepts related to MC simulation are introduced. This section also compares some of the best known commercial software packages that apply MC simulation to uncertainty evaluation. The third section describes the core of the uncertainty assessment strategic procedures. Finally, in the last section, we will use the municipal solid waste incineration process (MSWI) in Tarragona, Spain, as the practical case study of risk assessment for explaining the procedure better.

5.2 TYPES OF UNCERTAINTIES IN ENVIRONMENTAL IMPACT ANALYSIS

The various sources of uncertainty in environmental impact analysis can be systematically classified in accordance with the following categories (Huijbregts, 1998).

5.2.1 PARAMETER UNCERTAINTY

A life-cycle inventory (LCI) analysis and the models that calculate fate, exposure and effect within an impact and risk assessment usually need a large amount of data.

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Uncertainty of these parameters reflects directly on the outcome of any environmental impact method. Empirical inaccuracy (imprecise measurements), unrepresentative data (incomplete or outdated measurements) and lack of data (no measurements) are common sources of parameter uncertainty. Weidema and Wesnæs (1996) describe a comprehensive procedure for estimating combined inaccurate and unrepresentative LCI data qualitatively and quantitatively. Although this procedure may substantially improve the credibility of LCA outcomes, uncertainty analyses are generally complicated by a lack of knowledge of uncertainty distributions and correlations among parameters.

5.2.2 Model Uncertainty

According to conclusions presented by various authors, the predicted values for environmental impact and risk generally respond in a linear manner to the amount of emitted pollutant. Moreover, in life-cycle impact assessment (LCIA) and IPA, thresholds for environmental interventions are disregarded. Additionally, in LCIA the derivation of characterization factors causes model uncertainty because these are calculated with the aid of simplified environmental models without considering spatial and temporal characteristics.

5.2.3 Uncertainty due to Choices

In many cases, performing choices is unavoidable in environmental impact analysis. Considering the step of LCI from LCA, examples of choices leading to uncertainty include the selection of the functional unit (or definition of the allocation procedure for multioutput processes), multiwaste processes, and open loop recycling. Moreover, the socioeconomic evaluation step in LCA and IPA is an area in which choices play a crucial role. Although experts from the social sciences have suggested many different weighting schemes, only a few are operational and no general agreement exists as to which one should be preferred.

5.2.4 Spatial and Temporal Variability

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

5.2.5 Variability among Sources and Objects

In LCA or in other impact assessment methods, variability among sources and objects may influence the outcome of a study. This means, for example, that some variability in LCIs may result from differences in inputs and emissions of comparable processes within a product system (due to the use of different technologies in factories producing the same material). Furthermore, variability among objects exists in weighting environmental problems during impact assessment due to variability in human preferences. For instance, when a method such as the willingness-to-pay (WTP) method is used to determine the external environmental cost due to a specific damage, differences related to individual preferences cause inherent variation in the final result.

5.3 WAYS TO DEAL WITH DIFFERENT TYPES OF UNCERTAINTY

Huijbregts et al. (2000) have offered solutions on how to deal with the issues of uncertainty previously discussed. The tools available to address different types of uncertainty and variability in LCAs include probabilistic simulation, correlation and regression analysis, additional measurements, scenario modeling, standardization, expert judgment or peer review, and nonlinear modeling. Scenario modeling (Pesonen et al., 2000) should be especially useful in cases in which uncertainty about choices and temporal variability is present.

When a model suffers from large uncertainties, the results of a parameter uncertainty analysis may be misleading. In most cases, the consequence of decreasing model uncertainty will be the implementation of more parameters in the calculation, thereby increasing the importance of operationalizing parameter uncertainty in the model. In the following two sections, we present an overview of previous efforts to assess uncertainties in LCA.

5.3.1 Experiences to Assess Uncertainty in Life-Cycle Assessment

So far the influence of data quality on final results of LCA studies has rarely been analyzed. In spite of the lack of published case studies, several approaches to carry out this kind of evaluation have been proposed during recent years. Nevertheless, from a general point of view, the existing methods can be classified in qualitative and quantitative assessments.

Qualitative assessment means describing the data used by characterizing its quality. Weidema and Wesnaes (1996) and Weidema (1998) proposed using data quality indicators depending on categories like reliability, completeness, temporal correlation, etc. In turn, Finnveden and Lindfors (1998) suggested ranges for various inventory parameters as rules of thumb. Quantitative assessment means to quantify all inherent uncertainties and variations in an LCA. In order to perform this task, many different analytical procedures have been applied. For uncertainty analysis of LCI**,** Hanssen and Asbjornsen (1996) used statistical analysis, Ros (1998) proved the fuzzy logic, and Maurice et al. (2000) as well as Meier (1997) decided in favor of the stochastic methods. Regarding uncertainty assessment within the impact assessment stage of LCA, Meier (1997), Hofstetter (1998) and Huijbregts and Seppälä (2000) have reported results achieved using similar techniques. Even when it is strongly effective, quantitative assessment is continuously confronted with the problem that it is hardly possible to analyze all types of uncertainties.

5.3.2 Former Uncertainty Assessment in Impact Pathway Analysis

The IPA is a quite complex approach and hence risks lack of reliability in the final results. In the same way as with other environmental analysis methods, uncertainty is the key problem that makes it difficult to convince decision-makers based on the outcomes of a study.

One of the most interesting experiences is that reported by Rabl and Spadaro (1999), in which they evaluated the uncertainty and variability of damages and costs of air pollution by means of analytical statistical methods. In this case, the authors observed that the equation for the total damage is largely multiplicative, even though it involves a sum over receptors at different sites. This conclusion comes from the principle of conservation of matter, which implies that overprediction of the dispersion model at one site is compensated for 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.

In the same reference, the authors discuss the typical error distributions related to the factors in the equation for the total damage, in particular those related to two key parameters: the deposition velocity of atmospheric dispersion models and the value of statistical life; according to Rabl and Spadaro (1999), these are close to log-normal. They conclude that a log-normal 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, in which the geometric standard deviation is in the range of 3 to 5. Results and conclusions such as those presented by Rabl and Spadaro illustrate the necessity of dealing with uncertainty assessment in IPA in spite of its high level of complexity.

5.4 INTRODUCTION TO MONTE CARLO SIMULATION

Based on the previously mentioned experiences regarding uncertainty analysis in LCA and ERA studies, especially to IPA, it seems that the use of a stochastic model helps to characterize the uncertainties better, rather than a pure analytical mathematical approach. This can be justified because the relevant parameters follow a different frequency distribution. In this case, one of the most widespread stochastic model uncertainty analyses is the Monte Carlo (MC) simulation. In a wide approach to perform an MC simulation, the parameters under evaluation must be specified as uncertainty distributions. The method makes all the parameters vary at random because 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. According to LaGrega et al. (1994), MC simulation can be considered the most effective quantification method for uncertainties and variability among the environmental system analysis tools available.

The term simulation can be understood as an 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 would only reveal a single outcome, generally the most likely or average scenario. Spreadsheet uncertainty analysis uses a spreadsheet model and simulation to analyze the effect of varying inputs or outputs of the modeled system automatically. The random behavior of how MC simulation selects variable values to simulate a model is similar to that employed by games of chance. When a player rolls a die, he or she knows that a 1, 2, 3, 4, 5, or 6 will come up, but does not know which will occur in any particular roll. It is the same with the variables that have a known range of values but an uncertain value for any particular time or event (Decisioneering 1996).

5.4.1 Use of Software Packages to Propagate Uncertainty by Performing Monte Carlo Simulation

In order to simplify the task of determining the uncertainty of a parameter by MC simulation, various commercial software packages are available. Among them, Crystal Ball®,* @Risk®,** Analytica,*** Stella II®,**** PRISM®***** and Susa-PC®****** can be highlighted. Table 5.1 summarizes the main information about each of these software packages.

According to Metzger et al. (1998), @Risk was originally designed for business applications, but it has found wide use in human health and ecological risk assessment. This package includes uncertainty in estimates to generate results that show all possible outcomes. Because of its standardized spreadsheet backbone, @Risk is easy to use without a need for extensive statistical knowledge, modeling capability or programming ability.

Analytica and Stella II are two stand-alone programs designed for a wide variety of applications. Analytica is a model-building program that attempts to simplify sophisticated systems with the use of multilevel influence diagrams. Stella II is a multilevel hierarchical environment for constructing and interacting with models. These two programs are designed to simplify complex problems and therefore require additional effort in learning the software and building the models. However, because these models are displayed in diagrams, their interpretation for other users tends to be easier than for any other software package.

PRISM and Susa-PC are Fortran-based codes, designed specifically for use in risk analysis. PRISM is a simple free-access program that builds distributions for input into any model and then analyzes the output of that model. Susa-PC is a more

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**** Registered trademark of High Performance Systems, Inc., Lebanon, NH.

***** Registered trademark of SENES Oak Ridge, Inc., Oak Ridge, TN.

****** Registered trademark of Gesellschaft für Anlagenund Reaktorsicherheit (GRS) mbH, Köln, Deutschland.

TABLE 5.1
Uncertainty Analysis Software Commercial Packages

Software package	Version	Producer	Year	URL address
Crystal Ball	4.0	Decisioneering, Inc.	1996	www.decisioneering.com
@Risk	3.1	Palisade Corporation	1996	www.palisade.com
Analytica	1.0 Beta 1	Lumina Decision Systems, Inc.	1996	www.decisioneering.com
Stella II	3.0.6	High Performance System, Inc.	1994	www.hps-inc.com
PRISM	November 1992	Gardner, R.H. et al.	1992	www.senes.com
Susa-PC	1.0	Hofner, E. et al.; GRS–Garching	1992	—

Source: Metzger, J.N. et al., *Hum. Ecol. Risk Assess*., 4(2), 263–90, 1998. With permission.

involved package that offers advanced statistical analysis specific to risk analysis. It is Excel based and uses macros for everything except the actual results from the model. Because of their relatively wide focus, PRISM and Susa-PC are more difficult to use than spreadsheets and require some knowledge of Fortran and statistics.

Finally, the software Crystal Ball Version 4.0 from Decisioneering (1996) is a simulation program that helps analyze the uncertainties associated with Microsoft Excel spreadsheet models by MC simulation. Crystal Ball adds probability to the best, worst, and most likely case versions of the same model — which only predict the range of outcomes — and automates the what-if process. Crystal Ball is an add-on to Excel; the user does not need to leave Excel to undertake such forecasting. The program works with models existing already, so the calculations do not need to be recreated. As a fully integrated Excel add-on program with its own toolbar and menus, Crystal Ball picks up where spreadsheets end by allowing the user to perform the MC analysis. The user must define a range for each uncertain value in the spreadsheet and Crystal Ball uses this information to perform thousands of simulations. Another function of this software is the sensitivity analysis. Sensitivity charts show how much influence each assumption has on the results, allowing the user to focus further analytical effort on the most important factors. The results are dynamically summarized in forecast charts that show all the outcomes and their likelihood. 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 5.1).

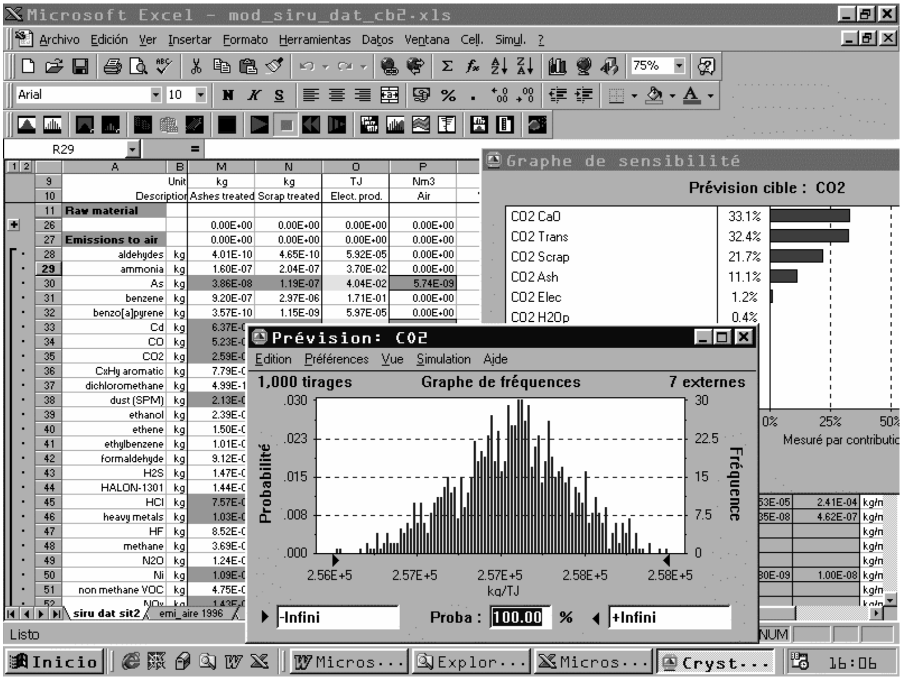

FIGURE 5.1 Crystal Ball screenshot (activated cells of an Excel spreadsheet, the result of a sensitivity analysis and the final provision).

5.4.2 Framework for Uncertainty Assessment by Monte Carlo Simulation in Environmental Impact Analysis

Based on the information about previous studies on uncertainty evaluation in environmental impact analysis methods and the knowledge of the MC simulation technique, the following general strategy for the assessment of uncertainties in impact assessment studies may be established:

- Classification of data (extensively available, based on little information and data that can be ignored)
- Identification of probability distribution for considered data
- Monte Carlo simulation
- Sensitivity analysis
- Analysis and discussion of results

As mentioned briefly earlier, by means of a sensitivity analysis it is possible to show which parameters are most relevant for the final result. If small modifications of one parameter characterized by a probability distribution strongly influence 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 the variables to be acted upon. Also it would be very useful to know the parameters that might be neglected, especially if it is difficult to get detailed information about them. Sensitivity can be analyzed by an approach that displays sensitivity as a percentage of contribution from each parameter to the variance of the final result. Crystal Ball Version 4.0, the software package selected to perform an example of application for the use of MC simulation, approximates this approach by lifting to square the correlation coefficients of ranks and normalizing them to 100%.

5.5 UNCERTANTY ASSESSMENT IN DIFFERENT ENVIRONMENTAL IMPACT ANALYSIS TOOLS

In this section, methods of uncertainty assessment in different environmental impact analysis tools are presented.

5.5.1 Uncertainty Assessment in Life-Cycle Inventory

Figure 5.2 presents an adaptation of the procedure for uncertainty analysis in LCI as it is reported in the literature (Meier, 1997; Maurice et al., 2000). The first step refers to the compilation of LCI data. If all the parameters that might have repercussions on the final result were considered, an exhaustive study would need to be carried out; however, not all these data are relevant. Hence, only the most relevant factors must be selected and, for some parameters, can be assumed to have fixed values. Once the essential factors have been selected, a characterization 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 expert

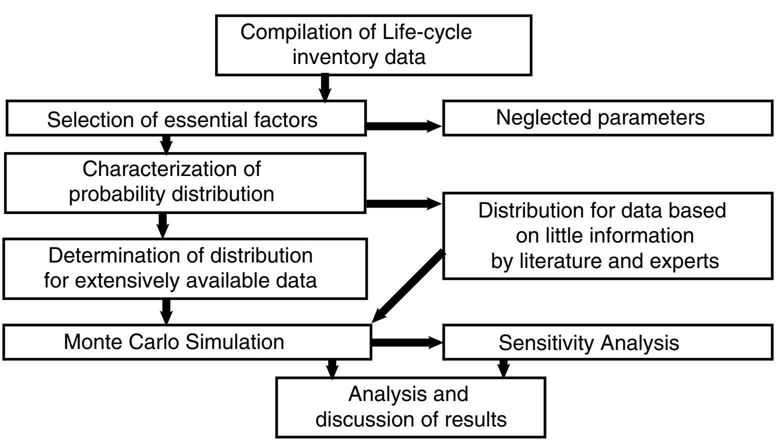

FIGURE 5.2 Procedure for the uncertainty and variability assessment in the life-cycle inventory. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

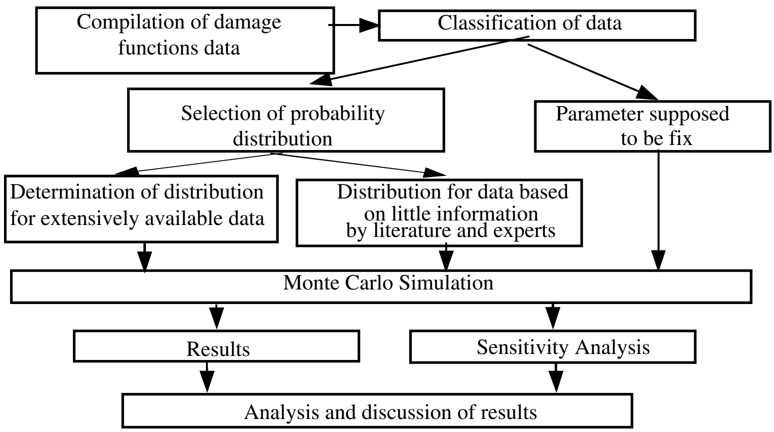

FIGURE 5.3 Framework for the assessment of uncertainty and variability in the impact pathway analysis.

estimations must be considered. All these parameters feed the MC simulation, which gives the results in the form of a probability distribution around a mean value and allows a detailed sensitivity analysis to be carried out.

5.5.2 Uncertainty Assessment in Impact Pathway Analysis

Figure 5.3 presents the framework for uncertainty assessment in the IPA. The first step of the so-called framework is the compilation of damage function data, in which an exhaustive study must be carried out on all the parameters that have a repercussion on the final result. Although the model is processing an enormous quantity of data that are not all relevant, only fundamental facts need really be considered. Thus, a classification must be made among the most significant parameters, for which probability distributions should be defined. In addition, the parameters are supposed to be invariant and are called point estimates. Significant data are further classified into the above-mentioned two groups for advanced evaluation: extensively available data and data based on little information. In the same way as in the uncertainty assessment for LCI, these parameters feed the MC simulation that gives the results in the form of a probability distribution around a mean value, and allows a detailed sensitivity analysis to be carried out. The last step of the framework consists of the analysis and discussion of the achieved results.

5.5.3 Risk Characterization and Uncertainty Analysis

Risk assessment uses a wide array of information sources and models. Even when actual exposure-related measurements exist, assumptions or inferences will still be required. Most likely, data will not be available for all aspects of the exposure assessment and may be of questionable or unknown quality. In these situations, the exposure assessor will depend on a combination of professional judgment, inferences based on analogy with similar chemicals and conditions, estimation techniques and the like. The net result is that the exposure assessment will be based on a number of assumptions with varying degrees of uncertainty (U.S. Environmental Protection Agency, 1992). Decision analysis literature has focused on the importance of explicitly incorporating and quantifying scientific uncertainty in risk assessment (Roseberry and Burmaster, 1991).

Several reasons lead to uncertainties concerning the validity and entirety of the results of a risk assessment. These uncertainties can be regarded in different manners and degrees depending on the methodology applied in the risk assessment process. One source of high uncertainties is the application of models that simulate the behavior of a pollutant in the environment and the uptake into the human body. Computer models that attempt to describe natural processes are always simplifications of a complex reality. They require the exclusion of some variables that in fact influence the results but cannot be regarded because of increased complexity or lack of data. Moreover, many natural processes can only be approximated but not exactly explained with mathematical correlations. Hence, a model is always affected with uncertainties and gives only an imperfect description of the reality. Different models for the same issue consider different uncertainties but disregard also different sources of uncertainty.

On the other hand, because many parameters in a model cannot be treated as fixed-point values, a range of values better represents them. This uncertainty of input parameter can result from real variability, measurement and extrapolation errors as well as the lack of knowledge regarding biological, chemical and physical processes. Uncertainties that are related with lack of knowledge or measurement and extrapolation errors can be reduced or eliminated with additional research and information. However, real parameter variability, e.g., spatial and temporal variation in environmental conditions or life-style differences, occurs always and cannot be eliminated. It leads to a persisting uncertainty of the modeling results.

Risk assessment is subject to uncertainty and variability. Specifically, uncertainty represents a lack of knowledge about factors affecting exposure or risk, whereas variability arises from true heterogeneity across people, places, and time. In other words, uncertainty can lead to inaccurate or biased estimates, whereas variability can affect the precision of the estimates and the degree to which they can be generalized.

Now let us consider a situation that relates to exposure, such as estimating the average daily dose by one exposure route — inhalation of contaminated air. Suppose that it is possible to measure an individual's daily air inhalation consumption (and concentration of the contaminant) exactly, thereby eliminating uncertainty in the measured daily dose. The daily dose still has an inherent day-to-day variability because of changes in the individual's daily air inhalation or concentration of the contaminants in air.

Clearly, it is impractical to measure the individual's dose every day. For this reason, the exposure assessor may estimate the average daily inhalation based on a finite number of measurements, in an attempt to "average out" the day-to-day variability. The individual has a true (but unknown) average daily dose, which has not been estimated based on a sample of measurements. Because the individual's true average is unknown, it is uncertain how close the estimate is to the true value.

Thus, variability across daily doses has been translated into uncertainty in the parameter. Although the individual's true value has no uncertainty, the estimate of the value has some variability (U.S. Environmental Protection Agency, 1992).

The preceding discussion pertains to the air inhalation for one person. Now consider a distribution of air inhalation across individuals in a defined population (e.g., the general U.S. population). In this case, variability refers to the range and distribution of air inhalation across individuals in the population. Otherwise, uncertainty refers to the exposure assessor's state of knowledge about that distribution, or about parameters describing the distribution (e.g., mean, standard deviation, general shape, various percentiles).

As noted by the National Research Council (1994), the realms of variability and uncertainty have fundamentally different ramifications for science and judgment. For example, uncertainty may force decision makers to judge how probable it is that exposures have been overestimated or underestimated for every member of the exposed population, whereas variability forces them to cope with a certainty that different individuals are subject to exposures above and below any of the exposure levels chosen as a reference point (U.S. Environmental Protection Agency, 1992).

To account for the uncertainty in ERA, process probabilistic models are used. These techniques generate distributions that describe the uncertainty associated with the risk estimate (resultant doses). The predicted dose for every 5th percentile to the 95th percentile of the exposed population and the true mean are calculated. Using these models, the assessor is not forced to rely solely on a single exposure parameter or the repeated use of conservative assumptions to identify the plausible dose and risk estimates. Instead the full range of possible values and their likelihood of occurrence are incorporated into the analysis to produce the range and probability of expected exposure levels.

In addition to establishing exposure and risk distributions, probabilistic analysis can also identify variables with the greatest impact on the estimates and illuminated uncertainties associated with exposure variables through sensitivity analysis. This provides some insight into the confidence that resides in exposure and risk estimates and has two important results. First, it identifies the inputs that would benefit most from additional research to reduce uncertainty and improve risk estimates. Second, assuming that a thorough assessment has been conducted, it is possible to phrase the results in more accessible terms, such as, the risk assessment of PCBs in smallmouth bass is based on a large amount of high-quality reliable data, and we have high confidence in the risk estimates derived. The analysis has determined that 90% of the increased cancer risk could be eliminated through a ban on carp and catfish, but there is no appreciable reduction in risk from extending such a ban to bass and trout.

5.6 TYPES OF PROBABILITY DISTRIBUTIONS USED

In the MC simulation, new values of the random variables are selected at least 10,000 times and a new estimate of the final damage is foreseen. The results of the calculations are summarized in a single histogram of damage values; mathematical operations such as multiplication, exponential functions, matrix calculations, etc. can be managed. Among the wide range of statistical distributions (normal, log-normal, uniform, etc.) found in MC simulation, we will refer only to the most common types of probability distributions, which are:

1. Normal distribution (Figure 5.4). Normal distribution is appropriate to describe the uncertainties of large samples that constitute stochastic events and are symmetrically distributed around the mean. The mean and the standard deviation will define the probability density function. The normal distribution is especially appropriate if data uncertainties are given as a percentage of the stadard deviation with respect to the mean, i.e., the coefficient of variation (CV).

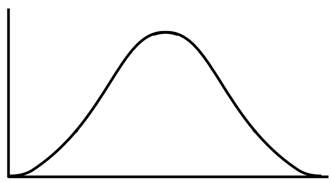

FIGURE 5.4 Normal probability distribution profile.

2. Log-normal probability distribution (Figure 5.5). This type of distribution can be used if large numbers of quantities must be presented, no negative values are possible, and the variance is characterized by a factor rather than a percentage.

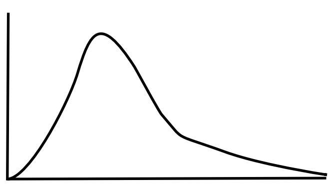

FIGURE 5.5 Log-normal probability distribution profile.

The 50th percentile of a log-normal distribution is related to the mean of its corresponding normal distribution. The log-normal distribution is calculated assuming that the logarithm of the variable has a normal distribution. Many environmental impacts follow the log-normal model. The geometric mean, μg, and the geometric standard deviation, σ_g, of the samples are very practical and correspond to the mean and coefficient of variation for the normal distribution. Moreover, they provide multiplicative confidence intervals such as:

$[\mu_g/\sigma_g, \mu_g.\sigma_g]$ for a confidence interval of 68%

$[\mu/\sigma_g^2, \mu_g.\sigma^2_g]$ for a confidence interval of 95%

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

The Crystal Ball software facilitates the adjustment of sample points in a density function. As an example, Figure 5.6 shows the variation of such sample points for different measurements of cadmium emissions. In the diagram, 17 measurements are classified according to their range of concentration. The most frequent value is the 10 μg/Nm^3 that appears four times. However two samples have more than 35 μg/Nm^3. The presented variation has been adjusted with normal and log-normal distribution. The different curves make evident that the log-normal distributions fit the variation of the measurements much better.

As previously explained, in the case study from Tarragona's MSWI, the variation of the pollutant concentrations in the incinerator emissions is enormous due to the heterogeneity of the incinerated waste. As can be expected, its elementary composition varies strongly at each moment. Often cadmium concentrations are low, but sometimes these increase due to the elevated cadmium amount in the waste. Thus the measured emissions are not constant over time, following a log-normal distribution.

5.7 EXAMPLE: RISK ASSESSMENT TO PCDD/FS IN TARRAGONA, SPAIN, USING THE MONTE CARLO APPROACH

Returning to the case of the MSWI and its application of the risk assessment seen in Chapter 4, now the same case — not as point estimation, but considering its

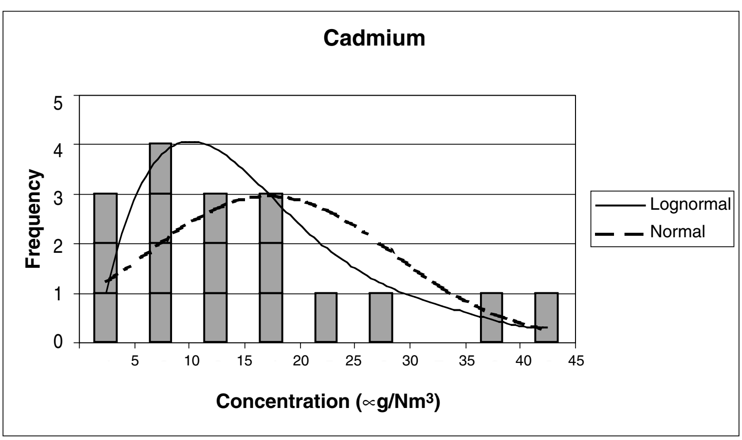

FIGURE 5.6 Log-normal distribution of cadmium emissions by adjustment of sample points in a density function. (Reprinted for *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

distribution — will be shown as an example of uncertainty assessment. Applying the uncertainty analysis to the risk assessment of the population living in its vicinity will be demonstrated. Only the "direct risk" due to air emissions will be considered. To account for variability and uncertainty, the Monte Carlo simulation will be applied to estimate the set of risk estimates. In this example model and data uncertainty will be considered. The sensitivity analysis will show how much each predictor variable contributed to the uncertainty or variability of the predictions.

5.7.1 Determination of Distribution Functions

The determination of which form of distribution function to assign to each parameter depends on site-specific data and judgment based on statistical analysis. The distribution employed in this example is assembled from site-specific data, data existing in the most current literature, and professional judgment; they are considered to be the most up-to-date description of the parameter (Katsumata, 1997). In the vegetation production in the area of study (Tarragona), only the adult population was considered. In this example, only exposition through the air, soil and 10% of the consumed vegetation will be considered as a direct vies of exposition. Because the MSWI is located close to a city, no impact to foods like meat, fish or dairy products will be considered.

Table 5.2 and Table 5.3 show a description of the Monte Carlo parameter distribution for risk assessment evaluation due to direct exposure for people living in the area surrounding the MSWI of Tarragona, Spain. After characterizing the uncertainty and/or variability associated with each parameter, the uncertainty in the risk can be estimated. For the risk assessment presented here, the commercially available software package Crystal Ball (Version 4.0) was used. For analyzing the results, the mean and median values and the percentiles 50 and 90% were extracted and presented. In this example, ranges of exposure rather than single point estimates are developed in order to account for the natural variability among members of a population and for uncertainties in the input variables. Even if it were possible to eliminate the uncertainty associated with the input variables, a probability density function would still be required because of natural variability.

Figure 5.7 shows the distribution of the different variables of direct exposure due to the incinerator emissions for the population living in the area surrounding the plant. The distribution of total direct exposure is also depicted. Figure 5.8 shows the sensitivity analysis for total direct exposure from the different exposure pathways due to MSWI emissions. This figure shows that inhalation of air from the area contributes to 50.4% of the variance and vegetation ingestion to 49.5%. The other pathways — exposure, dermal absorption, soil ingestion, and inhalation of resuspended particles — are irrelevant.

Table 5.4 summarizes the exposure to PCDD/Fs by the population living in the proximity of the MSWI of Tarragona, Spain, and Table 5.5 shows the PCDD/Fs dose (ng I-TEQ/day/kg) for the population living around the incinerator. The tolerable average intake levels of PCDD/Fs recently established by the WHO is between 1 and 4 pg I-TEQ/kg/day for lifetime exposure (Rolaf and Younes, 1998). Therefore, the current total exposures of 2.87×10^{-2} pg I-TEQ/day/kg for the 50th percentile and 5.37×10^{-2} for the 90th percentile (Table 5.5) are within this tolerable intake.

TABLE 5.2
Monte Carlo Parameter Distributions for Direct Exposure for Population Living in Areas Surrounding the MSWI

Parameter	Symbol	Units	Type	Distribution[a]	Reference
Soil ingestion rate	SIR	mg/day	Log-normal	3.44 ± 0.80	LaGrega 1994
Fraction absorption ingestion of soils	AFIS	unitless	Point	40	Nessel 1991
Vegetable ingestion rate	VIR	g/day	Log-normal	99 ± 80	Arija et al. 1996
Fraction absorption ingestion of vegetables	AFIV	unitless	Point	60	Nessel 1991
Fraction vegetables from the area	IA	unitless	Uniform	1–10	Generalitat Catalunya (Statistical Dept.)[b]
Resuspended particles from the soil	RES	unitless	Point	50	Hawley 1985
Ventilation rate	Vr	m^3/day	Log-normal	20–2	Shin 1998
Fraction retained in the lungs	RET	unitless	Uniform	60	Nessel 1991
Particle concentration	Pa	$\mu g/m^3$	Point	133	Generalitat Catalonia (Environmental Dept.)[b]
Fraction absorption inhalation	AFIn	unitless	Point	100	Nessel 1991
Contact time soil–skin	CT	h/day	Uniform	1–2	EPA 1990
Exposed skin surface area	SA	cm^2	Triangular	1980 (910–2940)	EPA 1990
Dermal absorption factor	Add	unitless	Triangular	0.003 (0–0.03)	Katsumata 1997
Soil to skin adherence factor	AF	mg/cm^2	Uniform	0.75–1.25	EPA 1990
PCDD/Fs soil concentration from the area	Sc	ng/Kg	Triangular	1.17 (0.10–3.88)	Schuhmacher et al. 1998a
PCDD/Fs vegetable concentration from the area	Vc	ng/Kg	triangular	0.197 (0.06-0.50)	Schuhmacher et al. 1998b
PCDD/Fs air concentration	Ac	pg/m^3	triangular	0.07 (0.01 - 0.22)	Generalitat Catalonia (Environmental Dept.)[b]

[a]Distribution: means and standard deviations are used for log-normal distributions, the low and high for uniform distributions, and the mean, low and high for triangular distributions.
[b]Personal communication.

TABLE 5.3
Monte Carlo Parameter Distributions for PCDD/Fs

Parameter	Symbol	Units	Type	Distribution[a]	Reference
Body weight	BW	kg	Log-normal	67.52 ± 12.22	Arija et al. 1996[b]
Noncancer potency factor	NCP	pg/kg day	Uniform	1–4	Rolaf and Younes 1998
Cancer potency factor	CP	$(mg/kg\ day)^{-1}$	Uniform	34,000–56,000	Katsumata 1997

[a]Distribution: mean and standard deviation are used for log-normal distributions, the low and high for uniform distributions.
[b]Personal communication.

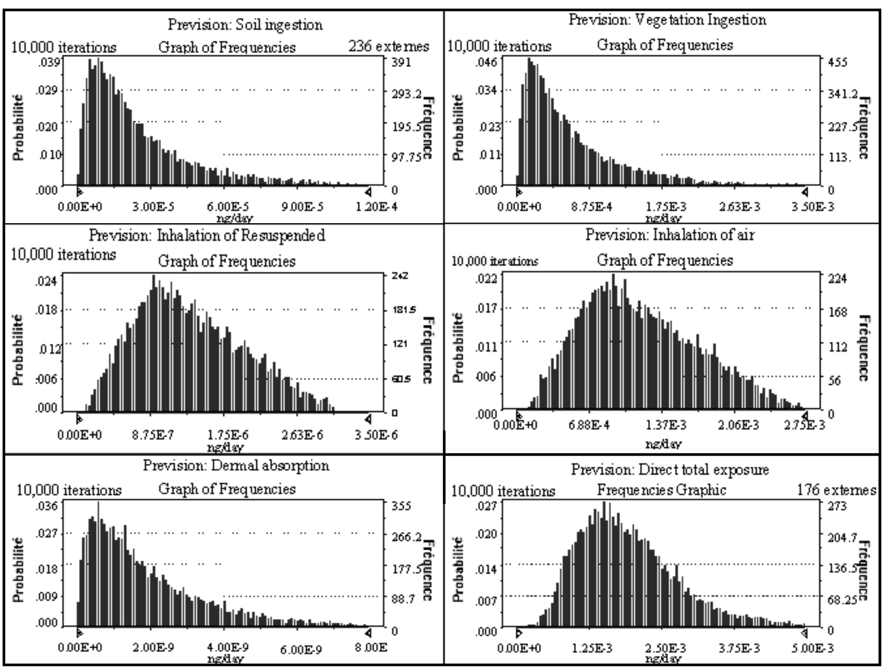

FIGURE 5.7 Distribution of the different variables of direct exposure due to MSWI emissions.

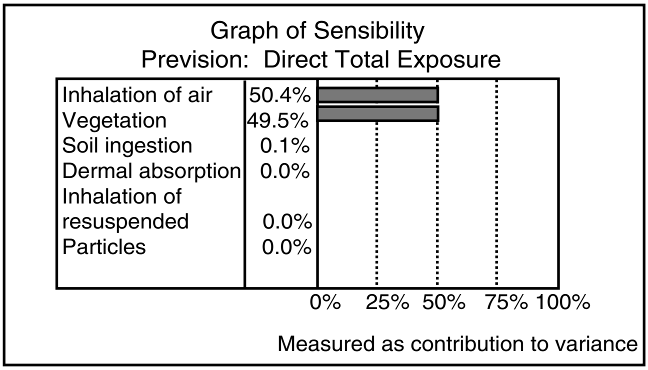

FIGURE 5.8 Distribution of total direct exposure.

TABLE 5.4
PCDD/Fs Exposure[a] by Different Ways of Direct Exposure of the Population Living in Proximity of the MSWI

			Percentiles		
	Mean	**SD**	**10th**	**50th**	**90th**
Soil ingestion	3.01×10^{-5}	3.37×10^{-5}	5.49×10^{-6}	1.98×10^{-5}	6.69×10^{-5}
Vegetable ingestion	7.95×10^{-4}	9.09×10^{-4}	1.36×10^{-4}	5.11×10^{-4}	1.76×10^{-3}
Inhalation of resuspended particles	1.37×10^{-6}	6.53×10^{-7}	5.73×10^{-7}	1.28×10^{-6}	2.31×10^{-6}
Inhalation of air	1.19×10^{-3}	5.53×10^{-4}	5.20×10^{-4}	1.11×10^{-3}	1.98×10^{-3}
Dermal absorption	2.15×10^{-9}	2.00×10^{-9}	3.78×10^{-10}	1.53×10^{-9}	4.70×10^{-9}
Total direct exposure	2.02×10^{-3}	1.07×10^{-3}	6.62×10^{-4}	1.64×10^{-3}	3.81×10^{-3}

[a]ng I-TEQ/day.

TABLE 5.5
PCDD/Fs Dose[a] for Population Living around the Incinerator

			Percentiles		
	Mean	**SD**	**10th**	**50th**	**90th**
Direct dose	3.24×10^{-5}	1.80×10^{-5}	1.46×10^{-5}	2.87×10^{-5}	5.37×10^{-5}

[a]ng I-TEQ/day/kg.

TABLE 5.6
Noncancer Risk: Mean, Standard Deviation and 10th, 50th and 90th Percentiles

	Mean	SD	Percentiles 10th	50th	90th
Direct risk	1.44×10^{-2}	1.06×10^{-2}	5.18×10^{-3}	1.15×10^{-2}	2.65×10^{-2}

TABLE 5.7
Cancer Risk: Mean, Standard Deviation and 10th, 50th and 90th Percentiles

	Mean	SD	Percentiles 10th	50th	90th
Direct risk	1.42×10^{-6}	8.28×10^{-6}	6.20×10^{-7}	1.25×10^{-6}	2.36×10^{-6}

5.7.2 Risk Evaluation

The noncancer and cancer risks from direct exposure are shown in Table 5.6 and Table 5.7, respectively. The results show the 10th percentile, the central tendency estimates of risk (50th percentile) and the RME (reasonable maximum exposure; 90th percentile). It can be seen that the median (50th percentile) of noncancer risk due to PCDD/Fs in the population living in the area surrounding the MSWI of Tarragona is 0.015 The results reveal that the uncertainty of the risk estimated as defined by the ratio of the 90th to the 10th percentile is 5.1 (Table 5.6). With respect to total cancer risk, the median increment in individual lifetime is 1.25×10^{-6}, and the ratio between the 90th percentile and 10th percentile is about 3.8 (Table 5.7).

It can be concluded that the exposure to PCDD/Fs due to the MSWI in the Tarragona area is not producing health risks for the general population.

5.8 CASE STUDY: UNCERTAINTY ASSESSMENT BY MONTE CARLO SIMULATION FOR LCI AND IPA APPLIED TO MSWI IN TARRAGONA, SPAIN

The frameworks for uncertainty assessment by MC simulation for LCI and IPA described in Section 5.5 are applied to formerly inroduced case study of the MSWI in Tarragona, Spain.

5.8.1 Application of the Framework to Life-Cycle Inventory of Electricity Produced by a Waste Incinerator

This section presents an application of the framework for uncertainty assessment corresponding to the case of LCIs mentioned earlier. For this purpose, the industrial process of electricity produced by the MSWI, discussed in Chapter 1 to Chapter 4, was once more selected. The following goals were proposed for the sake of a more didactical and practical example:

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

5.8.1.1 Assigning Probability Distributions to Considered Parameters

The predominant pollutants identified and quantified during the implementation of the LCI for the MSWI study were selected by a combined quantitative and qualitative approach. The quantitative selection consisted of a dominance analysis performed on the basis of the results in the impact assessment carried out using the eco-indicator 95 method (see Chapter 3). Figure 5.9 presents the contribution of the considered pollutants to the total environmental potential impact measured by the eco-indicator 95. As a selection criterion, only the emissions with a contribution to the total environmental impact higher than 1% will be selected for the uncertainty assessment. The results of the quantitative selection established that the atmospheric emission of cadmium (Cd), carbon dioxide (CO_2), chloridric acid (HCl), nickel (Ni), sulfur dioxide (SO_2), other heavy metals (HMs) and particulate matter (PM) would be taken into account. Moreover, because of their carcinogenity and consideration as primary air pollutants in the ExternE project (EC, 1995, 2000), arsenic (As), carbon monoxide (CO) and PCDD/Fs were also to be considered (Figure 5.9).

A proper determination of the probability distribution is possible if data are extensively available, as in the case of measured emissions, electricity production, working hours and flow gas volume. Here the probability distributions were calculated from experimental data provided by the LCA study (STQ, 1998) and by the MSWI director by means of a report (Nadal, 1999) or personally. Based on a relevant number of measurements and their inherent variations, the normal or log-normal distribution was selected as the best-fitting probability density function for the respective types of data. 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 parameters with more than 30 measurements.

The software Crystal Ball allowed carrying out this fitting of probability distributions. The variation of the emissions in the study was enormous due to the constant variation in the waste's incinerated composition. The concentrations of the

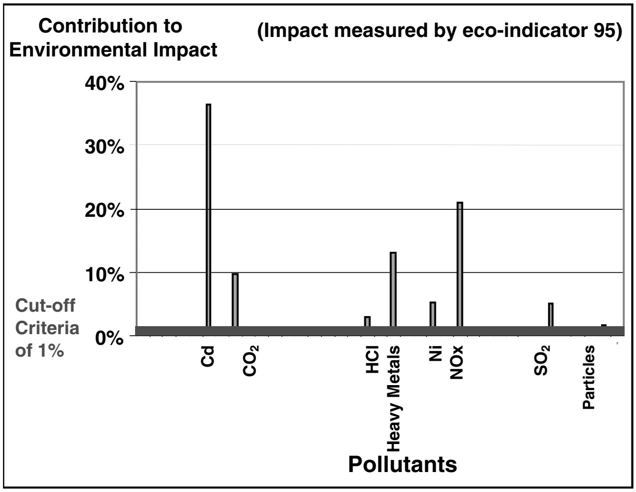

FIGURE 5.9 Selection of essential pollutants by dominance analysis. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

incineration process emissions together with their distribution type and deviations are presented in Table 5.8. Except for PCDD/Fs, the variations of the pollutant concentration emissions were fitted from experimental data by log-normal 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 HM concentrations in the emissions is in general much higher than that of the macropollutants SO_2, NO_x and CO. The heterogeneous feature of the waste used in the process can explain the enormous variation once more (Table 5.8).

The probability distributions for the PCDD/Fs concentrations in the emissions were considered to be log-normal with a geometric standard deviation of 2.0, according to the estimations published by Rabl and Spadaro (1999). The consideration was done due to a lack of sufficient experimental data on this substance. Also, in the case of the 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 the waste treatment process, count on little information about data quality. Consequently, the uncertainty estimations have been made according to the literature (Weidema and Wesnaes, 1996; Meier, 1997). Table 5.9 shows the technical site-specific data for the whole system, which embraces the operation of waste treatment, electricity production and consumption, and transportation, as well as its inherent inflows and outflows (see Chapter 2). All these technical data show a normal distribution. The variation in the data on the annual amounts of waste treated and electricity produced are described by their normal standard deviations because these statistical factors would be calculated from a sufficient amount of data.

For the annual working hours and gas volume flow, a 5% coefficient of variation was estimated according to the information given by the technical staff of the MSWI.

TABLE 5.8
Site-Specific Data: Concentrations in Emissions of the Incineration Plant[a]

Parameter	Unit	Distribution Type	Former situation[b] Mean value (σ_g)	Current situation[c] Mean value (σ_g)	Reference
As	mg/Nm^3	Log-normal	2.00×10^{-2} (3.4)	5.60×10^{-3} (3.4)	(STQ 1998)
Cd	mg/Nm^3	Log-normal	2.00×10^{-2} (1.7)	6.60×10^{-3} (1.7)	(STQ 1998)
CO	mg/Nm^3	Log-normal	4.00×10^{1} (1.5)	4.00×10^{1} (1.5)	(STQ 1998)
PCDD/Fs	pg/Nm^3	Log-normal	2.00×10^{1} (2.0)	2.00 (2.0)	(STQ 1998; Rabl and Spadaro 1999)
HCl	mg/Nm^3	Log-normal	5.16×10^{2} (1.6)	3.28×10^{1} (1.6)	(STQ 1998)
Heavy metals	mg/Nm^3	Log-normal	4.50×10^{-1} (2.5)	9.10×10^{-2} (2.5)	(STQ 1998)
Ni	mg/Nm^3	Log-normal	3.00×10^{-2} (2.2)	8.40×10^{-3} (2.2)	(STQ 1998)
NO_x	mg/Nm^3	Log-normal	1.91×10^{2} (1.5)	1.91×10^{2} (1.5)	(STQ 1998)
Particulate matter	mg/Nm^3	Log-normal	2.74×10^{1} (2.1)	4.80 (2.1)	(STQ 1998)
SO_2	mg/Nm^3	Log-normal	8.09×10^{1} (1.5)	3.02×10^{1} (1.5)	(STQ 1998)

[a] CO_2 emissions are determined stochiometrically.
[b] Without new filters.
[c] With new filters.
σ_g = geometric standard deviation

The probability distributions for other pieces of technical data had to be derived from the literature (Weidema and Wesnaes, 1996; Meier, 1997). Thus, a normal distribution with a CV of 10% was assumed for site-specific inflows and outflows, while for transportation a normal distribution with a CV of 20% was chosen due to the large uncertainty in the exact description of the waste transport. An enormous amount of the data used in the LCI is not directly related to the incineration process, but to the cycles of associated inputs, outputs and transport processes, as can be seen in Table 5.9. The data of the system under study were not obtained in a site-specific manner but from the ETH database (Frischknecht et al., 1996). These data have been collected from a Swiss perspective on a European scale. It is evident that the transfer of data to the Spanish situations definitely caused an uncertainty that, according to Meier (1997), differs depending on the considered pollutant. For infor-

TABLE 5.9
Site-Specific Data: Technical Variables of the Incineration Plant

Parameter	Unit	Distribution Type	Former situation[a] Mean value (CV)	Current situation[b] Mean value (CV)	Reference
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 and Wesnaes 1996; STQ 1998
Plastic proportion	%	Normal	13 (0.1)	13 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Electricity consumption	TJ	Normal	1.66 (0.1)	1.66 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Diesel	t	Normal	148.8 (0.1)	148.8 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Lubricant oil	t	Normal	2.3 (0.1)	2.3 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Lime (CaO)	kg	Normal	0 (0)	921,000 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Water deionized	m^3	Normal	19,665 (0.1)	19,665 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Water refrigeration	m^3	Normal	5,175 (0.1)	5,175 (0.1)	Weidema and Wesnaes 1996; STQ 1998

-- *continued*

TABLE 5.9 (continued)
Site-Specific Data: Technical Variables of the Incineration Plant

Parameter	Unit	Distribution Type	Former situation[a] Mean value (CV)	Current situation[b] Mean value (CV)	Reference
Water purified	m^3	Normal	7,360 (0.1)	7,360 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Unspecified water	m^3	Normal	8,122 (0.1)	33,120 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Ashes treated	kg	Normal	590,000 (0.1)	3,450,000 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Scrap treated	kg	Normal	2,740,000 (0.1)	2,740,000 (0.1)	Weidema and Wesnaes 1996; STQ 1998
Slag	t	Normal	42,208 (0.1)	42,208 (0.1)	Weidema and Wesnaes 1996; STQ 1998

[a]Without new filters.
[b]With new filters.
CV = coefficient of variation with the exception of electricity production and waste treated, which are expressed as normal standard deviation.

mation taken from databases, Meier (1997) proposed to assume classes of normal probability distributions with the following CVs:

- For data obtained by stochiometric determination, a CV of 2% needs to be considered.
- For actual emission measurements or data computable in well-known process simulation, a CV of 10% is expected.
- For well-defined substances or summed parameters, a CV of 20% can be assumed.
- For data taken from specific compounds by an elaborated analytical method, a CV of 30% is expected.

TABLE 5.10
Uncertainties in Measurements of Emissions in ETH Process Modules[a]

Parameter types	Distribution type	Uncertainty (CV)
Substances determined stochiometrically (CO_2)	Normal	0.02
Actual emission measurements or emissions of well-known processes depending on multiple parameters (CO, NO_x, SO_2)	Normal	0.10
Well-defined substances or sum parameters (As, Cd, HCl, HMs, Ni, PCDD/Fs)	Normal	0.20
Specific compounds with elaborated analytical methods (PM)	Normal	0.30

[a]According to Meier, M., *Eco-Efficiency Evaluation of Waste Gas Purification Systems in the Chemical Industry*, LCA Documents, Vol. 2, Ecomed Publishers, Landsberg, Germany, 1997, and Frischknecht, R. et al., Ökoinventare von Energiesystemen — Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. 3rd ed., ETH Zürich: Gruppe Energie-Stoffe-Umwelt, PSI Villigen: Sektion Ganzheitliche Systemanalysen, 1996.

CV = coefficient of variation.

For a better understanding of these estimates, Table 5.10 specifies which pollutant emission corresponds to which class under study. In addition, an example of 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. As a result of using this relatively certain assessment method, CO_2 received a CV of 2%. CO, NO_x 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. Because of more possibilities for errors, a CV of 10% was assumed for all of these compounds. For well-defined substances or summed parameters such as HM, PCDD/Fs and HCl, a CV of 20% was established. Finally, in the class for specific compounds with elaborated analytical methods, the uncertainty level was considered the highest with a CV of 30%. In the present case study, this CV was assumed for the life-cycle database information on particulate matter.

5.8.1.2 Assessing Uncertainties and Variations in the Calculation of LCI

Following the procedure related to uncertainty assessment for the proposed LCI, a Monte Carlo simulation was run for each situation with the probability distribution described previously. Its final result consisted of a set of histograms — one per selected pollutant — corresponding to the two scenarios proposed in the study: 1) Scenario 1: former situation and 2) Scenario 2: current situation.

Current situation (Scenario 2) refers to the incineration process carried out with an advanced acid gas treatment system (AGTS). On the other hand, in Scenario 1 the atmospheric emissions of the same unit without AGTS were evaluated. Each simulation has been made in one separate run per scenario. Because of the inherent variability of the MC model, it is not possible to affirm that the set of values relative to the input variables used in the run of the current situation are going to be the same as those in the former situation. The reason for this is that every run occurs in different ways due to the generation of random numbers. In order to verify the importance of this variability on the final outcome, both simulations were also carried out in one run, and the results obtained showed negligible variations.

Figure 5.10 presents the results of the substances considered in this study in the current situation with advanced AGTS. In the x-axis, it is possible to observe the amount of pollutant emission per energy produced. The y-axis shows the probability of each value of the life-cycle emissions. As mentioned before, 10,000 iterations were carried out with the software Crystal Ball. The mean atmospheric emissions of the heavy metals per 1 TJ of electricity produced by the incinerator were 7.50×10^{-1} kg with a normal standard deviation of 5.50×10^{-1} kg/TJ. Because the density distribution of the results is best adjusted by log-normal density function, a geometric mean of 6.10×10^{-1} kg for HMs/TJ and a geometric standard of 1.92 were calculated. On this basis, a 68% confidence interval from 3.18×10^{-1} kg/TJ to 1.17 kg/TJ was obtained.

Because the other pollutants can be adjusted well by a log-normal distribution, all the atmospheric emissions were treated in the same way, with their mean in the 68% confidence interval. In this framework, the calculated values of μ_g and σ_g for As, Cd and CO were, respectively, 3.37×10^{-2} kg/TJ (2.30), 2.62×10^{-2} kg/TJ (1.67) and 2.43×10^{2} kg/TJ (1.37).

The CO_2 emissions of the incineration process were not determined by the measurements assumed to have a log-normal distribution. In this case, the total amount of waste treated was multiplied by the percentage fraction of plastics present on it because this material is the only one in the mixture component that originally comes from fossil fuels. Taking into account that 1 kg of plastic burned produces approximately 2.0 kg of CO_2, the total CO_2 produced by the waste incineration was divided by the gas volume emitted to the atmosphere through the stack in order to determine the CO_2 concentration released. Thus, for this pollutant a normal distribution with a μ_g of 3.10×10^{5} kg/TJ and a σ_g of 1.13 was obtained.

The profile of the LCI results for SO_2 — 2.12×10^{2} kg/TJ (1.29) — to the same situation is similar to those for NO_x and CO due to the same order of magnitude in the σ_g for the incinerator's emissions. In Figure 5.10, the LCI results obtained by MC simulation for the particulate matter (particles) in the former situation (Scenario 1) are also illustrated (μ_g 1.50×10^{2} kg/TJ, σ_g 1.93). If the results of the former situation and the current situation are compared, a clear change can be seen in the probability distribution from a log-normal to a rather normal one after the installation of the advanced AGTS.

The mean values with confidence intervals for the former situation and the current situation related to all studied pollutants are presented in Figure 5.11. Heavy metals were only considered as a summed parameter. For the PCDD/Fs, HMs, SO_2 and HCl a clear reduction can be observed with the installation of the advanced AGTS, especially for the first one. On the other hand, for CO_2, CO, PM and NO_x

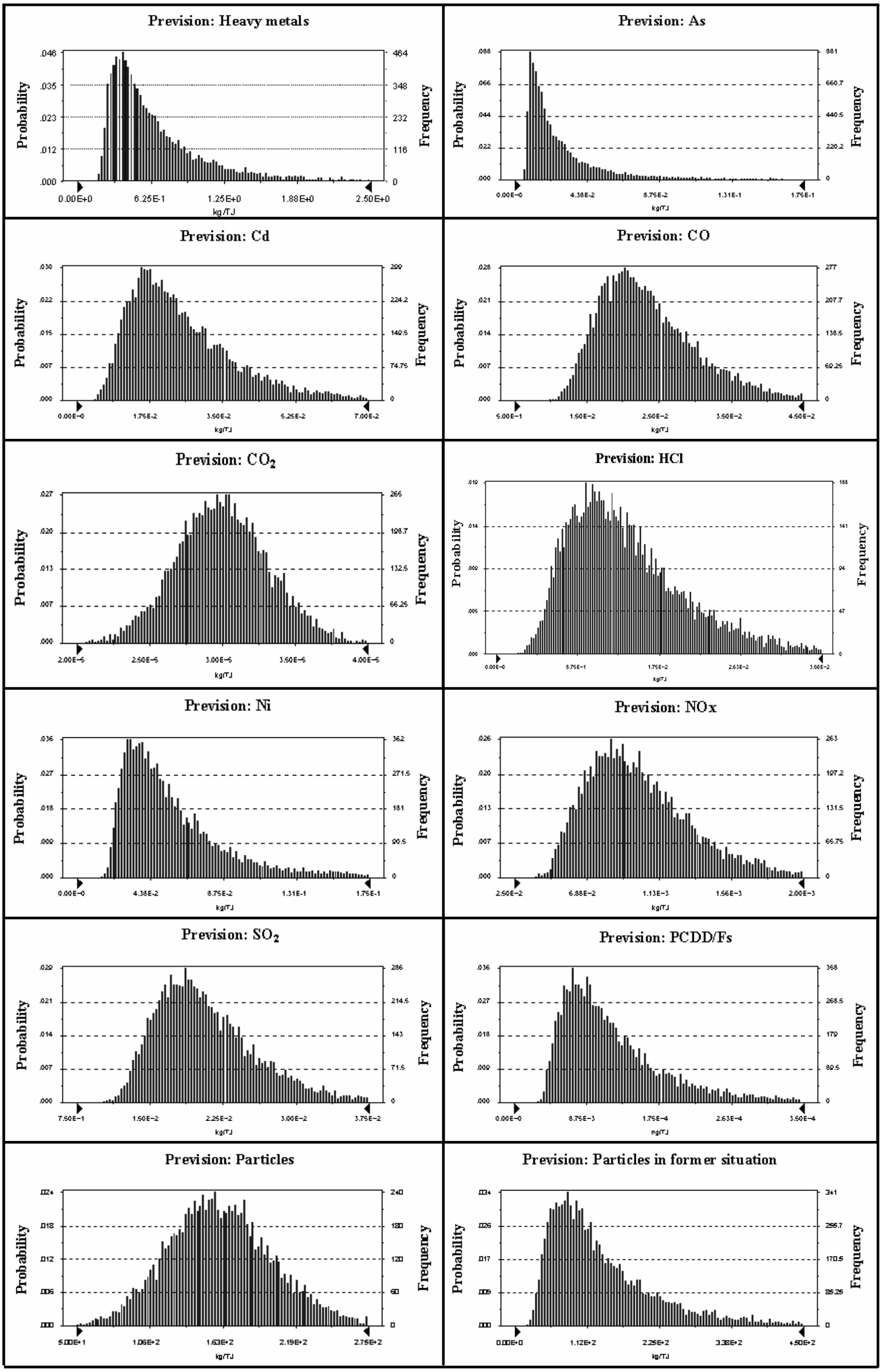

FIGURE 5.10 Monte Carlo simulation LCI results for the different substances. (Adapted from Sonnemann, G.W., et al., *J. Cleaner Prod.*, 11, 279–292, 2002.)

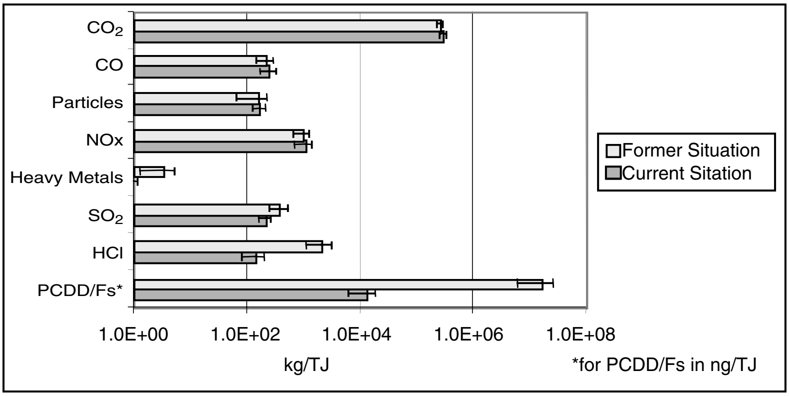

FIGURE 5.11 LCI results with confidence interval of 68%. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

no variation in the life-cycle emissions per TJ of electricity produced is found. For these cases, changes were smaller than the given confidence intervals. Here, it is evident that the detected uncertainty and variability interfere in the results and influence their interpretation (Figure 5.11).

5.8.1.3 Determining the Most Relevant Parameters in LCI by Sensitivity Analysis

In order to determine the most relevant parameters in the case of the MSWI of Tarragona, a sensitivity analysis of the LCI results was carried out. The results of the sensitivity analysis for the PCDD/Fs are presented for the former situation and the current situation in Figure 5.12 and Figure 5.13, respectively. In both situations the PCDD/Fs emitted by the incinerator process were the most important parameter, with contributions to the variance of 99.9 and 99.6%, respectively. The same results were obtained for the other pollutants: percentages over 95% with the exception of the particulate matter. Thus, this contaminant will be discussed in more detail.

In Scenario 1, which is defined by the former situation, the major emission of PM was due to the process of incineration. The effect of the other steps of the system under study was practically negligible, as can be seen in Figure 5.14.

On the other hand, in the current situation (Scenario 1) the process of production of the lime used in the advanced AGTS has been added to the life-cycle, especially considering that this process generates a huge amount of dust. Thus, as shown in Figure 5.15, it contributes with 83.6% to the variance of the PM from the global system. Finally, the particles emitted during the incineration concur with only 15.6% and all the other processes embraced by the boundaries sum less than 1%.

As a conclusion, the advanced AGTS reduces the concentration of heavy metals and PCDD/Fs, PM, SO_2 and HCl in the gas flow emitted to the atmosphere from the incinerator. The concentrations of other pollutants such as NO_x and CO emissions are kept constant by their turn. Consequently, from the point of view of environ-

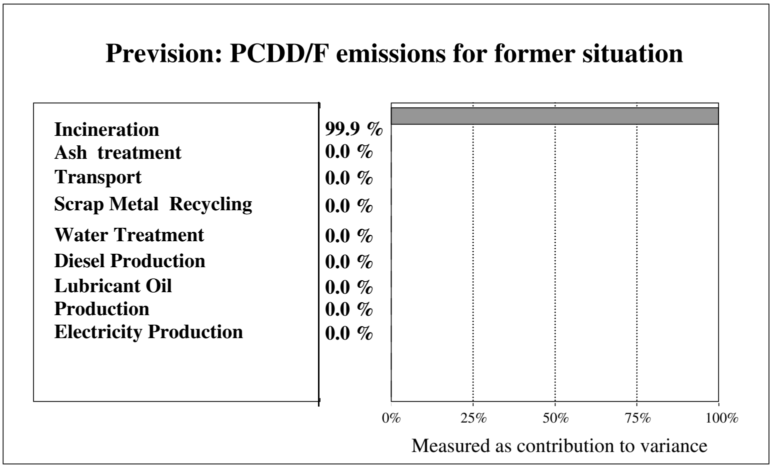

FIGURE 5.12 Sensitivity analysis of the LCI results for PCDD/Fs in the former situation. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

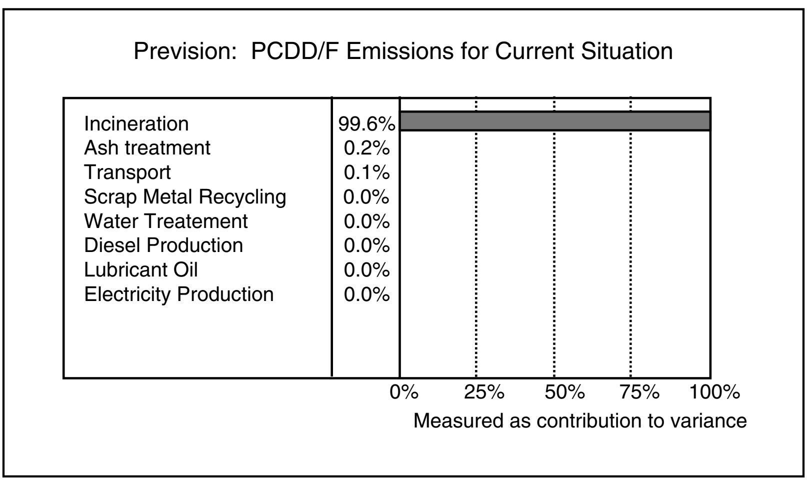

FIGURE 5.13 Sensitivity analysis of the LCI results for PCDD/Fs in the current situation. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

mental risk assessment, the risk of causing hazardous effects to human health and the environment in the area surrounding the incineration plant is clearly reduced. Also, on the basis of an LCA it was possible to observe a reduction of atmospheric emissions from the whole system per TJ of electricity produced and for the pollutant heavy metals, PCDD/Fs, HCl and SO_2. The same cannot be observed in the case of particulate matter. Moreover, the absolute values show a very slight, though not significant, increase of life-cycle emissions per TJ of electricity produced for CO_2,

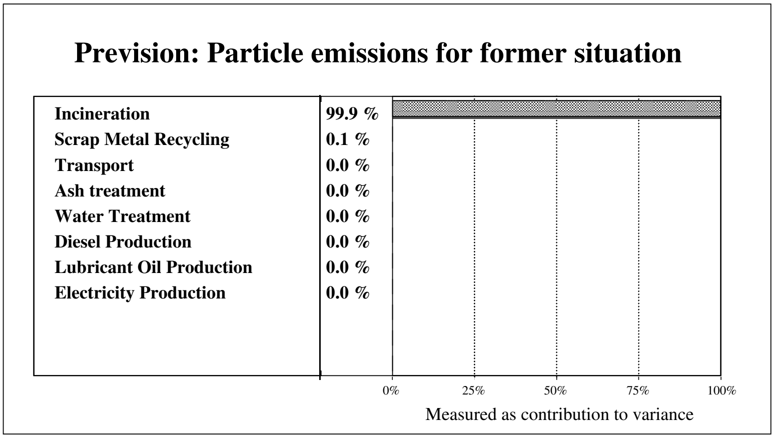

FIGURE 5.14 Sensitivity analysis of the LCI results for particles in the former situation. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

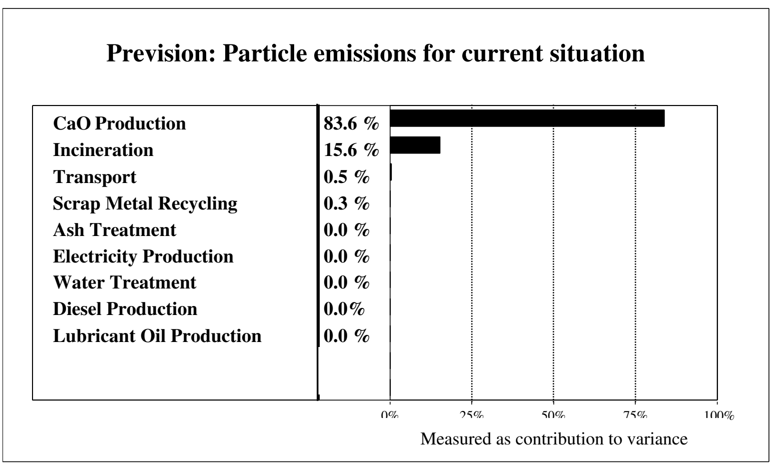

FIGURE 5.15 Sensitivity analysis of the LCI results for particles in the current situation. (Reprinted from *J. Cleaner Prod.,* 11, Sonnemann, G.W. et al., pp. 279–292, ©2002 with permission from Elsevier.)

CO, and NO_x. These emissions are higher in comparison with those measured in Scenario 1 due to the lime production process added to the system, the longer transportation distances resulting from higher inflows and outflows, and a lower efficiency in production of electricity (former situation = 158.56 TJ/yr.; current situation = 149.55 TJ/yr).

5.8.2 Application of Framework to IPA of Waste Incinerator Emissions on a Local Scale

Following the same procedure used for the LCI results, the framework for uncertainty assessment in IPA was also applied to the case study of local human health impacts due to the emissions of the MSWI in Tarragona. As in the approach used previously, some goals were proposed:

- Assigning probability distributions to the parameters considered in the study
- Assessing the uncertainties and the variation in application
- Determining the most relevant parameters in such an IPA by sensitivity analysis

5.8.2.1 Assigning Probability Distributions to Considered Parameters

As explained by Rabl and Spadaro (1999), the probability distributions mainly used in environmental damage estimations are the normal distribution and the log-normal probability distribution. As mentioned earlier, all normal distributions are symmetric and have bell-shaped density curves with a single peak. The log-normal distribution, in turn, is calculated assuming that the logarithm of the variable has a normal distribution. As in the previous case of uncertainty assessment in LCI, the proper determination of the probability distribution is only possible if measured data are extensively available, as in the case of atmospheric emissions, electricity production, working hours and flow gas volume. If the parameters are based on little proper information, literature values must be applied to determine the probability distribution. That is the case of PCDD/Fs emissions, dispersion modeling results, dose–response and exposure–response functions, population data and monetary valuation.

The necessary information for determining the probability distributions from measured data was taken from the LCA study of the Servei de Tecnologìa Química, STQ (1998), and from information given by the director of Tarragona's 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 about uncertainty analysis of environmental damages and costs by Rabl and Spadaro (1999). The estimation of the uncertainty for the dispersion model was taken from McKone and Ryan (1989). Further local information was obtained from the Public Health Plan of the Tarragona region (GenCat, 1997) and from a diagnosis on the socioeconomic development of the Tarragona province by Soler (1999).

The probability distributions used in the study are summarized from Table 5.11 to Table 5.13. "Variable mean" stands for an enormous number of values that are not constant but differ depending on the grid and pollutant considered.

In Table 5.11 the technology and modeling parameters are presented together with their respective probability distributions and characteristics. The technology parameters in Table 5.11 consist respectively of electricity production, working hours and specific characteristics of the incinerator (stack dimensions, geographical situ-

TABLE 5.11
Technology and Modeling Data (Emission Values for Former Situation 1)

Parameter	Units	Distribution	Mean	Dev.	Reference
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	Log-normal	81.13	(σ_g) 1.5	STQ 1998
NO_x (emissions)	mg/Nm^3	Log-normal	191	(σ_g) 1.5	STQ 1998
PM (emissions)	$mgNm^3$	Log-normal	28.57	(σ_g) 2.1	STQ 1998
CO (emissions)	mg/Nm^3	Log-normal	40	(σ_g) 1.5	STQ 1998
As (emissions)	$\mu g/Nm^3$	Log-normal	15.1	(σ_g) 3.4	STQ 1998
Cd (emissions)	$\mu g/Nm^3$	Log-normal	19.9	(σ_g) 1.7	STQ 1998
Ni (emissions)	$\mu g/Nm^3$	Log-normal	33.27	(σ_g) 2.2	STQ 1998
PCDD/F (emissions)	ng/Nm^3	Log-normal	2	(σ_g) 2	STQ 1998; Rabl and 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	°	Point estimate	41.19	—-	Nadal 1999
Geographical longitude	°	Point estimate	1.21	—	Nadal 1999
Elevation at site	m	Point estimate	90	—	Nadal 1999
Incremental emission concentration	mg/Nm^3	Log-normal	variable	(σ_g) 2	McKone and Ryan 1989

CV = coefficient of variation; σ = normal standard deviation; σ_g = geometric standard deviation; dev. = deviation.

ation, etc.) and the parameters properly related to the emissions (concentration of pollutants, total volume, temperature, etc.). The increment of the emission concentration is considered a modeling parameter. According to the probability distributions obtained using Crystal Ball, the variations of electricity production, working hours and flow gas volume have a normal distribution and the emissions behave like cadmium in a log-normal 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 characterized by geometric

TABLE 5.12
Impact Human Health Data

Parameter	Unit	Distribution	Mean	Dev.	Reference
		Dose–response and exposure–response functions			
Chronic YOLL		Log-normal	0.00072	(σ_g) 2.1	IER 1998; Rabl and Spadaro 1999
Acute YOLL		Log-normal	Variable	(σ_g) 2.1	Rabl and Spadaro 1999
Cancer		Log-normal	Variable	(σ_g) 3	Rabl and Spadaro 1999
Others		Log-normal	Variable	(σ_g) 2.1	Rabl and Spadaro 1999
		Damage factors			
Chronic YOLL		Log-normal	1	(σ_g) 1.5	Rabl and Spadaro 1999
Acute YOLL		Log-normal	1	(σ_g) 4	Rabl and Spadaro 1999
Cancer		Log-normal	1	(σ_g) 1.6	Rabl and Spadaro 1999
Others		Log-normal	1	(σ_g) 1.2	Rabl and 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	# inhab	Normal	Variable	CV 0.01	Soler 1999

CV = coefficient of variation; σ_g = geometric standard deviation; dev. = deviation.

standard deviations ranging from 1.5 to 3.4. The parameters corresponding to the stack dimensions and the geographical situation are point estimates and were provided by the director of Tarragona's MSWI (Nadal, 1999). The incremental emission concentration has log-normal distribution with a geometric standard deviation of 2, according to the uncertainty estimates for the dispersion model by McKone and Ryan (1989).

TABLE 5.13
Monetary Valuation Data

Parameter	Units	Distribution	Mean	Dev.	Reference
Chronic YOLL	Euro	Log-normal	84,330	(σ_g) 2.1	Rabl and Spadaro 1999
Acute YOLL	Euro	Log-normal	155,000	(σ_g) 2.1	Rabl and Spadaro 1999
Cancer	Euro	Log-normal	1,500,000	(σ_g) 2.1	Rabl and Spadaro 1999
Others	Euro	Log-normal	Variable	(σ_g) 1.2	Rabl and Spadaro 1999

σ_g = geometric standard deviation; dev. = deviation.

Table 5.12 presents the human health parameters. As can be seen in Table 5.12, uncertainty and variation are parts of the public health data (e.g., population, percentage of children, adults and elders, percentage of asthmatics or baseline mortality). The dose–response and exposure–response functions are characterized by the log-normal probability distribution provided by Rabl and Spadaro (1999). The mean value for chronic years of lost life (YOLL) due to particulate matter was provided by IER (1998) and the other means vary in function of the respective pollutants. The definition and calculation of the damage factors (e.g., chronic YOLL, acute YOLL and cancer) involve another factor of uncertainty. The probability distributions for these factors, which are used only for aggregation and further multiplication (therefore equal to 1), have been taken from Rabl and Spadaro (1999), who supposed them to have log-normal distribution with a geometric standard deviation between 1.2 and 4. The description of the population properties has been identified as a point estimate or with normal distribution according to the values provided once more by IER (1998) and by GenCat (1997). Finally, the possible variation of the number of inhabitants in each grid can be described by a normal distribution based on the study made by Soler (1999).

Table 5.7 shows the monetary valuation parameters. All probability distributions are log-normal and were taken from Rabl and Spadaro (1999).

5.8.2.2 Assessing Uncertainties and Variation in the Calculation of IPA

Following the IPA framework, 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 (Table 5.14). By using the obtained probability distributions for the essential parameters in an MC simulation, the result for environmental damage costs has been transformed from a concrete value into a probability distribution around a mean value. Because the distributions of most parameters in Table 5.11 to Table 5.13 are log-normal and not normal, the final distribution of each result has a log-normal distribution too. In the same way as in the application to LCI, each simulation has

TABLE 5.14
Statistical Parameter Describing Results of the Environmental Damage Estimation in Scenarios 1 and 2 as External Costs[a]

Parameter	Scenario 1[b]	Scenario 2[c]
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

[a]mU.S.$ per kWh (1E-3 U.S.$/kWh).
[b]Without filters.
[c]With filters.

been made in one separate run per scenario. Because of the inherent variability of the Monte Carlo model, it is not possible to affirm that the set of values for the input variables used in the run of Scenario 2 (current situation) will be the same as those used in Scenario 1 (former situation). The same was done with the calculation of LCI uncertainties and, due to 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 also been made in one run; it could be checked that the results are the same because the variations are negligible.

Figure 5.16 presents the first result, which is the case of the incineration process supported by the advanced AGTS. 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 mean of the environmental damage cost in Scenario 2 is 0.87 mU.S.$ per kWh (with $m = 10^{-3}$). 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 Table 5.14. The geometric standard deviation from them is 2.62.

The second case occurs in time before the first one, when the incinerator did not have an advanced AGTS. The emissions of pollutants were more important and consequently the environmental damage cost is much higher, with 3.73 mU.S.$ per kWh. The probability distribution of this result can be found in Figure 5.17. This is explained because the only important change corresponds to the mean values of 10 parameters, including pollutants and electricity production, which are lower with an advanced AGTS installed.

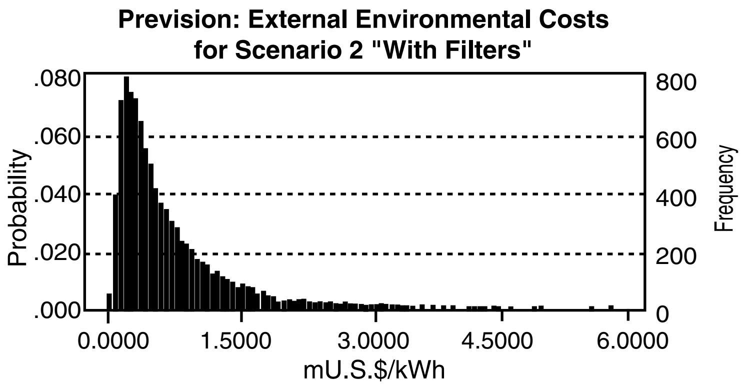

FIGURE 5.16 MC simulation results for IPA of MSWI emissions in the current situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

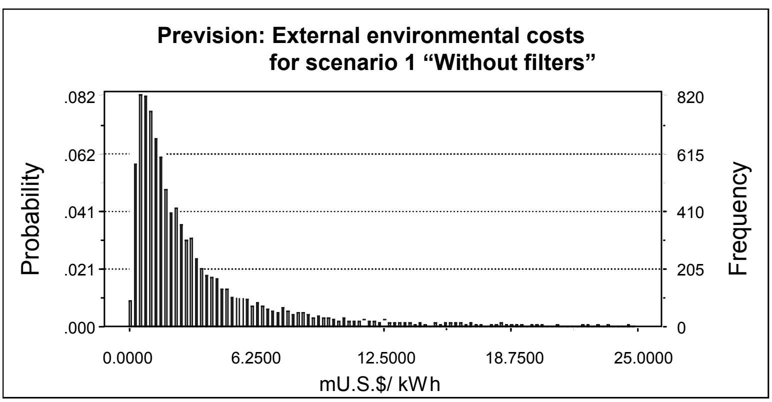

FIGURE 5.17 MC simulation results for IPA of MSWI in the former situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

The results in Table 5.14 show that the uncertainty and variability calculated using MC simulation are less than those calculated 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 higher than 4, according to Rabl and Spadaro (1999).

Figure 5.18 illustrates the differences between the means obtained in Scenarios 1 and 2, both with confidence intervals of 68%. It is possible to see a clear reduction of the damage cost. As can also be seen in Table 5.8, the 68% confidence interval for the scenario with advanced AGTS embraces a range from 0.21 to 1.44 mU.S.$ per kWh, while for the other the same confidence interval has a range between 0.78 and 6.15 mU.S.$ per kWh. The inferior bound of the confidence interval is the maximum range of possible errors; for the result of the former situation, the inferior bound is in the same order of magnitude as the mean for the current situation. Thus, according to the results, there are important uncertainties. However, a clear reduction in terms of the damage cost can be foreseen within a confidence interval of 68% when comparing the two different operation scenarios (Figure 5.18).

5.8.2.3 Determining the Most Relevant Parameters in IPA by Sensitivity Analysis

The results of the sensitivity analysis for the pollutants are presented in Figure 5.19 for Scenario 1 without advanced AGTS. Some interesting results can be extracted. The graph shows all the pollutants and their contribution to the final result. Obviously, the emission of particulate matter is the most important parameter, with 92.1% of the total damage. The NO_x seems to be the second important pollutant, while the rest produce negligible damage. Figure 5.20 for Scenario 2, i.e., with advanced AGTS, is very similar; the particulate emissions contribute more than 99.6% to the total environmental damage cost. Because of the major emissions of particulate matter, in the former situation (Scenario 1) the mentioned percentage of NO_x is practically negligible. Taking into account that the MSWI is the emission source and the major public concern of producing dioxins, the result of the IPA for the case study shows that little of the total human health damage is contributed by air emissions. Figure 5.21 and Figure 5.22 present the results of the sensitivity analysis on the sources of health impacts to total damage costs. Most damage is caused by the loss of life expectancy, expressed as YOLL. If the damage appears in the near term, it is called acute, and if it appears in the 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

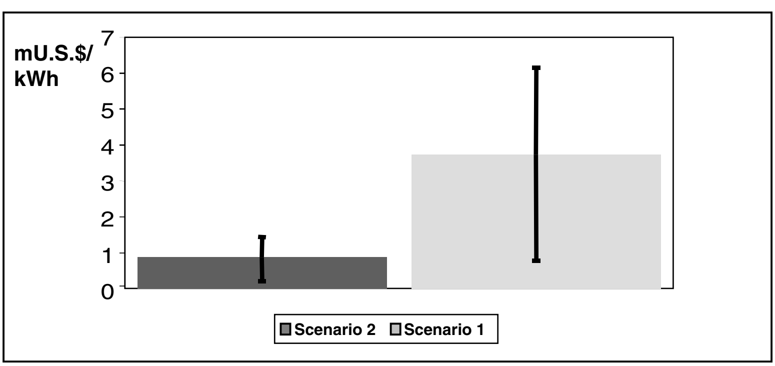

FIGURE 5.18 Comparison of the MC simulation results for the former situation (Scenario 1) without advanced gas cleaning system and the current situation (Scenario 2) with such an installation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

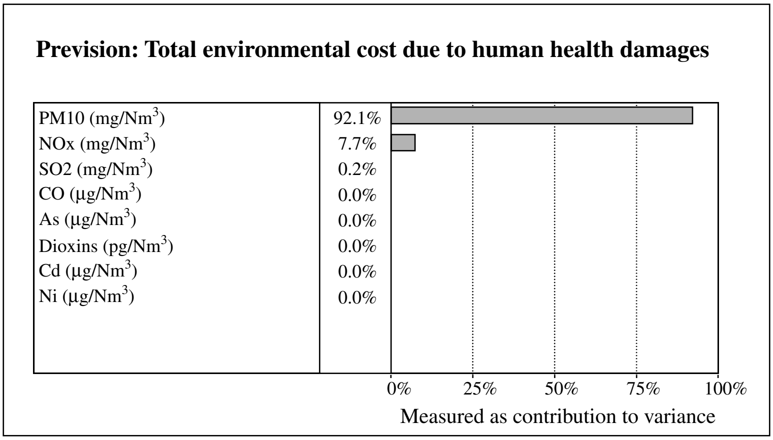

FIGURE 5.19 Sensitivity analysis for pollutants of the current situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

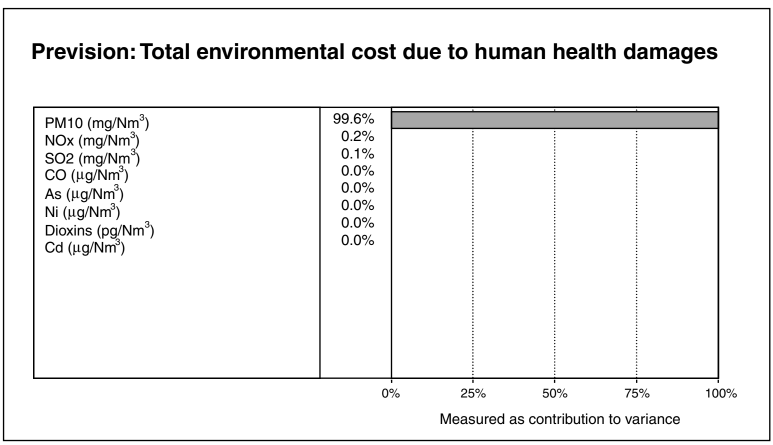

FIGURE 5.20 Sensitivity analysis for pollutants of the former situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

are less important by far. Figure 5.22 (the results for Scenario 1, without advanced AGTS) is a little bit different from Figure 5.21 because of the major concentration of particulate matter. This pollutant matter has more influence on the chronic YOLL and an increase in its concentration produces an increase in the importance of the chronic YOLL.

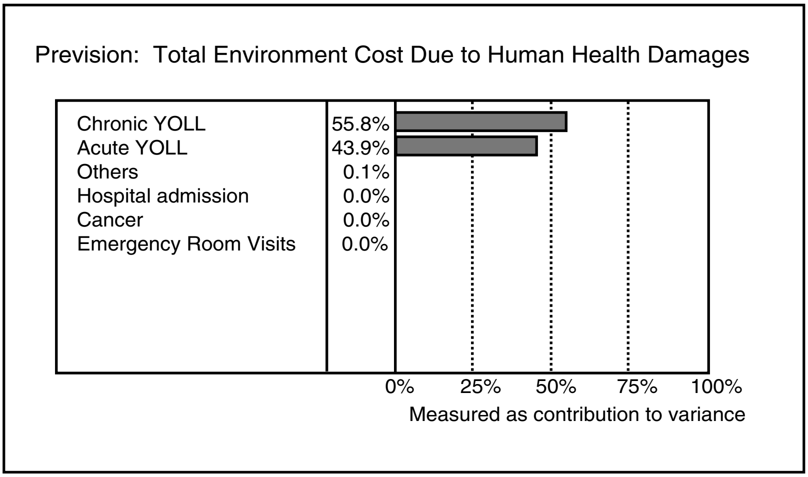

FIGURE 5.21 Sensitivity analysis for health impacts of the current situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

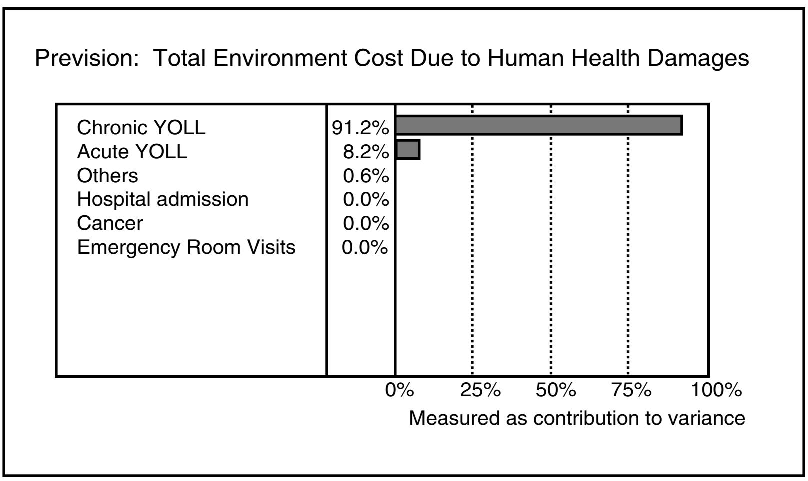

FIGURE 5.22 Sensitivity analysis for health impacts of the former situation. (Reprinted from *Environ. Int.,* 28, Sonnemann, G.W. et al., pp. 9–18, ©2002 with permission from Elsevier.)

5.8.3 Comparison of Uncertainties in Life-Cycle Inventory and IPA

As a general conclusion, it can be said that the damage estimations carried out in the IPA contain 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. In detail, the results for the LCI show a geometric standard deviation between σ_g 1.13 for CO_2, 1.29 for SO_2, and 1.92 for heavy metals or 2.30 for As. That means that 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 the 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 influenced less by the emissions (except for As with a σ_g 3.4, for Ni 2.2 and for the PM 2.1) than by the impact of 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 σ_g of 2.1 in the same way as the dispersion model with σ_g of 2. To overcome incomparability related to the dispersion models, using an internationally accepted reference model is proposed. Using homogeneous dose–response and exposure–response functions approved by the World Health Organization (WHO) could have the same effect in the future.

5.9 QUESTIONS AND EXERCISES

1. Explain the main differences between uncertainty and variability in environmental systems analysis.
2. In which categories are the sources of uncertainty classified in tools?
3. Distinguish which of the following related situations would be sources of uncertainties and which would be sources of variability in the tools presented in Chapter 5 of a coal power plant:
    - Location of the factory
    - Time of the study
    - Data related to the prevalence of the winds
    - Percentages of age, gender and diseases of population
    - Extrapolation from animal studies to humans
    - Sample sizes for animal and human studies
4. Associate the following situations in the same tool to the types of uncertainties and variability listed in the column on the right side:

Use of data corresponding to emissions associated with a former situation	**Uncertainty / choices**
• Selection of kJ per kg of product as functional unit	• Variability among sources and objects
• Use of data belonging to a system originally from a geographical area different from that subject to study	• Parameter uncertainty
• Human preferences involved in the system under study	• Model uncertainty
• Use of simplified models for the calculation of factors	• Spatial/temporal variability
• Presence of multiwaste processes	• Uncertainty choices

5. What are the main sources of parameter uncertainty and how are they reflected in the outcomes of an LCA?
6. Which implications for an LCIA have the nonconsideration of the model spatial and temporal characteristics?
7. Would the use of a parameter uncertainty analysis in an LCA be suitable when large model uncertainties have been detected?

8. What are the general strategies for the assessment of uncertainties in LCA studies?
9. List the main steps of an MC simulation.
10. Explain the relative importance of the different parameters for the final result when estimated by MC simulation.
11. In the framework of uncertainty assessment in IPA, a large amount of damage function data is compiled. Why can't all of them be considered? Give an explanation.
12. In an assessment study by MC simulation, data uncertainties appear as a percentage of the standard deviation, and a log-normal distribution is used. Discuss the suitability of this choice.
13. Explain the main sources of uncertainties in IPA and LCI, respectively.
14. What is the importance of determining the probability distributions of ERA data? Describe which kind of probability distribution can be associated with the following events:
    - The results of a weight study for a group of people: 155.6 lb; 141.7 lb; 158.6 lb; 174.2 lb; 172.1 lb; 164 lb; 168.9 lb; 139.2 lb; 147 lb; 156.2 lb; 171.7 lb
    - The concentrations in ppm of a toxin in samples taken at different points of a river: 0.123; 0.095; 0.154; 0.298; 0.365; 0.612; 0.389; 0.474; 0.299; 0.494; 0.341; 0.612; 0.511; 0.744; 0.519; 0.654; 0.476; 0.437; 0.365; 0.26; 0.166; 0.198; 0.108; 0.165
    - The results of a study about relative abundance (%) of endangered species in an ecosystem: 43; 26; 12.1; 7.4; 3.6; 2.5; 1.4; 1.2; 1; 0.6; 0.3; 0.2; 0.1; 0.08; 0.07; 0.06; 0.05; 0.04

   Do any of these distributions correspond to normal distribution and to log-normal?
15. Using the example of the LCA of a chair given in Chapter 2:
    - Assuming experimental data related to inputs and wastes of a similar process in Europe, discuss the probability distributions of the parameters considered in an eventual study in your country.
    - Identify the possible uncertainties and variability if an MC simulation is performed.

REFERENCES

Arija, V., Salas, J., Fernández-Ballart, J., Cucó, G., Martí-Henneberg, C. (1996), Consumo, hábitos alimentarios y estado nutricional de la población de Reus (IX). Evolución del consumo de alimentos, de su participación en la ingestión de energía y nutrientes y de su relación con el nivel socioeconómico y cultural entre 1983 y 1993, *Med. Clinica*, 106, 174–179.

Decisioneering (1996), Crystal Ball Version 4.0 — Guide de l'utilisateur, Editions MEV, Versailles, France.

EC (European Commission) (1995), *ExternE — Externalities of Energy*, 6 vols., EUR 16520–162525. DG XII Science, Research and Development, Brussels.

EC (European Commission) (2000), *Externalities of Fuel Cycles*, vol. 7: *Methodology Update*, report of the JOULE research project ExternE (Externalities of Energy), EUR 19083. DG XII Science, Research and Development, Brussels.

Environmental Protection Agency (EPA) (1990), Methodology for assessing health risks associated with indirect exposure to combustor emissions, PB90–187055.

Finnveden, G. and Lindfors, L.G. (1998), Data quality of life-cycle inventory data — rules of thumb, *Int. J. LCA*, 3(2), 65–66.

Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G., Hischier, R., Martin, A. (ETH Zürich), Dones, R., and Gantner, U. (PSI Villigen) (1996), Ökoinventare von Energiesystemen — Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. 3rd ed., ETH Zürich: Gruppe Energie-Stoffe-Umwelt, PSI Villigen: Sektion Ganzheitliche Systemanalysen.

GenCat (Generalitat de Catalunya) (1997), Pla de salut de la reió sanitària Tarragona, Servei Català de la Salut, Tarragona, Spain.

Hanssen, O.J. and Asbjornsen, A.A. (1996), Statistical properties of emission data in life-cycle assessments, *J. Cleaner Prod.*, 4(3–4), 149–157.

Hawley, J.K. (1985), Assessment of health risk from exposure to contaminated soil, *Risk Anal.*, 5(4), 289–302.

Hofstetter, P. (1998), *Perspectives in Life-Cycle Impact Assessment — A Structured Approach to Combine Models of the Technosphere*, *Ecosphere and Valuesphere*, Kluwer Academic Publishers, London.

Huijbregts, M.A.J. (1998), Application of uncertainty and variability in LCA (part I) — a general framework for the analysis of uncertainty and variability in life-cycle assessment, *Int. J. LCA*, 3(5), 273–280.

Huijbregts, M.A.J. and Seppälä, J. (2000), Towards region-specific, European fate factors for airborne nitrogen compounds causing aquatic eutrophication, *Int. J. LCA,* 3(5), 65–67.

Huijbregts, M.A.J., Norris, G., Bretz, R., Ciroth, A., Maurice, A., von Bahr, V., Weidema, B., and Beaufort, A.S.H. (2000), Framework for modeling data uncertainty in life-cycle inventories, *Int. J. LCA*, 6(3),127–132.

IER (Institut für Energiewirtschaft und Rationale Energieanwendung) (1998), EcoSense 2.1, software, University of Stuttgart, Germany.

Katsumata, P.T. and Kastenberg, W.E. (1997), On the assessment of health risks at superfund sites using Monte Carlo simulations, *J. Environ. Sci. Health*, A32 (9 and10), 2697–2731.

LaGrega, M.D., Buckingham, P.L., and Evans, J.C. (1994), *Hazardous Waste Management*, Mc Graw-Hill, New York.

Maurice, B., Frischknecht, R., Coehlo-Schwirtz, V., and Hungerbühler, K. (2000), Uncertainty analysis in life-cycle inventory — application to the production of electricity with French coal power plants, *J. Cleaner Prod.*, 8(2), 95–108.

McKone, T.E. and Ryan, P.B. (1989), Human exposures to chemicals through food chains — an uncertainty analysis, *Environ. Sci. Technol.*, 23, 1154–1163.

Meier, M. (1997), *Eco-Efficiency Evaluation of Waste Gas Purification Systems in the Chemical Industry*, *LCA Documents*, Vol. 2, Ecomed Publishers, Landsberg, Germany.

Metzger, J.N., Fjeld, R.A., Hammonds, J.S., and Hoffman, F.O. (1998), Evaluation of software of propagating uncertainty through risk assessment models, *Hum. Ecol. Risk Assess.*, 4(2), 263–290.

Nadal, R. (1999), Planta incineradora de residuos sólidos urbanos de Tarragona, report for Master en Enginyeria i Gestió Ambiental (MEGA), Universitat Rovira i Virgili, Tarragona, Spain.

National Research Council (1994), Science and Judgement in Risk Assessment, Committee on Risk Assessment of Hazardous Air Pollutants. Board on Environmental Studies and Toxicology. Commission on Life Sciences, National Academy Press, Washington, D.C.

Nessel, S.G., Butler, J., Post, G.B., Held, J.L., Gochfeld, M., and Gallo, M.A. (1991), Evaluation of the relative contribution of exposure routes in a health risk assessment of dioxin emissions from a municipal waste incinerator, *J. Exposure Anal. Environ. Epidemiol.*, 1(3), 283–307.

Pesonen, H., Ekvall, T., Fleischer, G., Huppes, G., Jahn, C., Klos, Z.S., Rebitzer, G., Sonnemann, G.W., Tintinelli, A., Weidema, B.P., and Wenzel, H. (2000), Framework for scenario development in LCA, *Int. J. LCA*, 5(1), 21–30.

Rabl, A. and Spadaro, J.V. (1999), Environmental damages and costs — an analysis of uncertainties, *Environ. Int.*, 25(1), 29–46.

Rolaf van Leeuwen, F.X. and Younes, M., (1998), WHO revises the tolerable daily intake (TDI) for dioxins, *Organohalogen Compounds*, 38, 295–298.

Ros, M. (1998), Unsicherheit und Fuzziness in ökologischen Bewertungen — Orientierungen zu einer robusten Praxis der Ökobilanzierung, Ph.D. Thesis, ETH Zürich, Switzerland.

Roseberry, A.M. and Burmaster, D.E. (1991), A note: estimating exposure concentrations of lipophilic organic chemicals to human via finfish, *J. Expos. Anal. Environ. Epidemiol.*, 1, 513–521.

Schuhmacher, M., Granero, S., Xifró, A., Domingo, J.L., Rivera, J., and Eljarrat, E. (1998a), Levels of PCDD/Fs in soil samples in the vicinity of a municipal solid waste incinerator, *Chemosphere*, 37, 2127–2137.

Schuhmacher, M., Domingo, J.L., Llobet, M., Süinderhauf, W., and Müller, L. (1998b), Temporal variation of PCDD/F concentrations in vegetation samples collected in the vicinity of a municipal waste incinerator (1996–1997), *Sci. Total Environ.*, 218, 175–183.

Shin, D., Lee, J., Yang, J., and Yu, Y., (1998), Estimation of air emission for dioxin using a mathematical model in two large cities of Korea, *Organohalogen Comps.*, 36, 449–453.

Soler, S. (1999), Diagnosi socioeconòmica i estratègies de desenvolupament de la provincia de Tarragona, report of the Unitat de Promició i Desenvolupament, Universitat Rovira i Virgili, Tarragona, Spain.

Sonnemann, G.W., Pla, Y., Schuhmacher, M., and Castells, F. (2002a) Framework for the uncertainty assessment in the impact pathway analysis with an applicaiton on a local scale in Spain, *Environ. Int.,* 28, 9–18.

Sonnemann, G.W. Schuhmacher, M., and Castells, F. (2002b), Uncertainty assessment by Monte Carlo simulation in a life-cycle inventory of electricity produced by a waste incinerator, *J. Cleaner Prod.*, 11, 279–292.

STQ (Servei de Tecnologia Química) (1998), Análisis del ciclo de vida de la electricidad producida por la planta de incineración de residuos urbanos de Tarragona, technical report. Universitat Rovira i Virgili, Tarragona, Spain.

U.S. Environmental Protection Agency (1992), Guidelines for exposure assessment; notice, Fed Reg. 57(104): 22888–22938.

Weidema, B. (1998), Multi-user test of the data quality matrix for product life-cycle inventory data, *Int. J. LCA*, 3 (5), 259–265.

Weidema, B. and Wesnaes, M.S. (1996), Data quality management for life-cycle inventories — an example of using data quality indicators, *J. Cleaner Prod.*, 4(3–4), 167–174.

6 Environmental Damage Estimations for Industrial Process Chains*

6.1 INTRODUCTION TO AN INTEGRATED APPROACH

Looking at the picture of life-cycle and risk assessment methods that has been presented, it seems necessary to come to a spatial differentiation of life-cycles in order to facilitate a more integrated way of calculating environmental damage estimations in a chain perspective. In this way the poor accordance between impact potentials and actual impacts can be overcome in life-cycle impact assessment (LCIA) and the results can become more consistent with the risk assessment approach. Thus, another approach is needed that differentiates life-cycles according to the number of processes considered. This means using different levels of sophistication for different applications that are defined by their chain length.

It seems to be unfeasible to estimate environmental damages for each process of a full life-cycle assessment (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 6.1.

However, if the LCA methodology is applied to industrial process chains, i.e., chains with a small number of industrial processes (e.g., <100 processes) — waste treatment process chains, for example — the localization of the processes is often known. 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 6.2. Therefore, for such applications of LCA the main individual processes can be assessed in their corresponding surroundings.

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

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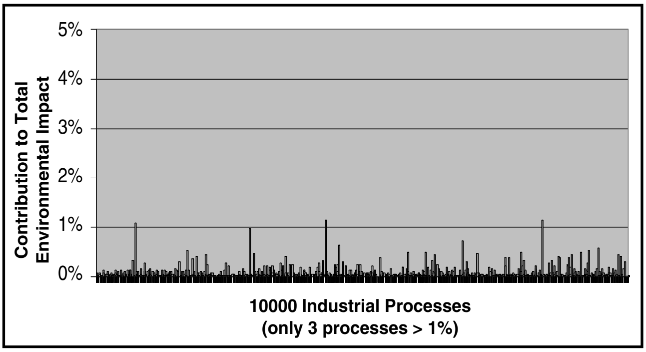


FIGURE 6.1 Full LCA studies of complex products with a huge number of industrial processes, e.g., a computer. In this example fewer than three processes contribute to more than 1% of the total environmental impact

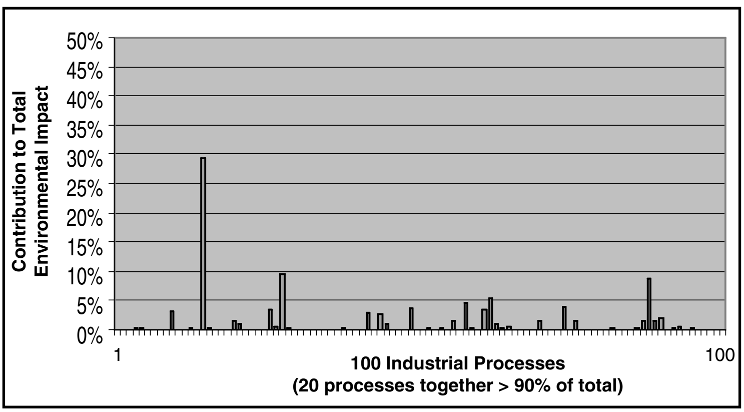


FIGURE 6.2 LCA methodology applied to industrial process chains with a small number of industrial processes, e.g., waste treatment. In this example, 20 processes together contribute to more than 90% of the total environmental impact.

Based on these considerations a methodology has been developed that allows estimating environmental damages for industrial process chains. These chains are understood here as chains of industrial processes with less than 100 processes. A framework is needed that allows evaluating environmental damages as accurately as possible because today's damage assessment methods generate results different from those of the evaluation of potential impacts. This is demonstrated, for instance, in the study undertaken by Spirinckx and Nocker (1999) that compares both approaches. This methodology is useful for certain life-cycle management (LCM) applications such as end-of-life strategies and supply chain management. Possible applications are further discussed in Section 6.6 of this chapter.

6.2 CHALLENGES AND STRATEGY FOR A COMBINED FRAMEWORK OF LIFE-CYCLE AND RISK ASSESSMENT

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 their emission, pollutants are transported through the environment and cause a concentration increase. On the pathway they then can affect sensible receptors, such as humans, and may produce damages. The receptor density clearly depends on local or regional geographic characteristics for nonglobal impact categories. These environmental damages can be evaluated and aggregated according to socioeconomic evaluation patterns as indicators or as 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:

1. Consider each process or at least the main ones.
2. Find a compromise between accuracy and practicability.
3. Apply the damage functions as far as possible to the emissions in their respective continent, region or location.
4. Aggregate the damages by economic evaluation or other forms of weighting to a small number of indicators.
5. Show transparency; analyze uncertainties and sensitivity.

First, a general strategy is necessary with regard to the environmental damage estimations for industrial processes. This strategy includes an approach to make the methodology more practicable. Starting with a conventional life-cycle inventory (LCI), such a strategy can be described as:

1. Creating an algorithm to consider site-specific aspects
2. Calculating the potential impact score
3. Estimating global damages by the best available midpoint indicators
4. Determining main media, pollutants and processes
5. Using fate models to obtain the concentration increment in the respective regions
6. Relating increments with dose– and exposure–response functions and receptors
7. Disposing of methods for aggregation by accepted weighting schemes
8. Relating to other environmental management tools

6.3 COMPARISON OF ENVIRONMENTAL RISK ASSESSMENT AND LIFE-CYCLE ASSESSMENT

Before presenting the methodology, in this section a comparison of LCA and environmental risk assessment (ERA), the two environmental tools that are further

TABLE 6.1
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 optimization by risk minimization	Environmental optimization 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	Characterization factor
Result	Probability of hazard	Environmental effect score

integrated, is outlined in Table 6.1. The comparison is illustrated by the example of electricity generated from coal and produced in the same way but in two different regions, in which the combustion of coal is obviously an important part of the life-cycle:

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

According to Sonnemann et al. (1999), the LCA will probably state the minimal total emissions and energy demand for Case 1 due to the importance of the additional transport and the negligence of the specific region. By contrast, the ERA will state the minimal risk to the environment for Case 2 because the focus is put only on the main process within 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 6.1. It also clearly demonstrates the need for a more integrated approach that does not so easily allow two environmental impact analysis tools to provide such contradictory and inconsistent results.

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

6.4 MATHEMATICAL FOUNDATION

Because LCA and ERA belong to the environmental system analysis and are based on models of the real world, it is evident that the first step is to base the general strategy described so far on a mathematical framework that enables carrying out further steps. It includes, in particular, a procedure that allows knowing the quantity, situation and moment of the generation of the specific environmental interventions in the LCI. 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; then different impact potentials can be calculated by the characterization 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 $\tilde{\mathbf{e}}_\mathbf{v}$, as shown in Expression 6.1, where e_v^i is the specific [EL (environmental load)/kg] of the ELi, $\tilde{e}_v^i$ is the specific weighted [EL_{eq}/kg] of the EL_i, and λ^i is the specific weighting factor of the EL_i.

$$\tilde{e}_v = \tilde{e}_v^i = e_v^i \quad \lambda^i \tag{6.1}$$

According to the goal of the general strategy, the eco-vector algorithm must be changed in order to make possible the assessment of the actual impacts caused by a specific process in a 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, PM_{10}, Cd, etc.) per functional unit to obtain the functional unit, the product. When the LCI analysis is applied to the three processes, an eco-vector with the environmental interventions for all three processes is obtained, as illustrated in Figure 6.3. It is evident that the following LCIA generates only potential impacts because all site-specific information is lost when the LCIA is carried out.

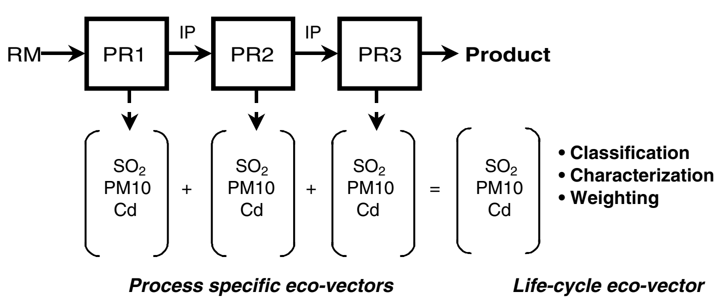


FIGURE 6.3 Life-cycle inventory analysis according to the eco-vector principle (IP = intermediate product, PR = process, RM = raw material). (Reprinted from *J. Hazardous Mater.,* 77, Sonnemann, G.W. et al., pp. 91–106, ©2000 with permission from Elsevier.)

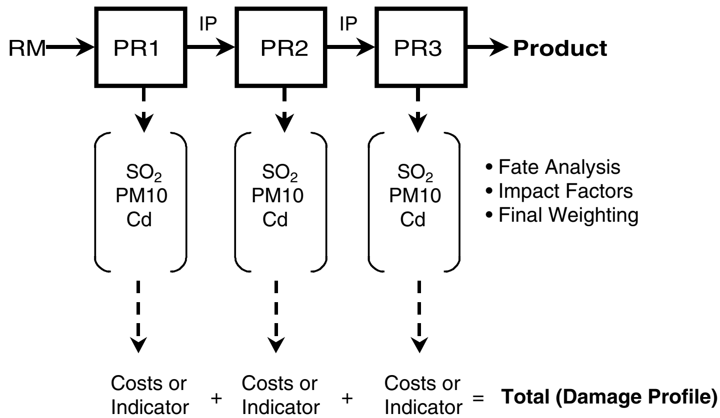
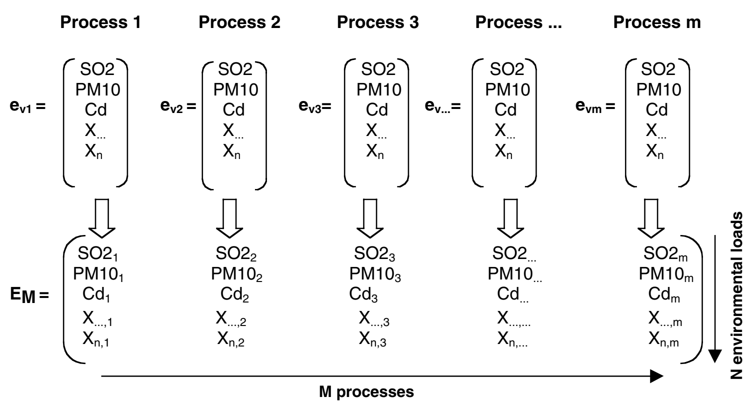


FIGURE 6.4 Determination of damage estimations by site-specific assessment (IP = intermediate product, PR = process, RM = raw material). (Reprinted from *J. Hazardous Mater.,* 77, Sonnemann, G.W. et al., pp. 91–106, ©2000 with permission from Elsevier.)

Another approach to performing the environmental assessment of a functional unit consists in analyzing the actual impacts of each process according to a site-specific damage estimation concept (Figure 6.4). Consequently, the environmental loads of each process are also accounted for, 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.

In order to express this different method in an algorithm based on the same principles as introduced before, the eco-vector must be transformed into an eco-technology matrix $\mathbf{E_M}$. In this matrix, which is similar to the technology matrix mentioned by Heijungs (1998), there are M columns for M linear processes, and N rows for the N environmental loads as those for the eco-vector. The example has three columns for the three processes and three rows for the three environmental loads, SO_2, PM_{10} and Cd. (See Expression 6.2 for an illustration.) It is evident that



the environmental loads must be in the same order for all the processes and that a certain environmental intervention must be expressed always in the same unit.

By 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, impact factors and final weighting. This matrix assigns the damage cost or another damage indicator caused by one specific EL_i to each process. In $\mathbf{W_M}$ there are N columns for the N environmental loads and M rows for the M linear processes. In this way an M × M matrix is obtained, the weighted eco-technology matrix $\mathbf{WE_M}$, as shown in the following expression:

$$\mathbf{W_M}.\ \mathbf{E_M} = \mathbf{WE_M} \tag{6.3}$$

The main magnitude will be the trace D of $\mathbf{WE_M}$ (Expression 6.4), 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 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.

$$\begin{aligned} D = \mathrm{tr}\,(\mathbf{WE_M}) &= \sum_{ji} (W_M^{ji}.\ E_M^{ij}) = \sum_{j}^{M} [\sum_{i}^{N} W_M^{ji}.\ E_M^{ij})] \\ &= \sum_{j}^{M} D^{jj}\ \lambda \sum_{j}^{M} D^{j} \end{aligned} \tag{6.4}$$

The algorithm is illustrated in Expression 6.5 for the example of three processes and two environmental loads SO_2 and Cd.

$$\begin{array}{cc} & \xrightarrow{\quad N \quad} \\ M \Big\downarrow & \begin{pmatrix} W_M^{1,\,SO2} & W_M^{1,\,Cd} \\ W_M^{2,\,SO2} & W_M^{2,\,Cd} \\ W_M^{3,\,SO2} & W_M^{3,\,Cd} \end{pmatrix} \end{array} \bullet \begin{array}{cc} & \xrightarrow{\quad M \quad} \\ N \Big\downarrow & \begin{pmatrix} E_M^{SO2,\,1} & E_M^{SO2,\,2} & E_M^{SO2,\,3} \\ E_M^{Cd,\,1} & E_M^{Cd,\,2} & E_M^{Cd,\,3} \end{pmatrix} \end{array} = \begin{array}{cc} & \xrightarrow{\quad M \quad} \\ M \Big\downarrow & \begin{pmatrix} D^1 & D^{12} & D^{13} \\ D^{21} & D^2 & D^{23} \\ D^{31} & D^{32} & D^3 \end{pmatrix} \end{array}$$

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

The values of the weighted eco-matrix $\mathbf{WE_M}$ in the diagonals D^1, D^2 and D^3 represent the environmental damage cost or damage indicator of processes 1, 2 and 3. If the values of $\mathbf{WE_M}$ are not in the diagonal, such 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 6.3 and in Expression 6.6 is given the abbreviated mathematical way of expressing the contents, where k stands for the region k and j for the process j:

$$D^{kj} = \sum_{i}^{N} (W_M{}^{ki} . E_M{}^{ij}) \tag{6.6}$$

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 global impacts such as for the GWP, where this simplification allows working with one eco-vector for all the processes and one weighting vector for all regions, as shown in Expression 6.7. In the case of global weighting factors, the weighting matrix has the same components for all processes.

$$W_M{}^{ji} = W_M{}^{ki} \;\; < = = > \text{region (j)} = \text{region (k)} \tag{6.7}$$

A special topic that must be considered is the question of mobile processes. If the process is a moving one, different regions may be involved so that Expression 6.8 holds true. By choosing the size of the region, the number of regions to consider for the corresponding mobile process is determined. If there is a mobile process with sufficient transport kilometers,

$$= = > \text{exists at least 1 i; } D^{kj} \neq D^{lj} \; V \; k \neq i \tag{6.8}$$

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.

6.5 OUTLINE OF THE COMBINED FRAMEWORK

6.5.1 Overview of the Methodology

With the mathematical framework at our disposal, the next task 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 media, processes and pollutants must 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 scores because each considers one particular type of environmental impact or weighting scheme. Then the relevant processes and pollutants obtained are spatially differentiated according to the site or region that should be taken into account for 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 and exposure and consequence analysis are carried out with different levels of detail for each process identified as relevant. The results of the fate and 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 and exposure and consequence analysis with different levels of detail. Thus, these indicators can be used directly in the damage-assigning matrix. In particular, the GWP and the ozone depletion potential (ODP) are considered global indicators. According to Bare et al. (2000) and Udo de Haes and Lindeijer (2001), these potentials are important for the subarea of protection life support systems, which belong to the area of protection (AoP) of the natural environment and might be seen as having intrinsic value in their own right. The life support functions concern the major regulating functions that enable life on Earth (human and nonhuman) — particularly, regulation of the Earth's climate, hydrological cycles, soil fertility and 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 important.

The flowchart in Figure 6.5 gives an overview of the procedure to generate the eco-technology and the damage-assigning matrices. The multiplication of the matrices yields the damage profile of each considered alternative, as well as interesting information for the optimization of process settings. In the case of using different impact scores, the same damage endpoints considered in the fate and exposure and consequence analysis related to these scores must be summed up.

The damage profile can be divided into damages to human health (mortality, cancer and morbidity), manmade environment, natural environment and global indicators (GWP and others). The application of a weighting and aggregation scheme, determined in the goal and scope definition, avoids a multicriteria analysis for a huge amount of impact parameters, for instance, emergency room visit, asthma attack, maintenance surface for paint, and yield loss of wheat, which are results of site-specific environmental evaluations. Next, each of the steps developed in Figure 6.5 will be explained in more detail to provide the user with a better understanding.

6.5.2 Goal and Scope Definition

In the goal and scope definition (Figure 6.6) the decision-maker determines the cornerstones of the environmental damage estimations for industrial process chains that, in his opinion, are best fit to answer the environmental management problem

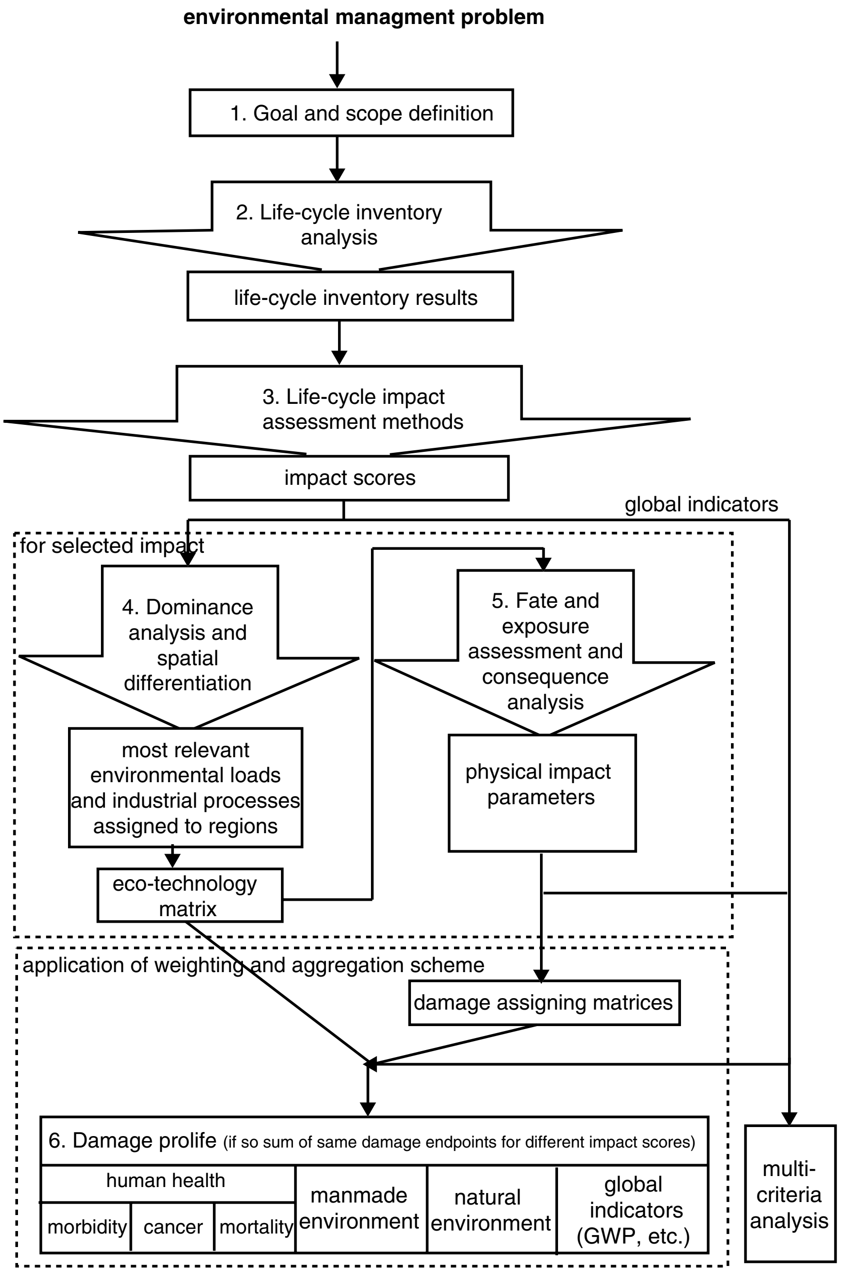


FIGURE 6.5 Overview of the procedure to generate the eco-technology and damage-assigning matrices.

of interest. Of course he must do this while taking into account budget restrictions. In the goal definition it must be decided which situations or scenarios will be assessed and compared. Here *situations* refer to existing process chains while *scenarios* mean process chain options for the future (Pesonen et al., 2000).

In the scope definition, the decision-maker must select the functional unit (e.g., 1 TJ electricity or 1 kg treated waste), the initial system boundaries (e.g., 1% contribution to functional unit), the LCIA characterization potentials (e.g., GWP, AP, NP, etc.) and/or single index methods or endpoint-orientated methods (e.g., eco-indicator 95 or 99, EPS, etc.). These requirements correspond to those for LCA 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. Deciding if an uncertainty analysis should be included, if accidents should be considered and if the eco-efficiency of the process chain should be calculated is optional.

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

The methodology is based on the principle of transparency in the way the results are obtained. The format in which the results are desired, such as monetary values or physical impacts (e.g., cases of cancer), determines this. Therefore, the subjective elements, in particular the different parts of the weighting step, are assembled in the goal and scope definition before carrying out any analysis. For the weighting, decision-makers can follow the general decision tree presented in Figure 6.7. There are different options to evaluate the ELs, whose choice depends on the worldview of the decision-maker. Thus, the methodology avoids implicit decision-making common in endpoint-orientated LCIA methods.

For environmental loads that cause a GWP with global impact, ODP and other global indicators can be calculated. In such a case these potentials have environmental relevance in the form of life-support functions and depletion of natural resources. First, decision-makers must select the environmental impacts they consider relevant for the environmental management problem under study; then they must decide, according to available knowledge, if the damages related to these potentials may be estimated. If they think that these damages cannot be estimated with acceptable reliability, then, for each global indicator, they must decide if they prefer to monetize the potential impact using abatement costs or to express it directly as a physical impact potential, e.g., CO_2 equivalent. These can be assessed in conjunction with the other environmental loads if they believe the damages to be estimable (see Figure 6.7).

Other environmental loads may cause local and regional damages, which can be divided into the AoPs published by Udo de Haes et al. (1999): manmade environment, natural environment and human health. In the cases of manmade environment and natural environment, the questions on which to decide are the same. Due

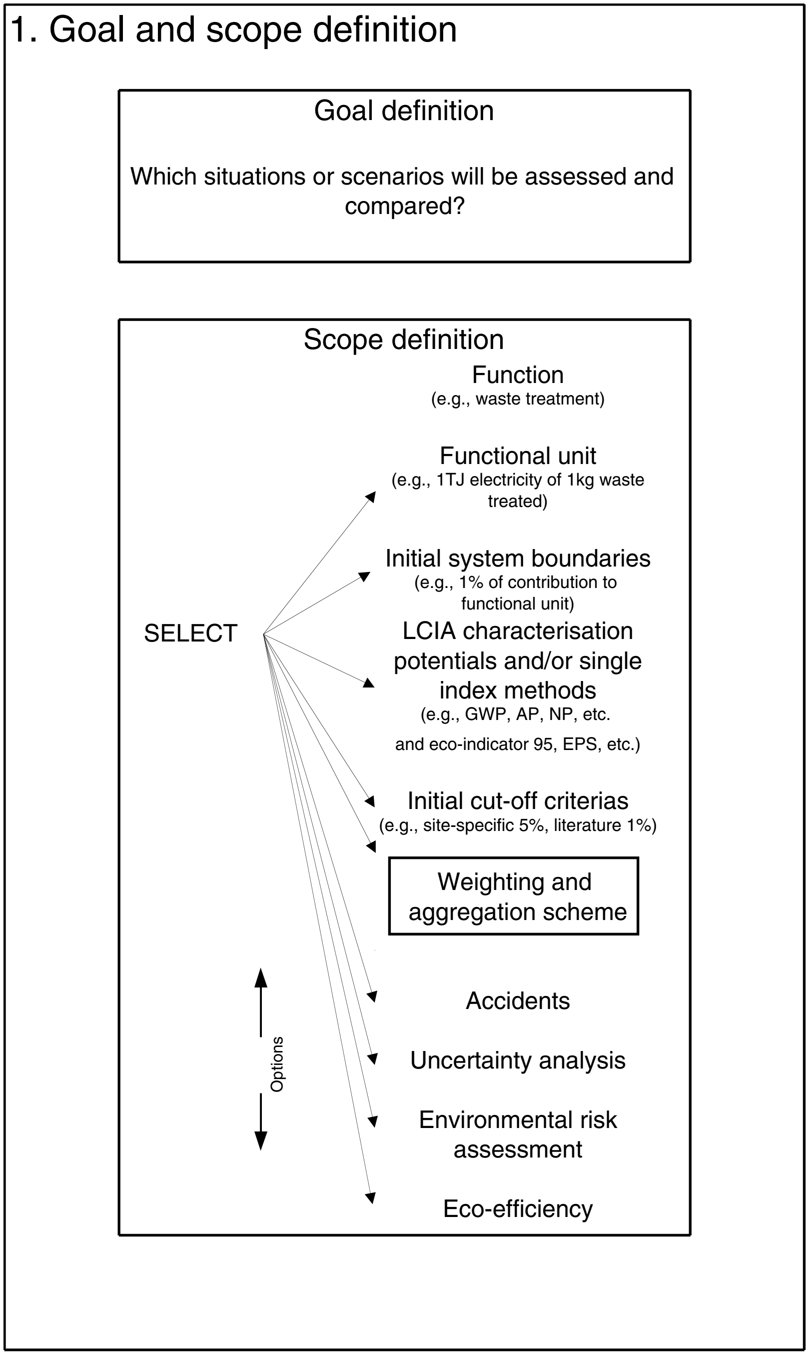


FIGURE 6.6 Goal and scope definition.

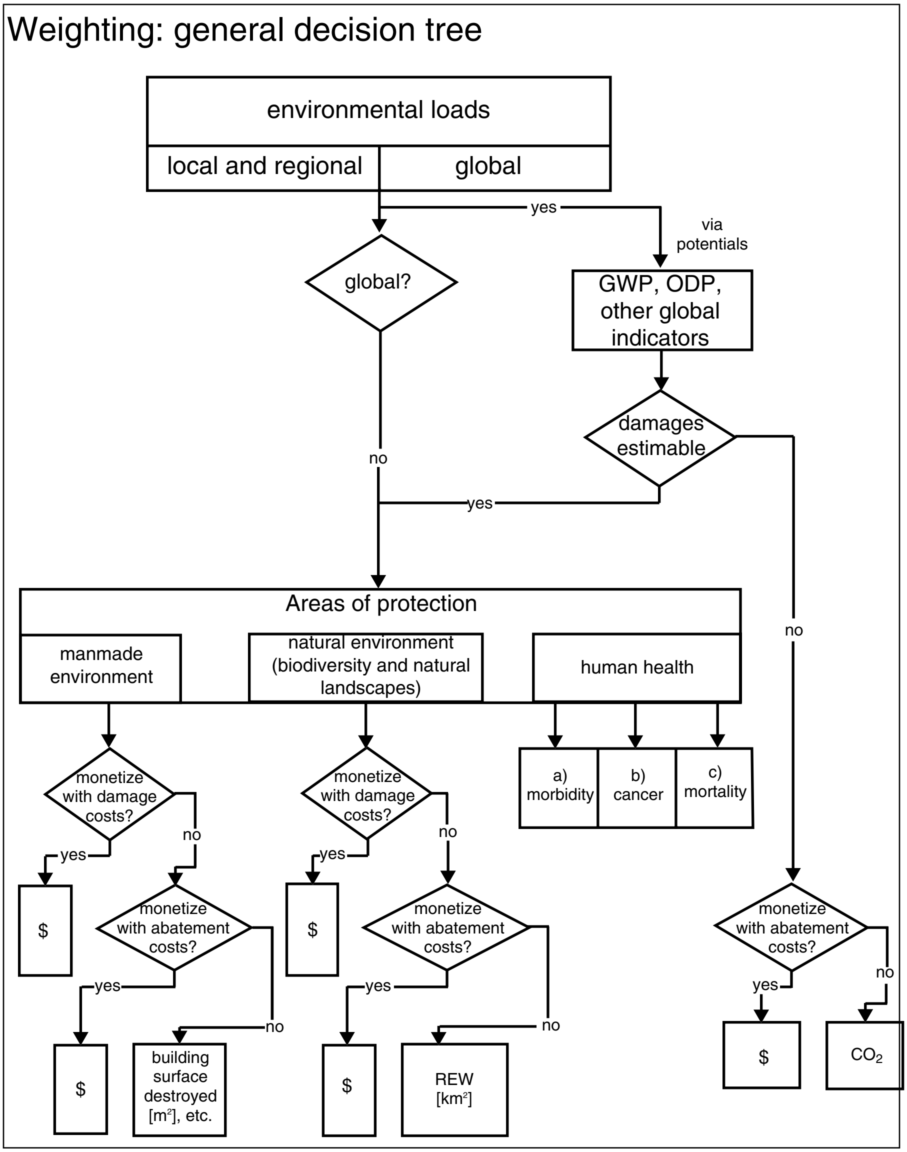


FIGURE 6.7 General decision tree for weighting.

to the complexity of the weighting options for the human health AoP, another decision tree has been drawn and is presented in Figure 6.8.

For the manmade environment and natural environment AoPs, the first question is whether the damage should be monetized in the form of environmental external costs, e.g., according to the ExternE approach (EC 1995). If decision-makers do not like this type of weighting, they must decide if they prefer to monetize the impacts using abatement costs or to express them directly as a physical impact parameter

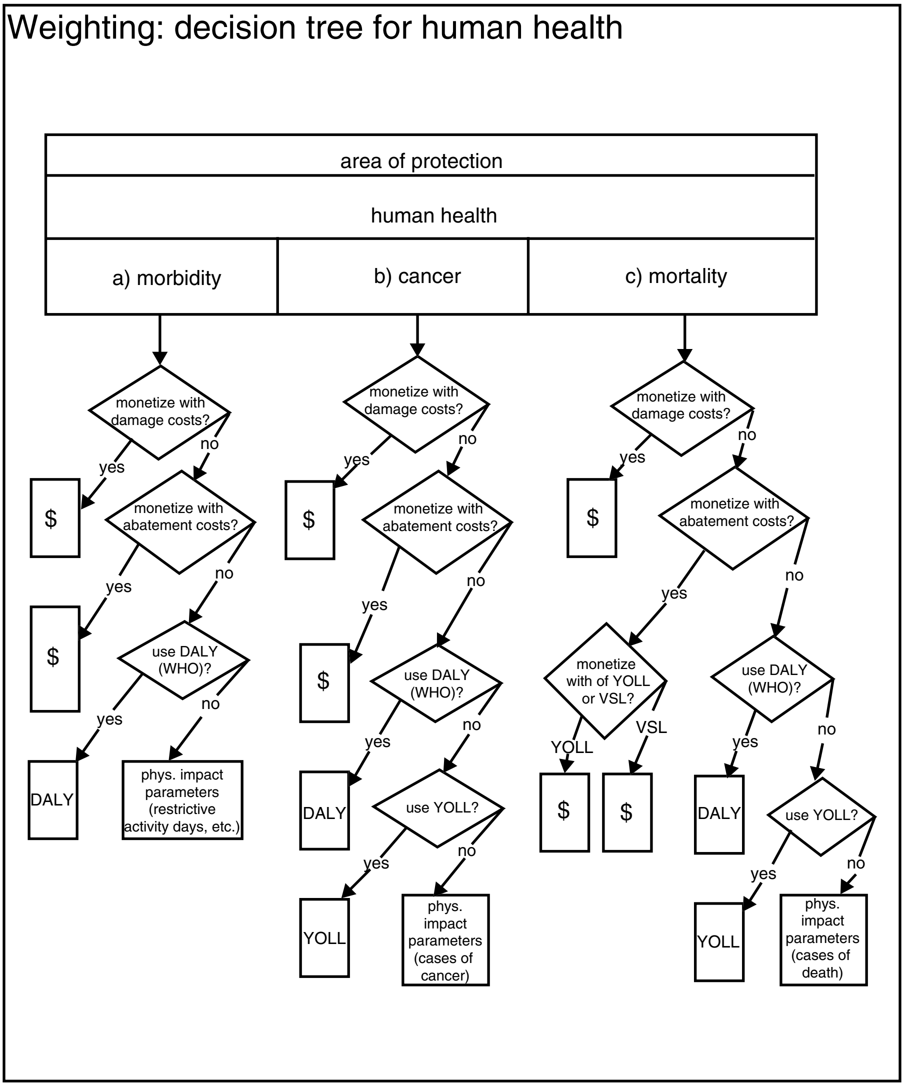


FIGURE 6.8 Decision tree for weighting for the AoP of human health.

— in the case of manmade environment, for example, in maintenance surface for paint (m^2) and yield loss of wheat (t) or for the natural environment in the REW in (km^2) as described in a previous chapter.

For all the damages to human health, the first questions concern the same decisions as for the other AoPs: whether to choose monetization and, if so, monetization by damage or external environmental costs, abatement costs or internal environmental costs. In the case of fatal effects, it must also be decided if the monetization of the damages should be done based on years of lost life (YOLL) or directly on value of statistical life (VSL). Additionally, due to the existence of internationally accepted damage indicators by the World Health Organization (WHO) in the form of disability adjusted life years (DALY) (Murray and Lopez, 1996; Hofstetter, 1998) and YOLL (Meyerhofer et al., 1998), other types of weighting across the different damage endpoints are available.

Thus, in the case of damages that cause morbidity, the decision-maker must select among the assessment by DALY or physical impact parameters, for instance, emergency room visits, asthma attacks, restrictive activity days, etc. For cancer, a selection must 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, individual risk can be evaluated. These different weighting options are summarized for the decision-maker in the first table which forms part of Figure 6.9, named *weighting of impacts*, in which all together four decision tables relevant for weighting and aggregation are presented. Two other tables concern the discount rate for monetization and the cultural theory for DALY, and the last one concerns decisions related to the aggregation of damages. In this figure, default selections are presented as an illustration of a typical case for study of environmental damage estimations in industrial process chains.

In the table for the weighting of impacts, one entry must be made for each damage class. These damage classes are the manmade environment, natural environment and human health AoPs, as well as all the so-called global indicators such as GWP and ODP that could be related to the life support functions and resources sub-AoPs if resource depletion is considered an environmental problem (see Figure 6.9).

Human health is divided into morbidity, cancer and mortality. In principle, the monetization 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. Although no more than the preferred option must be selected for monetization and DALY, in the case of using physical impact parameters the selected parameters should be mentioned here. For the monetization a discount rate must 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 must be selected. According to Hofstetter (1998), three archetypes represent human socioeconomic perceptions quite well: hierarchist, egalitarian and individualist.

Finally, it must 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, two options for aggregation exist. One option is to aggregate directly in the damage matrix, called intermediate aggregation, which is less laborious due to fewer matrix operations. The other option is to undertake a final aggregation reducing the number of components in the damage profile, which makes the steps more transparent but risks confusion. In any case, the final result will be the same. Also, according to certain criteria, groups (for instance, AoPs), can be created to show unity.

Weighting and aggregation: decision tables

Weighting of impacts

(for each damage class you need one entry)

damage class (areas of protection, global indicator)		monetization: damages	monetization: abatement	usage of DALY or YOLL	usage of physical impact parameters
manmade environment		yes			
human health	morbidity	yes			
	cancer	yes			
	mortality	yes			
natural environment (biodiversity and natural landscapes)				X	REW
GWP and other global indicator				X	GWP

select one		
0%	3%	10%
	yes	

select one		
hierarchist	egalitarian	individualist

damage class (areas of protection, global indicator)		intermediate aggregation in the damage-assigning matrix	final aggregation in the damage profile
		create	*groups to show unity*
manmade environment		A	
human health	morbidity	A	
	cancer	A	
	mortality	A	
natural environment (biodiversity and natural landscapes)			
GWP, ODP and other global indicators			

FIGURE 6.9 Decision tables for weighting and aggregation.

It is generally accepted that damages to the manmade environment can be best evaluated by external environmental costs; however, in the default selections, it has been decided that this can also be done for damages to the AoP human health. The natural environment will be assessed by REW (relative exceedance weighted) and as a global indicator only the GWP is chosen. The default discount rate is 3%. An intermediate aggregation is selected for damages to the manmade environment and to human health.

Apart from the described selection of clear weighting schemes in order to obtain meaningful indicators, it must be acknowledged that, in principle, determining 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– and exposure–response functions) can have very important influences on the final result. Thus transparency on this point is recommended as well as checking 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 in this chapter through a case study that has demonstrating the functionality of the weighting and aggregation scheme as one of its primary interests.

6.5.3 Life-Cycle Inventory Analysis

After the goal and scope definition, the life-cycle inventory analysis follows in the same way as in an LCA according to ISO 14040. An overview of the LCI analysis with its options is given in Figure 6.10.

If a situation of an existing process chain is assessed, the measured data (ELs) from the core processes and those obtained from up- and downstream 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 future scenario for process chain options will be assessed, data can be generated by a model of core processes and linear adaptations of current data obtained from up- and downstream processes. The model can be a modular model, as described in Chapter 1, or a process simulator.

If the consideration of accidents was chosen in the goal and scope definition, potential environmental loads through the accidents must be generated by simulations with the corresponding analysis of the undesired events or accidents (AICHE, 1985; Aelion et al., 1995), as mentioned in Chapter 1. The proper LCI can be created 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. The incorporated database is important in the LCI model or software, especially for background processes like electricity production (e.g., Frischknecht et al. 1996). Another optional element is the uncertainty analysis, which can be carried out, for instance, by Monte Carlo (MC) simulation, as described in Chapter 5. By using probability distributions for the essential factors in an MC simulation (LaGrega, 1994), the inventory result can be transformed from a concrete value into a probability distribution around a mean value.

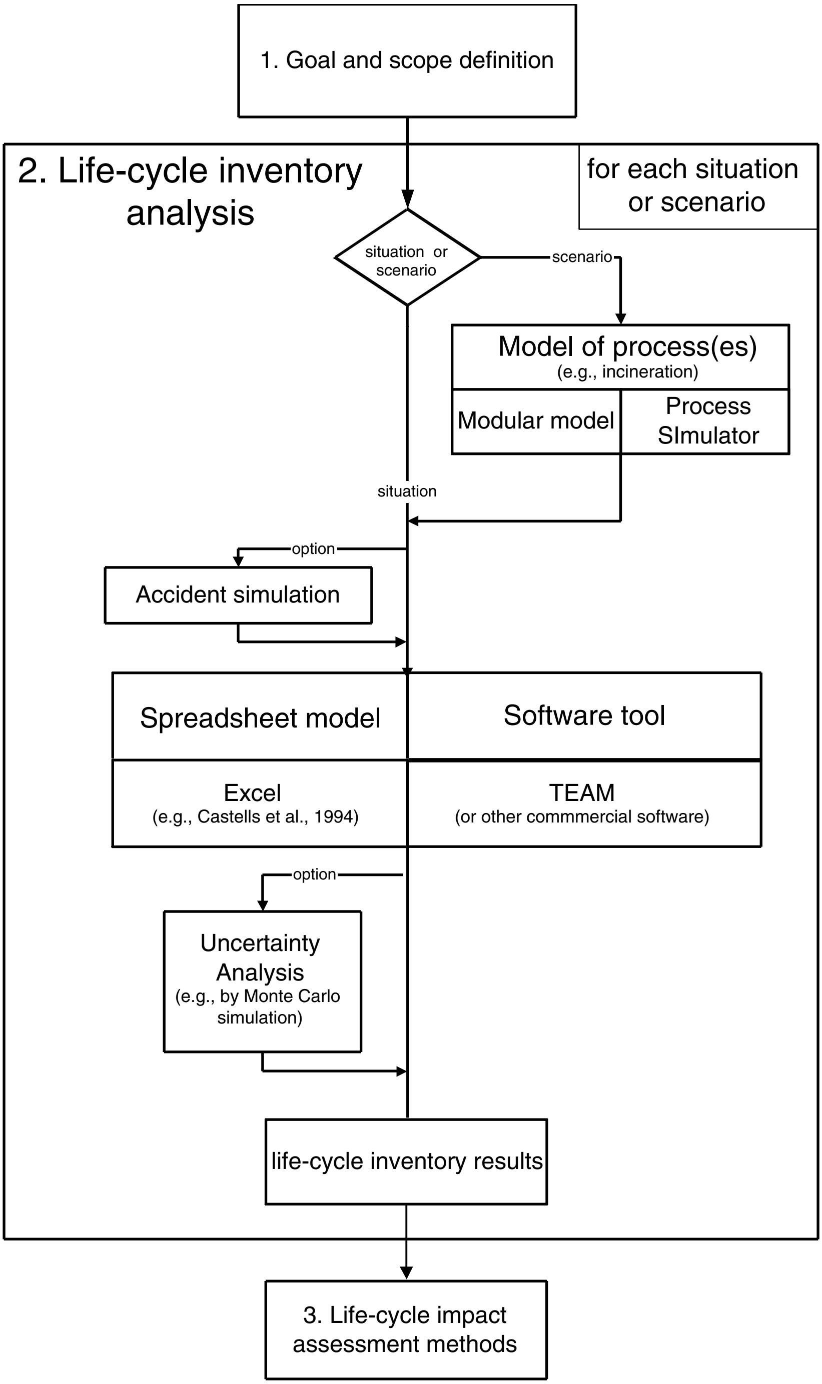


FIGURE 6.10 Life-cycle inventory analysis.

In principle, the proposed methodology does not need a complete LCI analysis with ISO 14041 as a basis. What it really needs is life-cycle inventory data about the process chain under study. These data can also be obtained by streamlined LCAs or simplified LCI approaches (Curran and Young, 1996). An important part of such methods is an iterative screening procedure (Fleischer and Schmidt 1997), which includes elements similar to those used in the methodology presented in this study by 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 during data collection (see Figure 6.10).

6.5.4 Life-Cycle Impact Assessment Methods

In the 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 methods were selected. In Figure 6.11, an overview of the usage of life-cycle impact assessment methods is given; the main options are shown schematically:

- 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 Chapter 3.

The global indicators selected in the weighting scheme are considered separately. They are obtained in the characterization step in both options in which midpoints are calculated. Each global indicator feeds directly into the damage profile. If required, they are first monetized by abatement costs.

The midpoint-related LCIA methods allow calculating the environmental potential of the respective impact category in the characterization step. All presented LCIA methods except the midpoint potentials permit obtaining a single index to measure the environmental impact performance. The midpoint-based weighting methods require carrying out normalization and then a weighting step. Direct weighting methods omit the characterization and the normalization step. As endpoint-orientated method, eco-indicator 99 (see Chapter 3 for further details) does not contain explicitly midpoint results.

The results of the LCIA methods are called impact scores in Figure 6.11. These scores allow comparing 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 in more detail the environmental damages of the main processes and pollutants in the studied process chain.

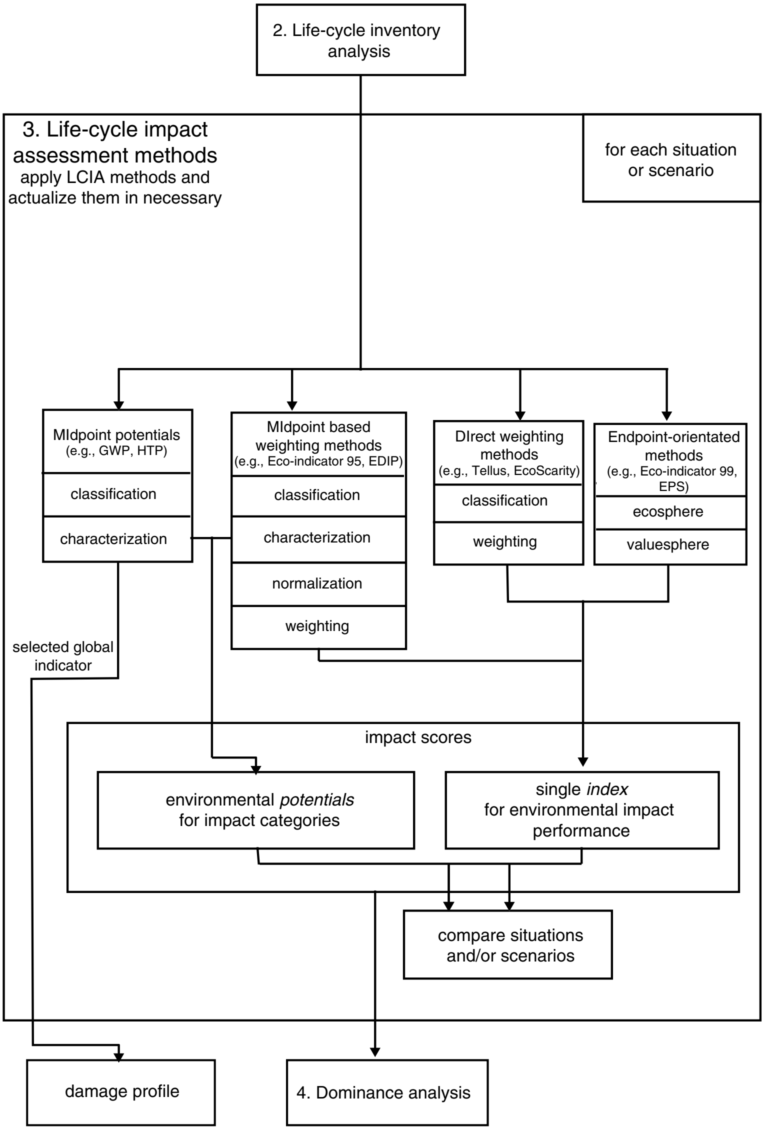


FIGURE 6.11 Usage of life-cycle impact assessment methods.

6.5.5 Dominance Analysis and Spatial Differentiation

Before determining the predominant processes and pollutants of the studied process chain for each selected impact score, it must be determined which media are primarily affected by the emissions of the process chain. Afterwards, the environmental loads and processes must be spatially differentiated. Therefore, in Figure 6.12, the dominance analysis for media and the spatial differentiation is presented together with a general overview of this selection procedure.

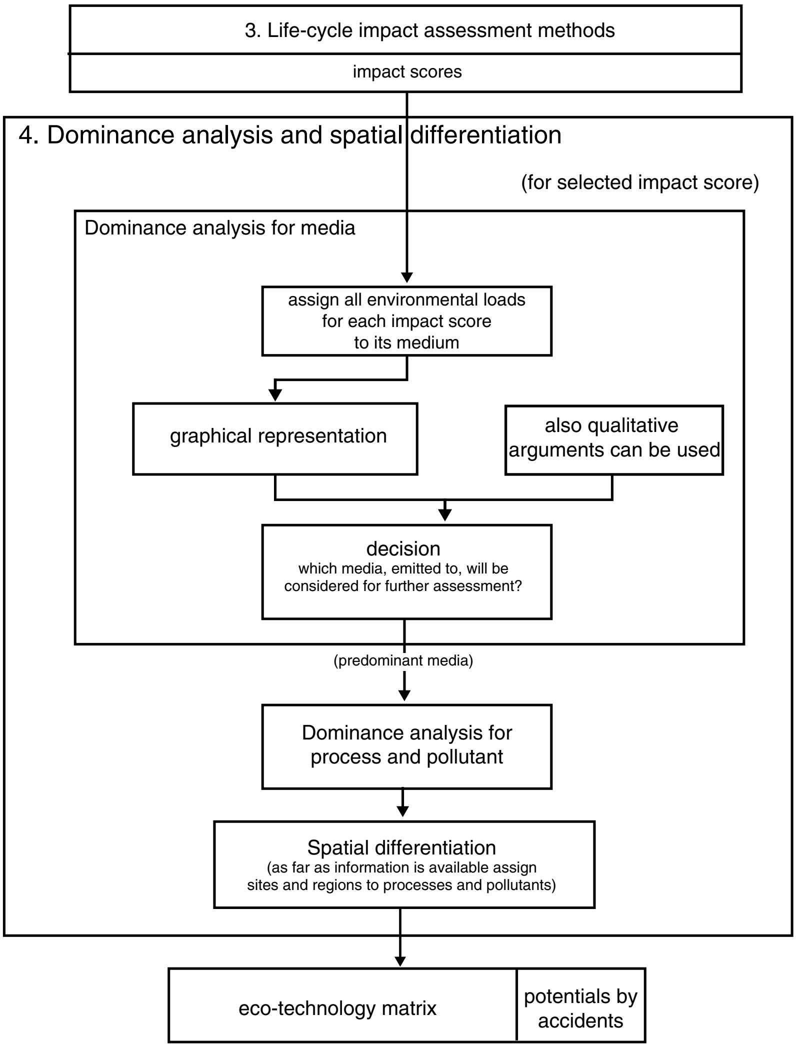


FIGURE 6.12 Dominance analysis for media and spatial differentiation.

In the dominance analysis for media (i.e., air, water and soil), all emissions considered in the selected impact score are assigned to their media. Then the contribution of each medium to the total impact score is presented graphically. Finally, a decision must be made about which of the media emitted to will be considered for further assessment. In this decision qualitative arguments can also 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 environmental loads (Figure 6.13). First, the percentage of the total of the selected impact score is calculated for each process and pollutant, then the defined initial cut-off criteria are applied. For processes and pollutant, respectively, the percentages and cut-off criteria are represented graphically to visualize the data. The graphical presentations may suggest redefining the respective cut-off criteria in order to obtain a manageable number of processes that can be assessed in detail. In principle, this is an iterative procedure to find the optimum. The decision-maker decides whether one or more impact scores are to be considered and in which relation to each other. This means also that 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 selection of processes, the processes to be assessed by site-specific and site-dependent factors (generally obtained by project-related impact assessment studies) and those to be assessed in a process-specific and/or region-dependent way (generally by values published in the literature) must be determined.

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 ways) are determined. Qualitative arguments can also be used in this decision and, in a sustainable perspective, social and economic aspects are important. Therefore, it seems evident that, for instance, PCDD/Fs must be considered in a case study on waste incineration due to their relevance for discussion in society, although their percentage of the total impact score is less than the lowest selected cut-off criterion (See Figure 6.13).

Finally, the identified predominant pollutants and regions must be assigned to sites or regions. How far this is possible depends particularly on the information available about the location of a specific process. Thus, it must be taken into account that the site might be unknown; in this case, only the approximate global region can be assigned, but not the specific site. The spatial scale of the pollutants depends on their residence time in the respective medium. Many 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, the LCI data for the electricity mix of a country generally are taken.

A problem that occurs in the assignment of sites is that often it is not the site that most influences the environmental damages, but the emission height. This can be concluded from the results for the site-dependent impact factors obtained in Chapter 7. Therefore, instead of regions, a differentiation is made essentially according to classes that have similar characteristics with regard to the emission situation (determined by the geographic site and the stack height). However, in this methodology they are called regions because this term illustrates the idea behind spatial differentiation much better.

Consistent with Expression 6.7, the world is the corresponding region for pollutants that cause a global environmental impact like CO_2. In agreement with Expression 6.8, in the case of mobile processes, i.e., transports, it must be decided if the environmental loads can really be assigned to one region only or must be differentiated among two or more regions if the distance is long enough. The determined processes assigned to sites and regions are the M processes and the chosen pollutants

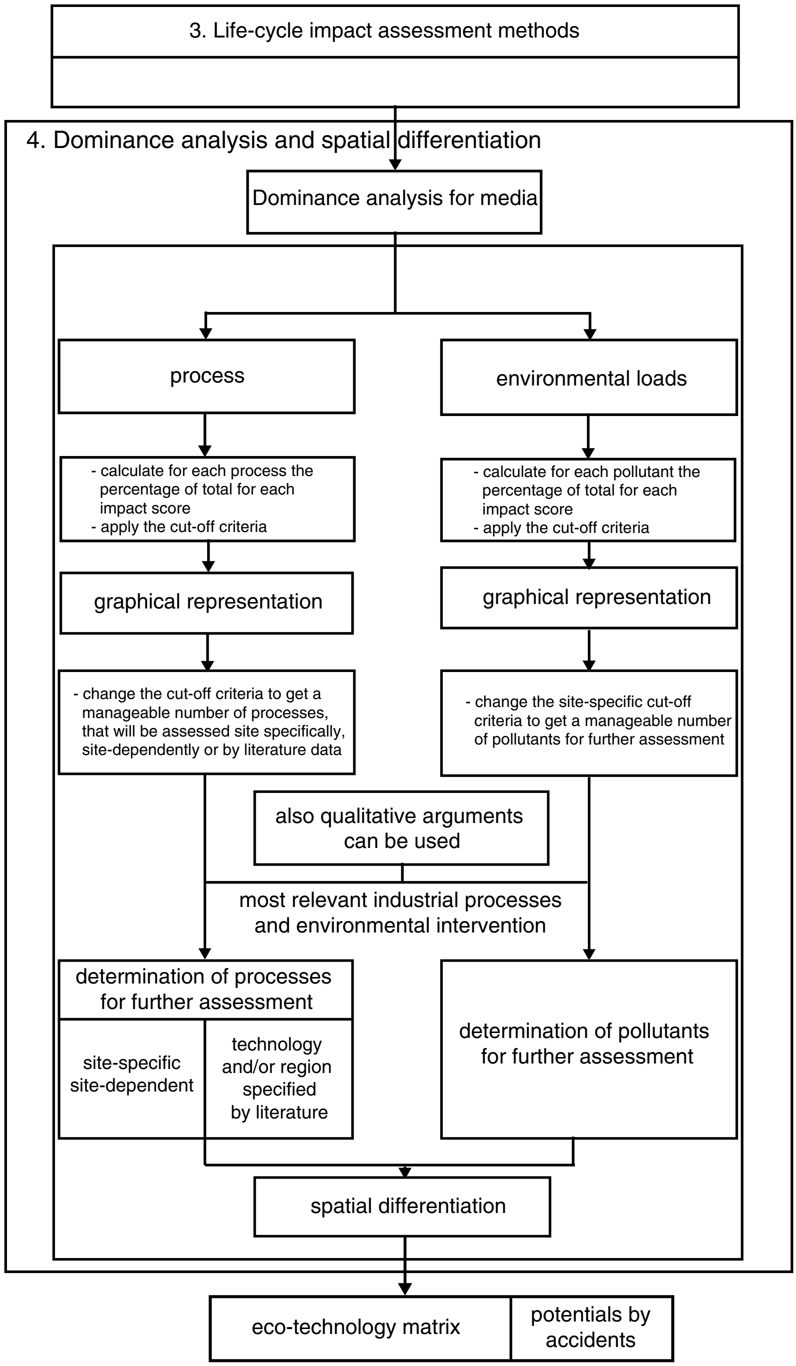


FIGURE 6.13 Dominance analysis for processes and pollutants

are the N pollutants of the eco-technology matrix. The eco-matrix may always contain a part of potential environmental loads if accidents have been considered.

6.5.6 Fate and Exposure Assessment and Consequence Analysis

The level of detail in the fate and 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 on a site-specific basis, 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 based on 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 explained in Chapter 7. However, it still must be considered as an approach in development.

Figure 6.14 gives an overview of the fate and exposure and consequence analysis with the different levels of detail. The results of this analysis are the basis for the damage-assigning matrices.

For the processes identified as 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 must be treated as those processes are that have been determined to be evaluated process- and/or region specified by literature values. If the site is known, the data about the emissions in the LCI must be divided into upstream-related data, which must be evaluated by literature values, and the foreground process-related data 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, in each case the kind of emission (continuous or one time) for a site-specific assessment must be checked. An example of a site-specific assessment of continuous emissions (the ERA and the IPA) carried out for Tarragona's MSWI as described in Chapters 4 and 5. An example of a one-time emission is that from an explosion. 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) in the VWM (see Chapter 3). Such a transfer is the use of results of the site-specific impact assessment of Tarragona's MSWI for another process within the region, e.g., for the ash treatment operation situated in Constanti, a few kilometers away from the municipality of Tarragona.

If site-dependent impact assessment factors according to the approach outlined in Chapter 7 are available, then these factors allow estimating 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) because it can be considered to be a number of industrial processes that take place (at one time after the other) in different regions. Through site-specific and site-dependent impact assessments, physical impact parameters are obtained that can be converted into

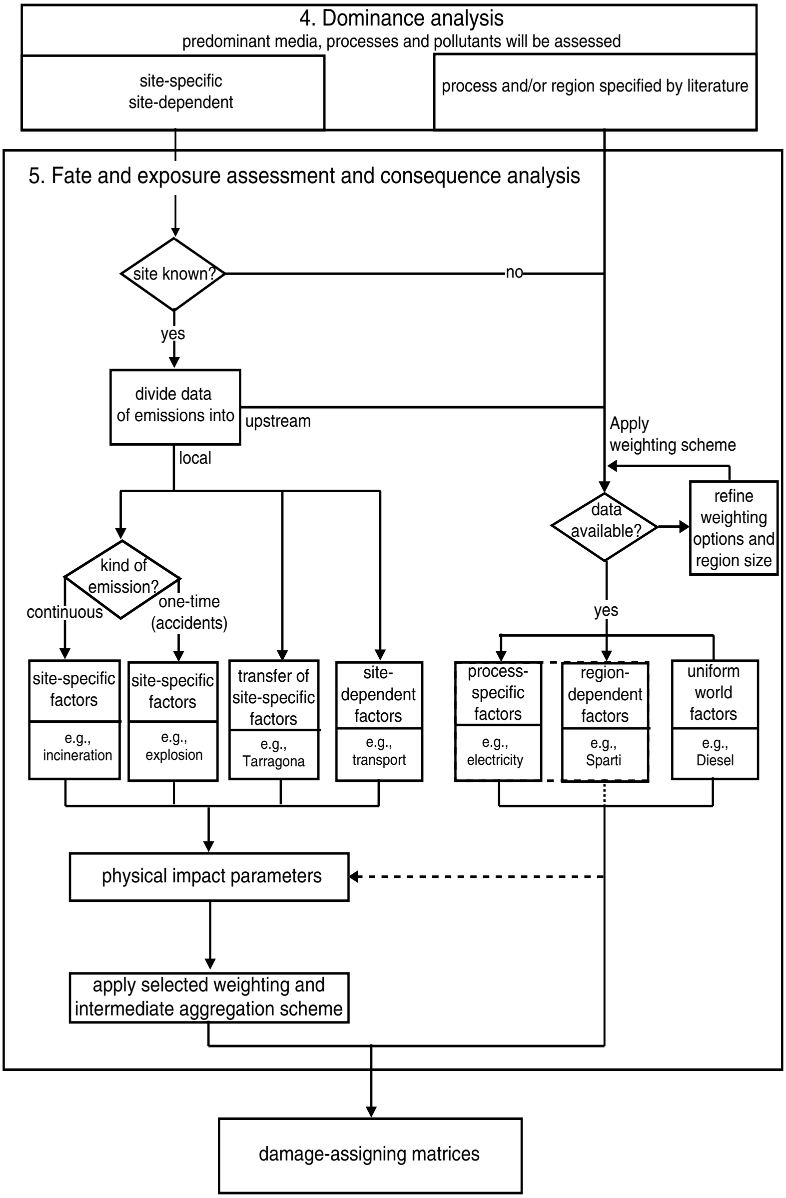


FIGURE 6.14 Fate and exposure assessment and consequence analysis.

indicators or environmental costs by applying the selected weighting scheme. Moreover, if laid out in the goal and scope definition, the intermediate aggregation can take place.

For processes in which literature values should be used, the first action is to check if the desired data are available in the literature. If this is not the case, the weighting options of the scope definition and/or the size of the region must be redefined. Because the assessment of site- and region-dependent damage endpoints is an issue that more or less started in the mid-1990s, not much data have been published. Therefore, it is quite possible that determined indicators or cost types are not available in the literature for the processes and pollutants of certain regions.

If data are available, then whether a classification is possible according to technology and/or region must be determined for each process. 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 regions of Spain. Another example is 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 must be applied. For instance, Rabl et al. (1998) have published a uniform world model for air emissions. Diesel production and the related process chain that takes place all over the world is an example of a process difficult to classify.

Depending on the selected weighting scheme and the available data in the literature, physical impact parameters, 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 6.15 for the medium of air; in principle, these can also be applied to other media. For example, Schulze (2001) presents site-orientated impact assessments for the medium of water in relation to LCAs for detergents, using the integrated assessment model GREA-TER in an adapted version 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 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 and exposure analysis can be carried out in a generic or detailed way. The generic way uses an integrated impact assessment model, e.g., EcoSense (described in Chapter 4). 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 also includes 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 limited, e.g., the resolution of population densities is not as detailed as it could be when using a geographic information system.

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). An independent Gaussian dispersion model (see Chapter 4) is applied (e.g., ISCST-3 in BEEST) for the transport near the emission point and more detailed geographic data like those in ERA provided by a geographic information system are employed.

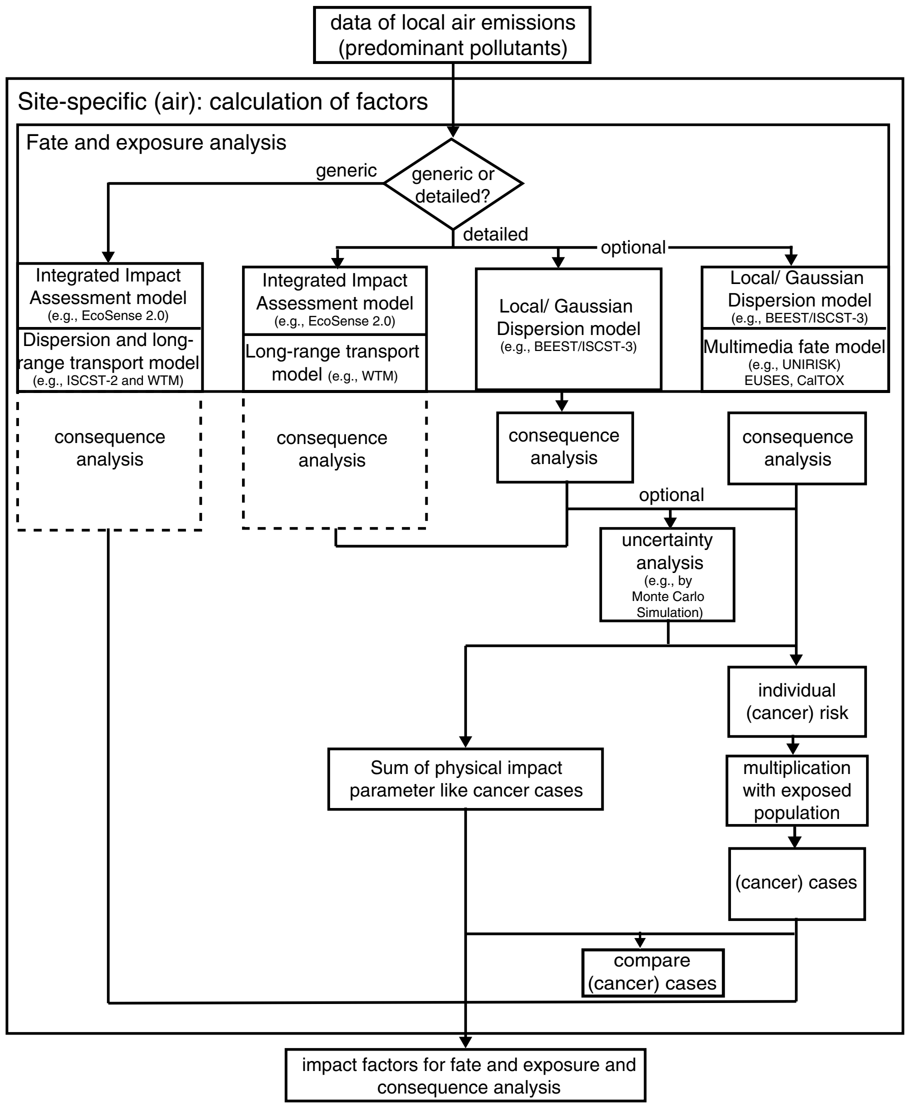


FIGURE 6.15 Site-specific impact assessment for air emissions.

Figure 6.15 proposes carrying out the consequence analysis for the damages due to long-range transport within the integrated impact assessment model and doing so separately, e.g., on a spreadsheet, for the damages generated by the emissions near the emission point. Then the physical impact parameter of both assessments can be summed up. This is a practical proposal and, in principle the consequence analysis can also be carried out with this.

When calculating the site-specific impact factors a lot of work is done to complete an ERA at the same time, as described in Chapter 4. The only additional step is the application of the multimedia fate model (e.g., UniRisk, EUSES, CalTOX, etc.,). Another way of stating this is to say that if an ERA has been carried out (as described in Chapter 4) 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 realization of an ERA is seen in Figure 6.15 as an optional element; this approach is further explored in Chapter 8.

Also, in the case of environmental risk assessment, a consequence analysis must be carried out. Due to the high level of detail in this type of study, thresholds can also be considered; thus, in the case of human health assessment, the consequence analysis is not restricted to carcinogenic effects and respiratory diseases (see Chapter 4 for further explanations) and other types of toxic effects can also be considered.

When carrying out an ERA, the individual risk must be calculated, e.g., developing cancer due to the increment of a certain pollutant in the atmosphere. Multiplication by the absolute number of population exposed would then allow obtaining an estimate of the damage in the form of physical impact parameters, e.g., cancer cases. These calculated impact parameters, like cancer cases, could also be compared with those provided by the application of the IPA on a local scale. Such a comparison can provide correction factors.

Another optional element is the uncertainty analysis that can be carried out, e.g., by MC simulation according to the framework proposed in Chapter 5. In the same way as for the LCI results, the outcomes of the fate and 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 and exposure and consequence analysis has been on site-dependent impact assessment as an adequate trade-off between accuracy and practicability. The entire method is largely explained in Chapter 7. The other options of the fate and exposure and consequence analysis (Figure 6.14) do not need further explanations because they are similar to the site-specific impact assessment or consist only in the application of published values for impact indicators.

6.5.7 Damage Profile

Figure 6.16 presents the last part of the obligatory steps for the methodology of environmental damage estimations for industrial process chains. In principle, this flowchart consists of an illustration of the developed mathematical framework. For each selected impact score, the eco-technology matrix is multiplied with the damage-assigning matrices. The result can be another matrix or a vector for each damage-assigning matrix, or a vector for the case of global impacts. In that case, the elements of the vector need only be summed up. In the case of the matrix, a sum must be made of the elements of the main diagonal, the trace. The matrix allows checking 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

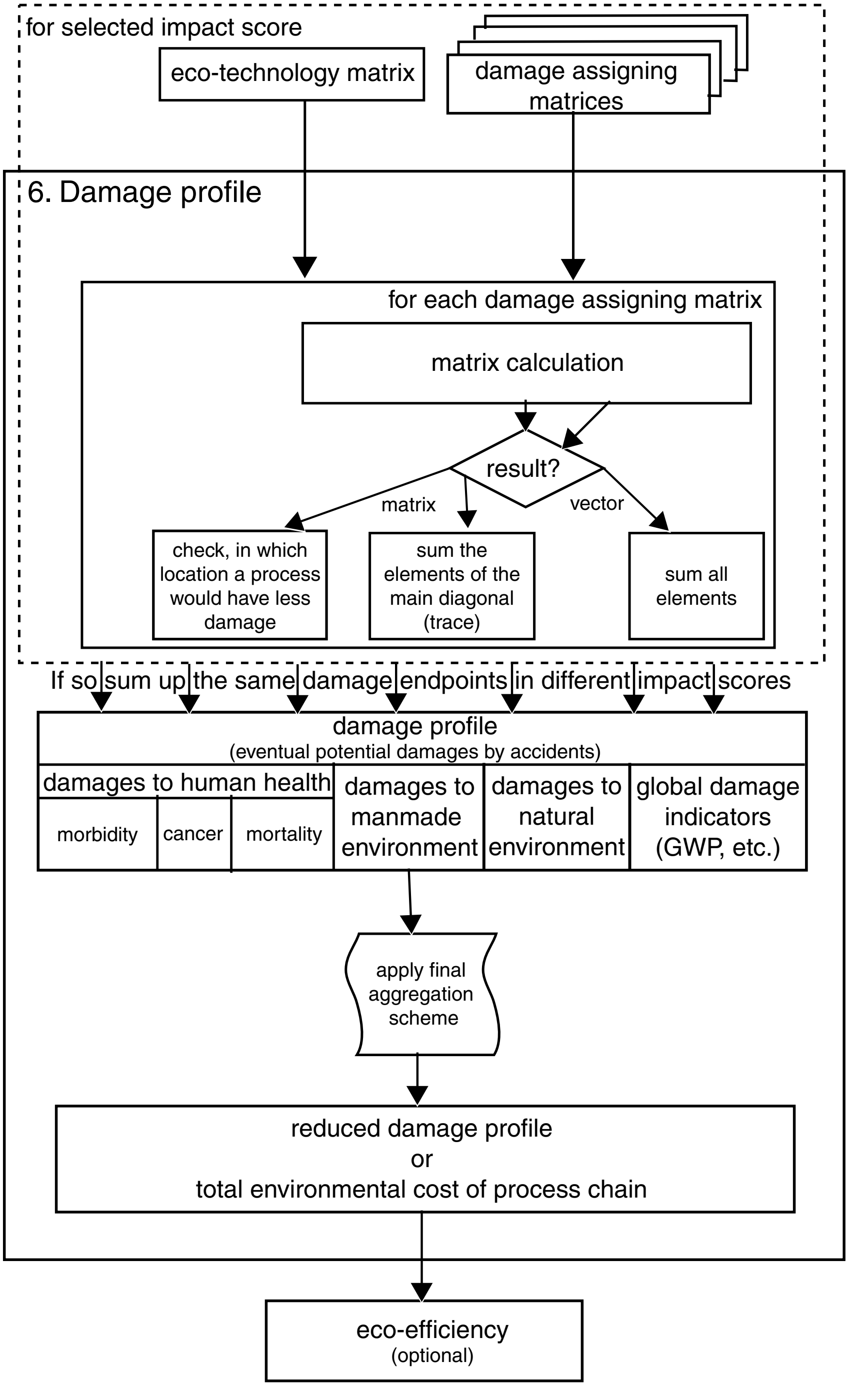


FIGURE 6.16 Damage profile.

have been estimated in different impact scores, they must be summed up. The damage profile might contain potential damages in 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 human health (morbidity, cancer, mortality), manmade environment and natural environment AoPs, as well as the so-called global damage indicators, which could be related to the life support functions and resources sub-AoPs if resource depletion is considered an environmental problem.

If the damage profile consists of different damage endpoints, their number can be further reduced by applying the selected aggregation scheme. In this way a reduced damage profile or an estimation of the total environmental cost of the process chain under study is finally obtained.

6.5.8 Eco-Efficiency

A further optional element is the calculation of the eco-efficiency of the industrial process chain for which environmental damages have been estimated. The concept of eco-efficiency has been proposed as an expression of sustainability for economic activities (see Chapter 1). Eco-efficiency has been defined as the delivery of competitively priced goods and services while progressively reducing environmental impacts.

Thus, measuring 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}, has been suggested. Although production costs are easy 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 of Figure 6.17 is applicable to the final result of the environmental damage estimations for industrial process chains if this is expressed in a monetary unit. This figure illustrates the procedure to calculate eco-efficiency according to this expression. Instead of production costs (operation and investment costs), the expected utility or net value could be used for determining eco-efficiency.

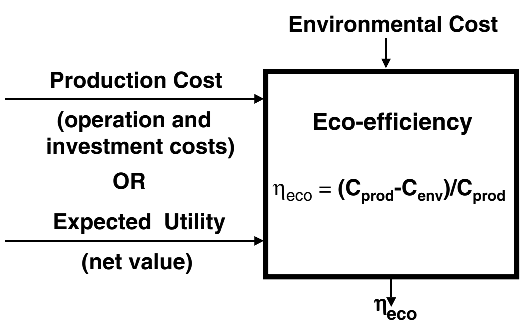


FIGURE 6.17 Eco-efficiency of an industrial process chain.

6.6 POSSIBLE ROLE OF THE FRAMEWORK IN ENVIRONMENTAL MANAGEMENT

Application patterns of LCA have been studied by Frankl and Rubik (1999). This section provides suggestions for possible applications of the presented methodology. The developed framework can be of interest for the following stakeholders:

- Public administration (to justify taxes and to optimize waste management strategies)
- Financial entities (to ascertain possible risks of payments)
- Companies (to contribute to a system of continuous environmental improvement, to choose between sites of a process chain from an environmental perspective, and to demonstrate reduction of environmental damages by the inversion in abatement technologies from a life-cycle viewpoint)
- Consumers and general community (to have the possibility of obtaining products and services of minor pollution and to learn about the environmental damages behind industrial process chains)

6.6.1 Public Administration

Table 6.2 shows in detail the possible applications for public administration, among which are 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 of end-of-life management is to develop a second generation of integrated waste management software, like Wisard (Ecobilan, 1999), that considers the problem setting also; this is quite related to environmental justice. By combining site-specific aspects with life-cycle considerations, new plans for waste management can assess transport processes and their routes in relation to waste treatment facilities and their sites and the respective environmental damages in an integrated manner.

TABLE 6.2
Applications for 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	Possible 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)	Possible eco-efficiency

TABLE 6.3
Applications for Financial Entities (Banks and Insurance Companies)

Application	Example	Optional element
Planning of new plants or changes	Waste incineration, land fill, cracker	Possible accidents
New insurance of existing plant	Waste incineration, land fill, cracker	Possible accidents
Business strategic planning	Future scenario assessment	Possible eco-efficiency

6.6.2 Financial Entities

The possible applications for financial entities (banks and insurance companies) are presented in Table 6.3. Main applications are the assessment of risk from 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.

6.6.3 Production and Other Service Companies

Table 6.4 represents the many possible applications for production and other service companies. Many of the applications proposed in this table could interest companies using the developed methodological framework. Especially important are the integrated end-of-life management and parallel optimization of the life-cycle and problem setting from an environmental point of view. Furthermore, all the questions related to chain responsibility with the assessment of supplier and waste treatment companies and their technologies are important. The latter includes the evaluation of accident risks with regard to processes like overseas petroleum transport, for which several big companies were declared responsible in recent years.

In addition to these companies, the methodological framework is certainly relevant for the waste treatment sector that wants to document improvement of the environmental profile to 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 assist greatly in this task by generating an important amount of quite objective, relevant information that should allow all interested parties to find a convincing solution.

6.6.4 Consumers and Society in General

The possible applications for consumers and society in general are shown in Table 6.5. The main application consists in education and communication as potential common ground for discussion about environmental damage estimations due to

TABLE 6.4
Applications for Production and Other Service Companies

Application	Example	Optional element
Planning of new plants	Waste incineration, landfill, cracker	Possible 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	Possible eco-efficiency
Documentation of environmental profile's improvement toward the public	Advanced gas treatment system in waste incineration plant	None
Process improvement	Change of process in chemical industry, additional flue gas cleaning in waste incineration	Possible eco-efficiency
Determination of most problematic processes	Production with a huge number of complex steps	Special dominance analysis, possible 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	Possible uncertainty analysis
Marketing strategies by communication of environmental profile to consumers	Waste incineration, chemical industry plant	None
Business strategic planning	Future scenario assessment	Possible eco-efficiency
End of life management	Waste incineration, landfill, integrated waste management planning (similar to Wizard by Ecobilan, 1999), but on a more detailed level with regard to spatial aspects (second generation)	Possible eco-efficiency
Optimization of setting (new plant)	Waste incineration, landfill, chemical industry	Possible ERA

TABLE 6.5
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-labeling 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 advances in information technologies.	Possibly uncertainty analysis

certain problematic industrial process chains such as the waste incineration that has been part of the public discussion on environmental aspects for the last decade.

Another potential application could be in future eco-labeling and environmental product declarations because the developed methodology would be a quite accurate way of obtaining relevant results about the environmental damages caused by product systems. At the moment, however, it seems to be not very practicable for this purpose. Nevertheless, in the future this information might be available due to advances in the information technologies.

6.7 EXAMPLE: NECESSARY TECHNICAL ELEMENTS

The methodology permits various linkages with other environmental management tools and concepts as well as technical elements. In the previous chapters we have seen several of them.

Since, the entire methodology is a combination of different analytical tools that, in general, have been developed for other applications, the reader is asked to identify those concepts, tools, and technical elements behind the presented framework.

In principle, the LCA methodology has been developed for the environmental assessment of product systems. LCA is an important element for the LCI analysis and LCIA methods and for providing region technology-dependent impact factors. The next tool is the impact pathway analysis (IPA) that is the fruit of a project to assess the externalities of electricity production. IPA is crucial for the fate and exposure and consequence analysis, including the weighting and aggregation schemes. Furthermore, ERA has its origin in assessment of the behavior of chemical substances in the environment. It is, of course, relevant in the fate and exposure and consequence analysis and has influenced not only 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 and English (1999). Due to the LCA part, a functional unit must be defined and allocation models must be used in the LCI analysis. In the fate and 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 tests are often the basis for dose–response functions and epidemiological studies are the basis for exposure–response functions. In order to make the weighting transparent, decision trees have been established. Socioeconomic impact assessment is conducted with the presented different methods to evaluate external costs. ERA uses individual risk- or population risk based on the lifetime average dose. Accident 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 and the uncertainty analysis carried out with MC simulation.

6.8 CASE STUDY: ENVIRONMENTAL DAMAGE ESTIMATIONS FOR THE WASTE INCINERATION PROCESS CHAIN

The methodology has successfully been applied to the case study on the process chain related to waste incineration, with interesting results. The information obtained by the developed methodology might be crucial in the future for decisions on further improvement of existing and new waste management systems. The presented algorithm is applied to the life-cycle inventory of the electricity produced by the MSWI of Tarragona, Spain.

6.8.1 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 (AGRS). More data on the MSWI are presented in Chapter 1.

The function of the MSWI is to treat the household waste of the surroundings of Tarragona. However, produced TJ of electricity is chosen as the functional unit, as was done in the existing LCA study (see Chapter 2). The study comprises in its system boundaries all the processes from municipal waste disposal in containers to the landfill of the final waste (Figure 6.18). The midpoint-based weighting method with single index, eco-indicator 95 (Goedkoop 1995) is used as LCIA method . The method is based on the characterization factors presented by Heijungs et al. (1992) and uses equal scores of distances to political targets (for more information see Chapter 3).

For reasons of resource economy, in the dominance analysis 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

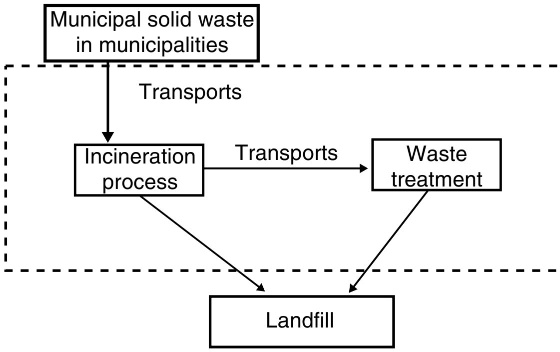


FIGURE 6.18 Boundaries of the studied system. (Reprinted from *J. Hazardous Mater.,* 77, Sonnemann, G.W. et al., pp. 91–106, ©2000 with permission from Elsevier.)

of values published in the literature to estimate environmental damages. The selected weighting and aggregation scheme corresponds to the one presented as an example in Figure 6.9, which means that three indicators have been selected for the weighting of impacts. For the human health and manmade environment AoPs, external environmental costs (EEC) according to the European Commission (EC 1995) have been used.

A lot of criticism exists concerning monetary evaluation of environmental damages. For this reason, special attention is paid to arguments to monetize these damages. The impossibility of summing up the noneconomic impact endpoints necessarily implies a value judgment. Because most decisions must confront the reality of the market place, 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 public institutions and how much of the damage cost should be internalized so that a functional unit is consistent with the market. Further information on this topic can be found in Chapter 3 and in the huge externality studies for electricity production carried out in parallel in the EU (EC, 1995) and U.S. (ORNL/REF 1995).

No acceptable economic method exists for damage evaluation of the natural environment AoP (biodiversity and landscape). Therefore, the evaluation must be carried out through an ecological damage parameter. In the present study, the parameter applied is the REW ecosystem area in which the critical load of a pollutant is exceeded (UN-ECE, 1991); see also Chapter 3 for details.

The global damages that might occur in the future due to the emission of greenhouse gases are highly uncertain for forecasting and monetization. Therefore, the climate change has been expressed in the form of the GWP as in the LCIA (Albritton and 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 5. An environmental risk assessment for the same plant has been carried out and the results are presented in Chapters 4 and 5. Based on the results of the environmental damage estimations of the waste incineration process chain, the eco-efficiency will also be calculated.

6.8.2 Life-Cycle Inventory Analysis

The LCI analysis is described in Chapter 2. Here, the existing results are used for creating the eco-technology matrix of the environmental damage estimation for industrial process chains.

6.8.3 Life-Cycle Impact Assessment Method

The detailed results of the application of the LCIA method, eco-indicator 95, can be found in Chapter 3, in which a comparison of the results for the two situations based on the eco-indicator 95 is conducted. In the presented methodology, the impact score is further applied for the dominance analysis.

6.8.4 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 5. Figure 6.19 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. Therefore, 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: 1) the data obtained from an IPA study of the incineration process in Tarragona are considered valid for all processes in the Tarragona region and 2) the remaining processes must 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 "production of CaO" and "treatment of ashes" processes 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 "transport" and "scrap-metal recycling" also depend on the Spanish region (see Figure 6.19).

The results of the inventory analysis of the current situation for the relevant processes and the selected environmental loads are presented in Table 6.6. This table includes the eco-technology matrix with submatrices: the ELs from kWh to NO_x correspond to the first matrix for the economic damage parameter; the second matrix for the ecological damage parameter consists only of SO_2 and NO_x, and the third matrix for the global damage parameter includes the other loads from CO_2 to trichloromethane (see Table 6.6).

6.8.5 Fate and Exposure and Consequence Analysis

In the next step, the corresponding three damage-assigning matrices for the selected three indicators are established. An attempt is made to give a particular value to each environmental load for a specific region or site because the respective indicator

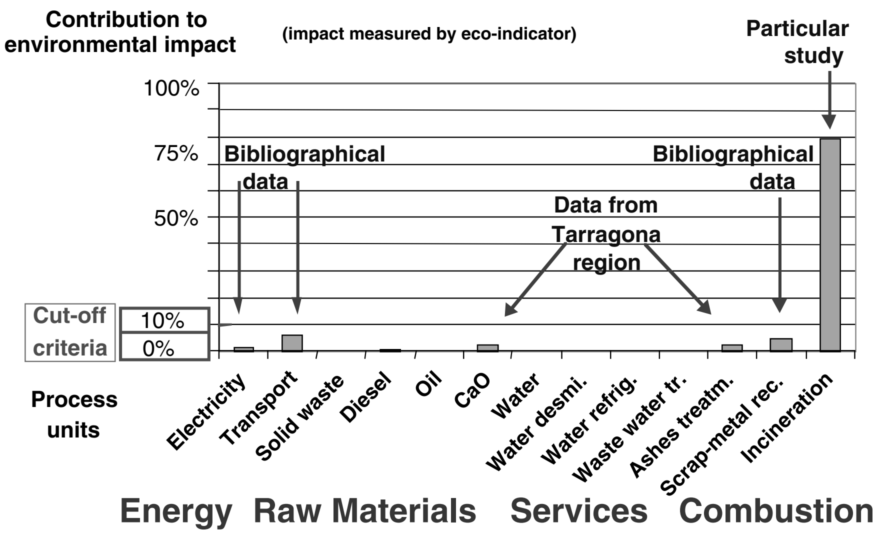


FIGURE 6.19 Dominance analysis for the relevant processes for site-specific damage assessment. (Reprinted from *J. Hazardous Mater.,* 77, Sonnemann, G.W. et al., pp. 91–106, ©2000 with permission from Elsevier.)

depends on the characteristics of the respective region or site for each environmental load. If no region-dependent or process-specific damage estimates are available for a pollutant, a general model, the uniform world model (UWM; Rabl et al., 1998; see Chapter 3) 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 6.7. 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 truck metric ton km (tkm) in South Europe (Friedrich et al., 1998) because data for Spain are not available. The CaO supply, ash treatment and incineration processes are located in the Tarragona region. For the incineration process described in Chapter 5, the IPA obtains the weighting factors of the Tarragona region for all environmental loads that accept VOC (volatile organic carbon). The VOC values as well as the evaluation of the scrap-metal recycling are taken from the UWM (Rabl et al., 1998).

Due to lack of literature data, the matrix of the REW ecological damage indicator 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 6.8 to illustrate the complete framework. However, in the case of the global damage indicator, the situation is different. The GWP is independent of the site where the substances like CO_2 or CH_4 are emitted; therefore a vector rather than a weighting matrix is obtained. See Table 6.9 for the current situation.

TABLE 6.6
Eco-Matrix with Life-Cycle Inventory Data for Selected Environmental Loads, Divided into Those Assessable by Environmental Costs and Those Responsible for Ecosystem Damages and Global Warming (Current Situation)

Relevant Process	Electricity	Transport	CaO	Ash treatment	Scrap-metal recycling	Incineration
Region	Spain	Tarragona/ Catalonia	Tarragona/ Catalonia	Tarragona	Madrid/ Spain	Tarragona
kWh	2.96×10^3	0.00	0.00	0.00	0.00	0.00
tkm	0.00	2.64×10^4	0.00	0.00	0.00	0.00
kg PM_{10}	2.08	8.45	1.11×10^2	4.73	6.03	2.82×10^1
kg As	3.12×10^{-4}	2.93×10^{-3}	3.74×10^{-4}	8.66×10^{-4}	2.09×10^{-3}	3.29×10^{-2}
kg Cd	6.97×10^{-4}	4.28×10^{-4}	4.74×10^{-5}	1.41×10^{-4}	3.06×10^{-4}	3.87×10^{-2}
kg Ni	1.29×10^{-4}	5.56×10^{-3}	1.31×10^{-3}	2.42×10^{-3}	3.98×10^{-3}	4.93×10^{-2}
kg VOC	4.74×10^{-1}	1.68×10^1	5.84×10^{-1}	4.86	1.21×10^1	0.00
ng Dioxins TEQ	8.05×10^1	7.32×10^2	3.83×10^2	7.57×10^2	5.22×10^2	1.17×10^4
kg CO	3.36×10^{-1}	2.92×10^1	2.46	1.16×10^1	2.09×10^1	2.35×10^2
kg SO_2	1.86×10^1	1.93×10^1	1.16×10^1	1.17×10^1	1.38×10^1	1.77×10^2
kg NO_2	3.30	9.31×10^1	5.76	3.17×10^1	6.65×10^1	1.12×10^3
kg CO_2	1.48×10^3	9.13×10^3	8.02×10^3	5.76×10^3	6.53×10^3	2.25×10^5
kg dichloro-methane	9.90×10^{-7}	3.17×10^{-6}	5.64×10^{-7}	1.11×10^{-6}	2.26×10^{-6}	0.00
kg Halon-1301	2.27×10^{-5}	1.09×10^{-3}	3.61×10^{-5}	3.21×10^{-4}	7.80×10^{-4}	0.00
kg methane	4.51	1.41×10^1	7.57	8.19	1.01×10^1	0.00
kg N_2O	1.27×10^{-2}	1.00	3.55×10^{-2}	2.75×10^{-1}	7.16×10^{-1}	0.00
kg tetrachloro-methane	3.03×10^{-7}	1.03×10^{-5}	6.41×10^{-6}	3.78×10^{-6}	7.36×10^{-6}	0.00
kg trichloro-methane	1.55×10^{-8}	1.10×10^{-6}	7.06×10^{-7}	4.03×10^{-7}	7.83×10^{-7}	0.00

6.8.6 Damage Profile

The multiplication of the damage-assigning matrices with the eco-technology matrix (the respective parts of the inventory table) yields the damage profile (see Table 6.10 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 U.S.$/TJ electricity, which 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 becomes clear that the damage generated by the functional unit would be higher if all the processes were in the UWM and less if they were all in the Tarragona region (situated on the Mediterranean coast). In the case of the

TABLE 6.7
Damage-Assigning Matrix for External Environmental Cost (EEC) in U.S.$

Relevant process	Region and source	kWh	tkm	kg PM 10	kg As	kg Cd	kg Ni	kg VOC	PCD D/Fs ng TEQ	kg CO	kg SO_2	kg NO_2
Electricity	Spain[a]	0.040	0.00	0	0.0	0	0.0	0.00	0.00	0.00	0	0
Transport	South Europe [b]	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	2.6×10^{-8}	1.03	13	11
Ash treatment	Tarragona, site-specific transfer	0.00	0.00	23	3.0	27	61	0.73	2.6×10^{-8}	1.03	13	11
Scrap-metal recycling	UWM [c]	0.00	0.00	14	156	19	2.6	0.73	1.7 x 10^{-5}	0.00 2	13	19
Incineration	Tarragona, site-specific	0.00	0.00	23	3.0	27	61	0.73	2.6 x 10^{-8}	1.03	13	11

[a]CIEMAT, ExternE national implementation Spain — final report, Contract JOS3-CT95-0010, Madrid, 1997.
[b]Friedrich, R. et al., *External Costs of Transport*, Forschungsbericht Band 46, IER, Universität Stuttgart, Germany, 1998.
[c]Rabl, A. et al., *Waste Manage. Res.*, 16(4), 368–388, 1998

TABLE 6.8
Damage-Assigning Matrix for Relative Exceedance Weighted (REW) Area in km²

Relevant process	Region and source	kg NO_2	kg SO_2
Electricity	—	—	—
Transport	—	—	—
CaO	Tarragona, site-specific factor	1.31×10^{-5}	1.36×10^{-6}
Ash treatment	—	—	—
Scrap-metal recycling	Tarragona, site-specific factor	1.31×10^{-5}	1.36×10^{-6}
Incineration	Tarragona, site-specific	1.31×10^{-5}	1.36×10^{-5}

ecological damage parameter, only a few accurate values in the weighted eco-matrix are known. For the REW ecosystem area, the diagonal elements sum up to 0.0155 km² per TJ electricity — a relative small value because the studied region is not sensitive to acidification. For the GWP, a weighted eco-vector is obtained. The

TABLE 6.9
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[a]	1	15	4900	11	270	1300	25

[a]Data from Albritton, D. and Derwent, R., *IPCC, Climate Change*, Cambridge University Press, Cambridge, 1995.

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).

The damage profile of the current situation, Scenario 2, can now be compared to the situation of the MSWI in 1996 without AGRS as a different process design option, Scenario 1. The damage profile for the former situation in 1996 is presented in Table 6.11. Although the eco-indicator 95 shows a reduction of 60% in the total score between the former and the current situations (Chapter 3), the environmental damage estimations show less reduction. The external environmental costs decrease 10%, but the REW increases 3% and the GWP increases 10%. These 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 2) that affects the emissions of NO_x and CO_2, which are crucial for these damage factors.

Figure 6.20 offers a comparison of the external environmental costs with the eco-indicator 95 for the plant before and after installation of the advanced gas removal system. It can be seen that the environmental external cost estimations give much more weight to the transport processes. The transport contributes approximately 25% to the external environmental costs before installation of the advanced gas treatment system and 30% afterwards. 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. More than 90% of the eco-indicator 95 is attributed to the incineration in the former situation and it still accounts for 80% in the current situation. The external environmental cost estimations only assign 70% to the incineration process for the situation in 1996 and even less in the current situation, namely, a little more than 50%. This different relation between the transport and incineration processes also explains why the external environmental costs have decreased less than the eco-indicator 95 score. Another important process in the current situation is CaO production. Moreover, the selected relevant processes make up nearly 100% of the total impact (Figure 6.20).

TABLE 6.10
Damage Profile for the Current Situation (with Advanced Acid Gas Treatment System Scenario 2)

Regions	Electricity	Transport	CaO	Ash treatment	Scrap-metal recycling	Incineration	TJ Electricity
			EEC in U.S.$				
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	—	—	9.12×10^{-5}	4.31×10^{-4}	—	1.50×10^{-2}	1.55×10^{-2}
Tarragona	—	—	9.12×10^{-5}	4.31×10^{-4}	—	1.50×10^{-2}	1.55×10^{-2}
Uniform world model	—	—	0	0	—	0	1.55×10^{-2}
Tarragona	—	—	9.12×10^{-5}	4.31×10^{-4}	—	1.50×10^{-2}	1.55×10^{-2}
Life-cycle region	—	—	9.12×10^{-5}	4.31×10^{-4}	—	1.50×10^{-2}	1.55×10^{-2}
			GWP in kg CO_2 equivalents				
World	1.54E+03	9.56×10^{3}	8.11×10^{3}	5.93×10^{3}	6.83×10^{3}	2.25×10^{5}	2.57×10^{5}

TABLE 6.11
Damage Profile for the Former Situation in 1996 (without Advanced Acid Gas Treatment System Scenario 1)

Regions	Electricity	Transport	CaO	Ash treatment	Scrap metal recycling	Incineration	TJ Electricity
				EEC in U.S.$			
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 km²			
Spain	—	—	0	0	—	0	0.0000
(South) Europe	—	—	0	0	—	0	0.0000
Tarragona	—	—	0	7.03×10^{-5}	—	0.0150	0.0151
Tarragona	—	—	0	7.03×10^{-5}	—	0.0150	0.0151
Uniform world model	—	—	0		—	0	0.0000
Tarragona	—	—	0	7.03×10^{-5}	—	0.0150	0.0151
Life-cycle region	—	—	0	7.03×10^{-5}	—	0.0150	0.0151
				GWP in CO_2 equivalents			
World	1.47×10^3	9.11×10^3	0	9.69×10^2	6.51×10^3	2.14×10^5	2.32×10^5

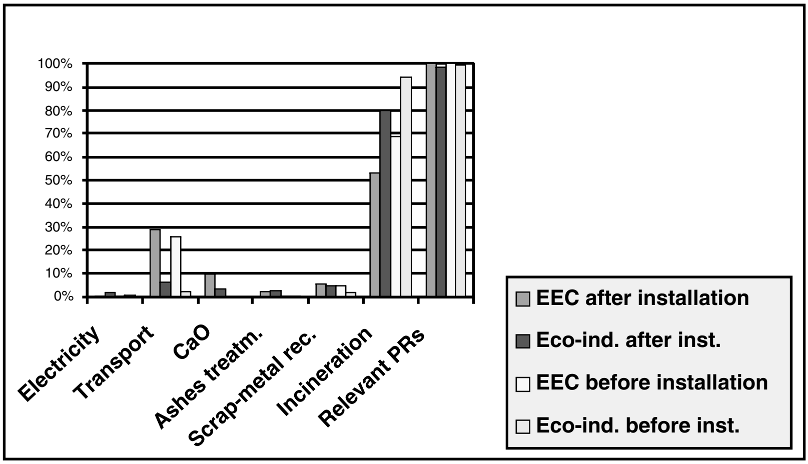


FIGURE 6.20 Comparison of external environmental cost with eco-indicator 95 for a plant before and after installation of an advanced acid gas removal system.

6.8.7 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 the expression in Figure 6.17, production costs are needed as an additional element. According to information from SIRUSA (Nadal, 1999), the production costs are:

- Former situation: 32,000 U.S.$/TJ
- Current situation: 37,000 u.S>$/TJ

These costs are based on the yearly total operation costs of the MSWI, including the financial costs for the investments made. The production costs are higher than the market price of 0.07 U.S.$/kWh; the deficit is paid by the price for the waste treatment, which can be considered a service the MSWI is providing to society. By applying the presented formula, the following eco-efficiency for the process chain is obtained:

- Former situation: η = 5%
- Current situation: η = 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 is done in the existing LCA of the MSWI process chain in Tarragona (see Chapter 2 for further explanations). Nevertheless, installation of an advanced acid gas removal system increases eco-efficiency significantly.

6.9 QUESTIONS AND EXERCISES

1. Sum up the challenges and key points of a strategy for a methodology that integrates life-cycle and risk assessment.
2. Which types of impacts are global and which regional? When is spatial differentiation important?
3. What are the steps to obtain the eco-technology and damage-assigning matrices?
4. Explain which decisions must be made in the weighting and aggregations phase.
5. Give examples of LCIA methods with impact score for
   - Midpoint based
   - Direct weighting
   - Endpoint weighting
6. Describe the fate and exposure and consequence analysis procedure.
7. Which technical elements are used in the different steps of the methodology?
8. For which reasons do you think the external cost approach gives less importance to the incineration process and more to the transports than the eco-indicator method?
9. Mention possible applications of an integrated life-cycle and risk assessment methodology.
10. The provided data correspond to the emissions of three different pollutants emitted in three different processes. By means of the correct eco-vector, express the environmental loads associated with each pollutant and process:
    1. Pretreatment process: 51.10 kg/kg feed of $CO_{2,}$; 2.7 kg/kg feed of $SO_{2;}$ 1kg/kg feed of NO_x
    2. Steam generation: 196 kg/kg feed of CO2; 1.2 kg/kg feed of SO2; 0.099 kg/kg feed of NO_x
    3. Electricity production: 293 kg/kg feed of CO2; 1.8 kg/kg feed of SO2; 2.3 kg/kg feed of NO_x
11. The data in Table 6.12 to Table 6.14 correspond to three different scenarios of a real process. Discuss which scenario corresponds to the minimum environmental damage for the pollutants under consideration.
12. Using the following fictive data for the eco-technology matrix and the damage-assigning matrix, calculate the corresponding damage profile:

    SO_2 emission of process 1: 100 g SO_2/kg product $_{process1}$
    SO_2 emission of process 2: 10 g SO_2/kg product $_{process2}$
    SO_2 emission of process 3: 30 g SO_2/kg product $_{process3}$
    Cd emission of process 1: 0.15 g Cd/kg product $_{process1}$
    Cd emission of process 2: 0.10 g Cd/kg product $_{process2}$
    Cd emission of process 3: 0.02 g Cd/kg product $_{process3}$
    Ni emission of process 1: 0.25 g Ni/kg product $_{process1}$
    Ni emission of process 2: 0.15 g Ni/kg product $_{process2}$
    Ni emission of process 3: 0.08 g Ni/kg product $_{process3}$

TABLE 6.12
Scenario 1: Steam Production with Refining and Electricity Production (Spanish Mix, 1995)

Mixture (kg emission/t C_3H_8)		Separation (kg emission/t C_3H_8)		
		Steam	Electricity	Venting
CO_2	504.49	619.94	151.59	—
SO_2	2.14	5.67	0.87	—
COD	0.055	0.0041	0.00046	—
VOC	2.73	8.51	0.77	0.00422

TABLE 6.13
Scenario 2: Steam Production with Refining and Electricity Production (Norwegian Matrix, 1995)

Mixture (kg emission/t C_3H_8)		Separation (kg emission/t C_3H_8)		
		Steam	Electricity	Venting
CO_2	504.49	619.94	110.23	—
SO_2	2.14	5.67	0.00108	—
COD	0.055	0.0041	0.00000173	—
VOC	2.73	8.51	0.00084	0.00422

TABLE 6.14
Scenario 3: Steam Production with Refining and Electricity Production (Natural Gas Combustion)

Mixture (kg emission/t C_3H_8)		Separation (kg emission/t C_3H_8)		
		Steam	Electricity	Venting
CO_2	504.49	619.94	573.29	—
SO_2	2.14	5.67	0.0137	—
COD	0.055	0.0041	0.00035	—
VOC	2.73	8.51	0.185	0.00422

Damage due to SO_2 in region 1: 0.01 U.S.$/kg $SO_{2\ region1}$
Damage due to SO_2 in region 2: 0.006 U.S.$/kg $SO_{2\ region2}$
Damage due to SO_2 in region 3: 0.008 U.S.$/kg $SO_{2\ region3}$
Damage due to Cd in region 1: 0.005 U.S.$/kg $Cd_{region1}$
Damage due to Cd in region 2: 0.004 U.S.$/kg $Cd_{region2}$
Damage due to Cd in region 3: 0.009 U.S.$/kg $Cd_{region3}$
Damage due to Ni in region 1: 0.003 U.S.$/kg $Ni_{region1}$
Damage due to Ni in region 2: 0.002 U.S.$/kg $Ni_{region2}$
Damage due to Ni in region 3: 0.008 U.S.$/kg $Ni_{region3}$

13. Knowing the impact scores given in Figure 6.21 for different processes of a product life-cycle, discuss for this case which cut-off criteria and which further treatment method (no consideration, site specific, site dependent or technology region dependent) would be most appropriate for each of the given processes.

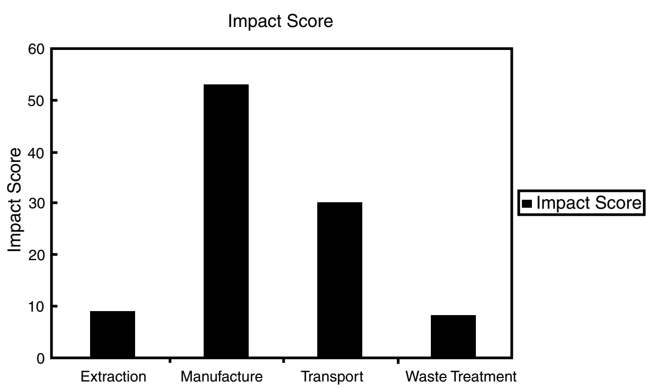


FIGURE 6.21 Impact scores for different processes of a product life-cycle.

REFERENCES

Aelion, V., Castells, F., and Veroutis, A. (1995), Life-cycle inventory analysis of chemical processes, *Environ. Prog.*, 14(3), 193–200.

AICHE (American Institute of Chemical Engineers) (1985), Guidelines for hazard evaluation procedures, prepared by Batelle Columbus Division for The Center for Chemical Process Safety.

Albritton, D. and Derwent, R. (1995), Trace gas radiative forcing indices, in *IPCC, Climate Change*, Cambridge University Press, Cambridge.

Bare, J.C., Hofstetter, P., Pennington, D.W., and Udo de Haes, H.A. (2000), Midpoints versus endpoints — the sacrifices and benefits, *Int. J. LCA*, 5, 319–326.

Castells, F., Aelion, V., Abeliotis, K.G., and Petrides, D.P. (1995), An algorithm for life-cycle inventory, *AICHE Symp. Series Pollut. Prev. Process Product Modifications*, 90(303), 151–160.

CHAINET (European Network on Chain Analysis for Environmental Decision Support) (1998), Definition document, CML, Leiden University, The Netherlands.

CIEMAT (1997), ExternE national implementation Spain — final report, contract JOS3-CT95-0010. Madrid.

Curran, M.A. and Young, S. (1996), Report from the EPA conference on streamlining LCA, *Int. J. LCA*, 1(1), 57–60.

Dale, V. and English, M. (1999), *Tools to Aid Environmental Decision-Making*, Springer-Verlag, New York.

EC (European Commission) (1995), *ExternE — Externalities of Energy*, 6 vols., EUR 16520-162525. DG XII Science, Research and Development, Brussels.

Ecobilan Group (1999), WISARD — Waste-Integrated Systems Assessment for Recovery and Disposal, Software, Paris.

Fleischer, G. and Schmidt, W.P. (1997), Integrative screening LCA in an ecodesign tool, *Int. J. LCA*, 2(1), 20–24.

Frankl, P. and Rubik, F. (1999), *Life-Cycle Assessment in Industry and Business — Adoption Patterns, Applications and Implications*, Springer-Verlag, Berlin.

Friedrich, R., Bickel, P., and Krewitt, W. (1998), *External Costs of Transport*, Forschungsbericht Band 46, IER, Universität Stuttgart, Germany.

Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G., Hischier, R., Martin, A. (ETH Zürich), Dones, R., and Gantner, U. (PSI Villigen) (1996), *Ökoinventare von Energiesystemen - Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz*, 3rd ed., ETH Zürich: Gruppe Energie-Stoffe-Umwelt, PSI Villigen: Sektion Ganzheitliche Systemanalysen.

Goedkoop, M.J. (1995), Eco-indicator 95 — final report, NOH report 9523, Pré Consultants, Amersfoort, The Netherlands.

Heijungs, R. (1998), Economic drama and the environmental stage, Ph.D. thesis, CML, Leiden University, The Netherlands.

Heijungs, R., Guinée, J.B., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., and Wegener-Sleeswijk, A. (1992), Environmental life-cycle assessment of products — guide and backgrounds, technical report, CML, University of Leiden, The Netherlands.

Hofstetter, P. (1998), *Perspectives in Life-Cycle Impact Assessment — a Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere*, Kluwer Academic Publishers, London.

Krewitt, W., Trukenmüller, A., Bachmann, T., and Heck, T. (2001), Country-specific damage factors for air pollutants — a step towards site-dependent life-cycle impact assessment, *Int. J. LCA*, 6(4), 199–210.

LaGrega, M.D., Buckingham, P.L., and Evans, J.C. (1994), *Hazardous Waste Management*, McGraw-Hill, New York.

Meyerhofer, P., Krewitt, W., and Friedrich, R. (1997), Extension of the accounting framework, final report. ExternE Core Project, IER, Universität Stuttgart, Germany.

Murray, C.J.L. and Lopez, A.D.(1996), *The Global Burden of Disease*, vol. 1 of *Global Burden of Disease and Injury Series*, WHO, Harvard School of Public Health, World Bank, Harvard University Press, Boston.

Nadal, R. (1999), Planta incineradora de residuos sólidos urbanos de Tarragona, report for Master en Enginyeria i Gestió Ambiental (MEGA), Universitat Rovira i Virgili, Tarragona, Spain.

Nigge, K.M. (2000), *Life-Cycle Assessment of Natural Gas Vehicles, Development and Application of Site-Dependent Impact Indicators*, Springer-Verlag, Berlin.

Olsen, S.I., Christensen, F.M., Hauschild, M., Pedersen, F., Larsen, H.F., and Toerslov, J. (2001), Life-cycle impact assessment and risk assessment of chemicals — a methodological comparison, *Environ. Impact Assessment Rev.*, 21, 385–404.

ORNL/REF (Oak Ridge National Laboratory and Resources for the Future Oak Ridge) (1995), *U.S. — EC Fuel Cycle Study*, reports No. 1 through 8, McGraw-Hill/Utility Data Institute, Washington, D.C.

Pesonen, H., Ekvall, T., Fleischer, G., Huppes, G., Jahn, C., Klos, Z.S., Rebitzer, G., Sonnemann, G.W, Tintinelli, A., Weidema, B.P., and Wenzel, H. (2000), Framework for scenario development in LCA, *Int. J LCA*, 5(1), 21–30.

Rabl, A., Spadaro, J.V., and McGavran, P.D. (1998), Health risks of air pollution from incinerators — a perspective, *Waste Manage. Res.*, 16(4), 368–388.

Schulze, C. (2001), Modeling and evaluating the aquatic fate of detergents, Ph.D. thesis, Universität Osnabrück, Germany.

Sonnemann, G.W., Schuhmacher, M., and Castells, F. (1999), From LCIA via risk assessment to environmental external cost evaluation — what are the possibilities? Presented at 9th SETAC-Europe Annual Meeting, Leipzig, Germany.

Sonnemann, G.W., Schuhmacher, M., and Castells, F. (2000), Framework for the environmental damage assessment of an industrial process chain, *J. Hazardous Mater.*, B77, 91–106.

Spirinckx, C. and de Nocker, L. (1999), Comparative life-cycle analysis and externality analysis of biodiesel and fossil diesel fuel, presented at 9th SETAC-Europe Annual Meeting, Leipzig, Germany.

Udo de Haes, H.A. and Lindeijer, E. (2001), The conceptual structure of life-cycle impact assessment, final draft for the Second Working Group on Impact Assessment of SETAC-Europe (WIA-2), in Udo de Haes et al. (2002), *Life-Cycle Impact Assessment: Striving towards Best Practice*, SETAC Order #SB02-5.

Udo de Haes, H.A., Joillet, O., Finnveden, G., Hauschild, M., Krewitt, W., and Mueller-Wenk, R. (1999), Best available practice regarding categories and category indicators in life-cycle impact assessment — background document for the Second Working Group on LCIA of SETAC-Europe, *Int. J. LCA*, 4, 66–74; 167–174.

UN-ECE (United Nations Economic Commission for Europe) (1991), Protocol to the 1979 convention on long-range transboundary air pollution concerning the control of emissions of volatile organic compounds or their transboundary fluxes, report ECE/EB AIR/30, UN-ECE, Geneva, Switzerland.

7 Site-Dependent Impact Analysis*

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7.1 INTRODUCTION

In this chapter we present a provisional approach to overcoming the disadvantages of site-generic and site-specific methods. This approach is one of the recently developed site-dependent impact assessment methods that can be considered a trade-off between exactness and feasibility. As with site-specific approaches, fate, exposure and effect information are taken into account, but indicators applicable for classes of emission sites rather than for specific sites are calculated. 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 regard them as the most promising alternative for future estimations of environmental damages in industrial process chains. The approach is adapted so that it fits perfectly 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.

The consideration of spatial differentiation in LCIA was proposed first by Potting and Blok (1994). However, it took time until developments for site-dependent impact assessment such as those by Potting (2000) and Huijbregts and Seppälä (2000) were made in an operational way, especially for acidification and eutrophication. 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 and Rabl (1999). Potting (2000) establishes impact indicators that take into account different release heights, population density, and 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.

* Selected text in Sections 7.2, 7.3, and 7.4 has been reprinted with permission from Nigge, K.M., *Life-Cycle Assessment of Naturlal Gas Vehicles: Development and Application of Site-Dependent Impact Indicators,* ©2000 by Springer-Verlag, Berlin.

Moriguchi and Terazono (2000) present an approach for Japan in which meteorological conditions are set to be equal for all examples. Nigge (2000) offers a method for statistically determined population exposures per mass of pollutant that considers short-range and long-range exposure separately and allows taking 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 also valid 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 that 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 and 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 of 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.

7.2 GENERAL FRAMEWORK FOR SITE-DEPENDENT IMPACT ANALYSIS

Udo de Haes (1996) distinguished the following dimensions of impact information that are relevant for life-cycle impact assessment (LCIA): information concerning (1) effect, (2) fate and exposure, (3) background level, and (4) space. The first two dimensions directly refer to the cause–effect chain. The last two 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 7.1. 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 connected to the source (emission, distribution/dispersion, and concentration increase). The third dimension comprises the third dimension from Udo de Haes (1996) and, additionally, 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 characterization modeling addresses different levels of sophistication. In the effect analysis, the sophistication increases from the use of legal threshold standards via the application of NOEC (no observable effect concentration) to the integration of slope in the impact factors. These factors

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 7.1 Dimensions of impact information and levels of sophistication in life-cycle impact assessment. (From Wenzel, H. et al., *Environmental Assessment of Products. Volume 1 – Methodology, Tools and Case Studies in Product Development*, Chapman & Hall, London, 1997. With permission.)

may include no, some or comprehensive fate information. The same holds true for the information on the target system.

Category indicators for toxicity, but also for other endpoint-oriented 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 in which the transfer of the pollutant across different environmental media or compartments (e.g., air, water, soil) needs to be considered, Expression 7.1 gives the category indicator of incremental damage, ΔD_p^{nm} (damage such as cases of cancer, a for YOLLs and DALYs or U.S.$ for external environmental costs), characterizing the effect in the compartment m of the pollutant p emitted in the initial compartment n:

$$\Delta D_p^{nm} = E_p^m \cdot F_p^{nm} \cdot M_p^n \tag{7.1}$$

where

M_p^n is the mass of pollutant p (kg) emitted into the initial medium n (air, water or soil).

F_p^{nm} is the fate and exposure factor for the emission of substance p into the initial medium n and transfer into medium m in the form of (m^2.yr/m^3) or (m^3.yr/m^3), depending on the compartments considered and taking into account the propagation, degradation, deposition, transfer among media and food chain or bioconcentration routes.

E_p^m is the effect factor (damage/m^2.(µg/m^3).yr)) 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 the 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 air compartment can have impacts in the air (inhalation), soil (via deposition) and water (absorption) target compartments. The pollutants can be transported farther to other target compartments by other routes (soil–plant etc.). A large variety of possible routes is between the release and target compartments.

Expression 7.1 does not explicitly consider the distribution and 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 fish population. The effective receptor density $\rho^{m}_{eff,r}$ 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 7.1 then reads as follows:

$$\Delta D^{nm}_{p,r} = E^{m}_{p,r} \cdot \rho^{m}_{eff,r} \cdot F^{nm}_{p} \cdot M^{n}_{p} \tag{7.2}$$

where

$\Delta D^{nm}_{p,r}$ is the incremental damage caused (damage) due to the emission of pollutant p into the initial medium n (air, water or soil) on the receptor r in the target compartment m (air, water, soil or food chain).

$\rho^{m}_{eff,r}$ is the 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 need to consider a density, but a concentration.

$E^{m}_{p,r}$ is the effect factor (damage/receptors.(μg/m^3).yr) 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 the following expression:

$$\Delta RE^{nm}_{p,r} = \Delta D^{nm}_{p,r} / E^{m}_{p,r} = \rho^{m}_{eff,r} \cdot F^{nm}_{p} \cdot M^{n}_{p} \tag{7.3}$$

where

$\Delta RE^{nm}_{p,r}$ is the incremental receptor exposure during a certain period of time (receptors.(μg/m^3).yr) due to the emission of pollutant p into the initial medium n (air, water or soil) on the receptor r in the target compartment m (air, water, soil or food chain) — for example, for airborne emissions with the receptor human population (persons.μg/m^3 air.yr) and for the receptor material surface (maintenance surface. μg/m^3 air.yr) and for water with the receptor fish population (fishes.mg/m^3 water.yr).

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 and 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 $\Delta D^{nm}_{p,r}$ is independent of the time pattern of the emission. The incremental damage can then be calculated by Expression 7.4, using an incremental receptor exposure per mass of pollutant emitted $I^{nm}_{p,r,i} = \Delta RE^{nm}_{p,r,i} / M^{n}_{p,i}$. 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 such as the stack height for emissions to air or the release depth into the lake or river for emissions to water.

$$\Delta D^{nm}_{p,r,i} = E^{m}_{p,r} \cdot M^{n}_{p,i} \cdot I^{nm}_{p,r,i} \tag{7.4}$$

where

$\Delta D^{nm}_{p,r,i}$ is the incremental damage caused (damage) due to the emission of pollutant p 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).

$M^{n}_{p,i}$ is the mass of pollutant p (kg) emitted into the initial medium n (air, water or soil) at the emission situation i.

$I^{nm}_{p,r,i}$ is the incremental receptor exposure per mass of pollutant emitted (receptors.(μg/m^3).yr/kg) due to the amount of pollutant p 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 illustrated in Figure 7.2. 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 7.1.

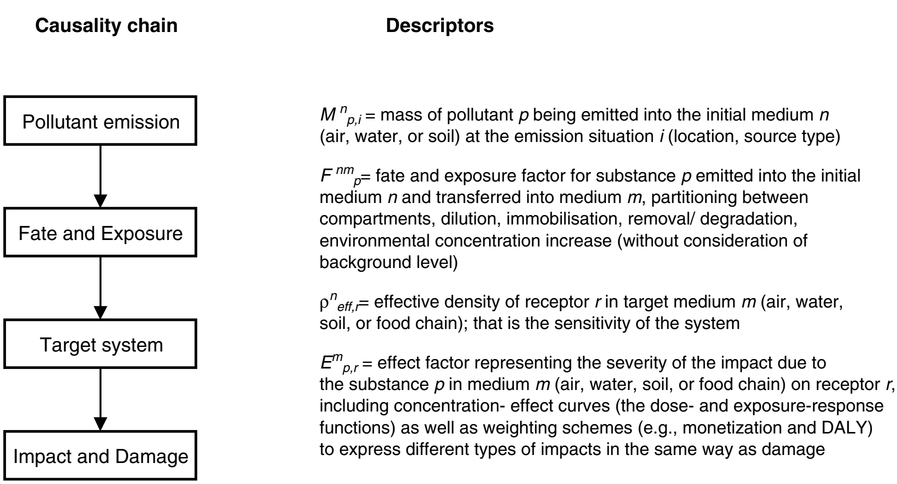

FIGURE 7.2 Relation of the factors used in the presented framework with the general cause–effect chain for the environmental impact of an emitted compound.

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. The site-dependent impact assessment can be carried out for various compartments using a multimedia fate and exposure model (see Chapter 4) or for only one release compartment n and one target compartment m by the application of a spatially explicit medium-specific model. In accordance with Potting (2000), we believe that the relevance of LCIA can be enhanced by the inclusion of a few general site parameters in the assessment procedure; we would call this site-dependent impact assessment.

For the effect analysis we propose to use the dose–response and exposure–response functions described in Chapter 4. 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. For a further discussion of this issue, see Crettaz et al. (2002), Nigge (2000) and Potting (2000).

7.3 STATISTICALLY DETERMINED GENERIC CLASSES OF AIRBORNE EMISSIONS

Considering only one pollutant, p, and one receptor, r, as well as one release compartment, n, and one target compartment, m, then $I^{nm}_{p,r,i} = I_i$ ($= \Delta RE_i/M$), the incremental receptor exposure per mass of pollutant emitted (receptors.($\mu g/m^3$).yr/kg), which represents the concentration increment multiplied by 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 (Nigge 2000):

$$I_i = \frac{1}{Q}\int_0^R r \int_0^{2\pi} \Delta c_i(r,\varphi)\, \rho_i(r,\varphi)\, d\varphi\, dr \qquad (7.5)$$

where

$Q = M_i/T$ is the constant emission rate (kg/yr) with M_i as mass of one pollutant (kg) emitted at the emission situation i and T as the duration of the emission (yr).

r is the radius (m).

R is the outer boundary of the modeling area (m).

$\Delta c_i(r,\varphi)$ is the concentration increment at a receptor point with the polar coordinates (r, φ) for the emission situation i ($\mu g/m^3$).

ρi(r, φ) is the receptor density at a receptor point with the polar coordinates (r, φ) for the emission situation i (receptors/m^2).

Generally speaking, the integration of Expression 7.5 should be carried out over the entire planet. Because most of the pollutants (even in the air compartment) do not disperse over the entire planet due to their residence time and dispersion characteristics in the environment, and because the calculation effort should be kept appropriate, R is chosen so that most of I_i is covered by the area. Another limitation is the spatial range of the models chosen, which often does 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 further the pollutant is transported and therefore the greater R must be chosen.

The idea of the methodology further developed and applied in this study is to define classes of emission situations i statistically, 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 should be kept small to enable easy handling.

On the one hand, neglect of the spatial distribution around the emission point, I, of the receptor density, $\rho(r,\varphi)$, is the main reason for the discordance between the potential impact results and the actual impacts. To be precise, this means, for example, that in conventional LCIA potentials the impact is the same for an air pollutant over the ocean as 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 must be based on statistical reasoning. For each class, receptor incremental exposures per mass of pollutant, ΔRE, must 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.

7.4 GENERIC CLASSES FOR HUMAN HEALTH EFFECTS

For the case of human toxicity impacts of airborne emissions, the receptors are the persons of a population and the release, as the target compartment is air. Thus, $\rho^{m}_{eff,r}$ corresponds to the population density and the receptor incremental exposures, $\Delta RE^{nm}_{p,r,i}$, are then called population incremental exposures to airborne pollutants ΔPE, expressed in units of (persons μg/m^3·yr). PE is also called pressure on human health (Nigge 2000).

Many laws and regulations with respect to emission limitation and pollution prevention exist for pollutants emitted into and solely transported by the air, so this study is confined to air as the only release and target compartment. Considering only human beings as receptors, for one pollutant Expression 7.4 can then be simplified to:

$$\Delta D_i = E \cdot M_i \cdot I_i \qquad (7.6)$$

where

ΔD_i is the incremental damage caused (damage) due to the exposure to one pollutant being emitted at the emission situation i.

E is the effect factor (damage/persons.(μg/m³).yr) representing the severity of the impact due to one pollutant.

M is the mass of one pollutant (kg) being emitted at the emission situation i.

I_i is the incremental population exposure per mass of one pollutant emitted at the emission situation i (persons.(μg/m³).yr/kg).

Expression 7.5 can be divided into two integrals accounting for the short-range dispersion and the long-range transport to the outer boundary of the modeling area R:

$I_{i,near}$ is the short-range contribution to the incremental receptor exposure per mass of one pollutant emitted at the emission situation i (persons.(μg/m³).yr/kg).

$I_{i,far}$ is the long-range contribution to the incremental receptor exposure per mass of one pollutant emitted at the emission situation i (persons.(μg/m³).yr/kg).

The reason for this procedure is that the concentration increment is usually highest within the first kilometers 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 kilometers. Consider, for example, that the population density often rapidly decreases from a big city to the countryside; therefore, the population exposure is subject to drastic changes within a few kilometers as well. The long-range contribution, however, only depends on the average receptor density of the region to which the pollutants are transported and is not particularly subject to changes on a local scale; the concentration increment is small due to dilution on transport and the concentration does not change very much with the distance. Long-range contributions are known and have been well studied for pollutants like SO_2 and NO_x (due to their importance for 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 such as Scandinavia or, in the U.S., the East Coast and the less populated Rocky Mountains. 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, which can vary significantly within a few kilometers for the different emission sites. It is therefore desirable to eliminate φ in order to simplify Expression 7.5. Nigge (2000) assumes 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, no direction is preferable for the spatial variation of the population density. Moreover, a simplification (Expression 7.7) can be introduced into Expression 7.5 (Nigge 2000):

$$\Delta c_i(r) \equiv \frac{1}{2}\pi \int_0^{2\pi} \Delta c_i(r, \varphi)\, d\varphi \tag{7.7}$$

According to Nigge (2000), Expression 7.5 then reads:

$$I_{C,near} = \frac{1}{Q} \int_0^{100km} r \int_0^{2\pi} \Delta c_C(r)\, \rho_C(r)\, 2\, \pi dr \tag{7.8}$$

In Expression 7.8, 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 Expression 7.8 does not refer to a single emission situation, i, but to a generic class of emission situations, statistically correlated with respect to dispersion conditions and population density and with 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:

$$I_C \equiv I_{C,near} + I_{far} \tag{7.9}$$

where

I_C is the incremental population exposure per mass of one pollutant emitted (persons.(μg/m^3).yr/kg) at the generic class of emission situations that are statistically correlated with respect to dispersion conditions and receptor density and have the same source characteristics.

In order to compute the impact indicator for each class, the following elements of Expression 7.8 must be calculated:

$\rho_C(r)$ is the radial receptor density (receptors/m^2) for each class (in our case population density; persons/m^2).
$\Delta c_C(r)$ is the 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.

7.4.1 Fate Analysis to Characterize Dispersion Conditions

This section discusses the transport of airborne pollutants from the emission source to the receptors. For the purpose of short-range dispersion modeling (dispersion up to the radius of 100 km around the stack), the program ISCST-3 (US EPA 1995), incorporated in the software BEEST (Beeline, 1998), can be used. The calculations for the long-range transport can be carried out using the program EcoSense (IER, 1998). Both programs are mentioned in Chapter 4 and will be applied to more cases in Chapter 8.

For modeling 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 must be used. Because 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 program requires input data as presented in Table 7.1, where the data used in the example of this chapter are also indicated. Test runs have shown that the concentration increment results are not

TABLE 7.1
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 15°
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 modeling area	Entirely flat
Release heights	5, 100, and 200 m
Exit temperature of stack	423.15 K
Stack diameter	5 m
Volume flow	$\dot{V} = 26862 \cdot 10^{0.0196 \cdot h_{stack}}$ (Nm³/h) (7.10)
Exit velocity	Function of volume flow and stack diameter
Emission mass flow	$\dot{M}_p = \dot{V} \cdot C_{threshold,p}$ (kg/h) (7.11) where $C_{threshold,\,p}$ is legal threshold concentration of pollutant p

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.

The remaining task is to determine a statistical distribution of wind speed, u, and stability class, s, for each class of meteorological conditions in the region under study. 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 that of wind speed classes are correlated. As a consequence, the only parameter required as additional input for using the impact indicator is the mean annual wind speed of the considered district.

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 by changing certain parameters of source characteristics. Therefore, statistical values for these parameters (volume flow and mass flow) must be defined.

A set of nine different industrial processes with stack heights ranging from 10 to 250 m were evaluated for the example 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 7.10, where $\dot{V}$ is the volume flow in (Nm^3/h) and h_{stack} the stack height in (m), and the regression coefficient r^2 equals 0.799. Expression 7.10 is only a rough approximation to calculate the volume flow. Nine processes is not at all a representative statistical number that allows making general conclusions.

The mass flow of each pollutant is obtained from the volume flow and the respective threshold of each pollutant according to Expression 7.11, where $\dot{M}_p$ is the mass flow of pollutant, p, and $c_{threshold,p}$ is the legal threshold concentration of pollutant p in flue gas. The threshold values for municipal waste incinerators are taken from the regulations valid for the region under study (in the example, the Catalan District 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 in this way 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, basing the mass flow also on statistical reasoning according to industry types should be attempted. An alternative would be 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, 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} (e.g., Nigge [2000]), the effective stack height is calculated for the indicators derived in this study (Table 7.2). The calculation of h_{eff} is carried out according to Israel (1994). The comparison of the results from different studies must be done with care, thoroughly checking the congruence of the applied algorithms for $h_{eff.}$

TABLE 7.2
Effective Stack Heights Depending on Wind Speed and Actual Stack Height

h_{stack} (α)	h_{eff} (m)		
	0 to 2 m/s	**2 to 3 m/s**	**3 to 4 m/s**
5	133	60	43
100	297	195	167
200	676	435	379

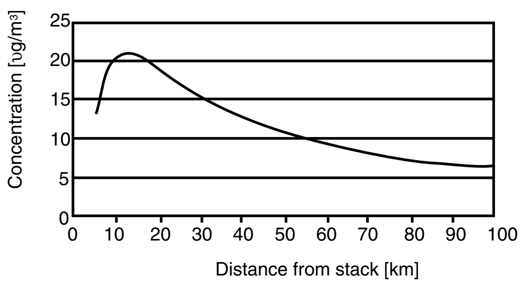

FIGURE 7.3 Concentration curve for PM_{10}, 100 km around the stack of 5 m height (average annual wind speed of 2.5 m/s).

Figure 7.3 shows an example of a concentration curve for PM_{10} and the wind speed class 2 to 3 m/s, 100 km around the stack of 5 m height. The resolution of the grid is higher close to the stack in order to represent the sharp decrease of concentration there.

7.4.2 Exposure Analysis to Determine the Population Density around the Emission Source

In order to make results comparable, the basic classification of regions and districts according to population density is taken from Nigge (2000), who calls the combination of region and districts *settlement structure classes*.

For the calculation of the radial population density, the radius of 100 km around each municipality in the region under study is considered, which corresponds to the modeling area of the short-range transport covered by the Gaussian dispersion model used. Annuli are formed in intervals of 10 km that lead to 10 annuli around each municipality. Each municipality will be counted to the respective annulus if its center lies within the considered annulus. The interval of 10 km is chosen assuming that the linear extension of the municipalities in the region under study is in the range of up to 10 km of both sides of the center of the municipality. Assuming that every municipality in the region under study has the shape of a circle, the maximum area allowed for one municipality lying in the center of all annuli (origin) is 314 km^2. If the biggest municipality in the region comprises less than 314 km^2, this assumption is valid.

The calculation of the radial population density has two problems: 1) the radial population density in districts close to the sea is not independent of the direction in which one looks (the population density at the coast falls down abruptly to $\rho = 0$ persons/km^2) and 2) no data might be available for the municipalities lying 100 km outside the regional borders of the region under study.

The first problem implies that the total population of all municipalities lying in the considered annulus is divided by the area of the annulus. Thus, the fact that a considerable area around the municipality has $\rho = 0$ persons/km^2 (municipalities close to the sea) is respected. Due to the absence of population living 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 population, data on a district level are used rather than the population data for the municipalities.

Because most districts are bigger than municipalities, uncertainties are introduced. As discussed earlier, the area of every municipality is assumed not to exceed a circle with a radius of 10 km if the municipality lies in the center of the circle. The average area of districts definitely exceeds this value. This means that the population density of the annuli is often determined by the entire population of one district, even though 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 because the average exceeding the center 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 the region under study, while the region is dealt with using a higher resolution, so the overall uncertainties related to the radial population density are considered to be quite low.

7.4.3 Effect Analysis to Transform Incremental Exposures into Damage Estimations

The effect analysis links the results of the fate and exposure analysis to the damage due to the emitted pollutant. This analysis is independent of the fate and exposure analysis and based on epidemiological and toxicological studies as well as on socioeconomic evaluation. See Chapter 3 and Chapter 4 for further details.

The effect factor represents the number of health incidences (like asthma or 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 because they 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 and Lopez (1996) is implemented. However, an economic valuation by external costs could also be applied easily, for example, the scheme used by the European Commission (1995). The DALY value not only depends on the pollutant and the type of disease, but also on the socioeconomic perspective of each person. Thompson et al. (1990) introduced the concept of the cultural theory for different perspectives; Hofstetter (1998) distinguished individualist, egalitarian and hierarchist cultural perspectives to represent the archetypes of socioeconomic behavior and related this to the DALY concept. Corresponding to age weighting for the different cultural perspectives, the economic evaluation discount rates of 0 and 3% can be chosen. An example of different factors for the conversion of population exposure values into damage estimates is given in Table 7.3.

7.5 SITE-DEPENDENT IMPACT ASSESSMENT IN THE METHODOLOGY OF ENVIRONMENTAL DAMAGE ESTIMATIONS FOR INDUSTRIAL PROCESS CHAINS

On the basis of the calculation of site-dependent impact assessment factors described beforehand, in this section the site-dependent method is introduced as a further element into the methodology of environmental damage estimations for industrial process chains outlined in the previous chapter. Site-dependent impact factors can

TABLE 7.3
Example of Conversion Factors between Impact Indicators and DALY and External Costs

Pollutant	DALY I	DALY E	DALY H	Costs[a]	Costs[b]
Acetaldehyde	2.7×10^{-7}	4.1×10^{-7}	4.1×10^{-7}	0.05	0.05
As	2.2×10^{-4}	3.5×10^{-4}	3.5×10^{-4}	43.1	31.5
BaP	1.3×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2,497	1,825
1,3-Butadiene	3.4×10^{-5}	5.2×10^{-5}	5.2×10^{-5}	6.6	6.6
Cd	2.7×10^{-4}	4.1×10^{-4}	4.1×10^{-4}	51.7	37.8
Ni	5.0×10^{-5}	7.8×10^{-5}	7.8×10^{-5}	10.3	7.3
NO_x	—	2.5×10^{-6}	—	0.27	0.43
SO_2	—	5.4×10^{-6}	5.4×10^{-6}	0.57	0.89
$PM_{2.5}$	2.8×10^{-4}	2.9×10^{-4}	2.9×10^{-4}	136.5	135.6
PM_{10}	1.7×10^{-4}	1.8×10^{-4}	1.8×10^{-4}	79.7	79.1
Nitrate	9.0×10^{-5}	1.8×10^{-4}	1.8×10^{-4}	79.7	79.1
Sulfate	2.8×10^{-4}	2.9×10^{-4}	2.9×10^{-4}	132.7	131.8

Notes: DALY: a/(persons·µg/m^3); external costs: U.S.$/(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%

be perfectly used in the mathematical framework developed and are particularly 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. Figure 7.4 gives such an overview of the different working steps for this site-dependent method.

For the calculation of the impact factors, first the considered region must 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 7.2 of this chapter.

In the fate and exposure analysis, each class is divided into the near (≤100 km) and the far (>100 km) contribution. For the near contribution, the radial concentration increment for each pollutant and the radial receptor density are calculated. The Gaussian dispersion model ISCST-3, as incorporated in BEEST, has been used in the application to Catalonia as transport model on the local scale. 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 by 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 specific region $I_{\text{total, pollutant, class\&region}}$ is determined.

By carrying out the consequence analysis factors of physical impact, parameters are obtained. Additionally, when the selected weighting and aggregation scheme is applied, the factors can also be expressed directly in the form of environmental costs or damage endpoint indicators. These two steps have been summed up as effect analysis in Section 7.4.3 of this chapter.

7.6 EXAMPLE: CALCULATION OF SITE-DEPENDENT IMPACT FACTORS FOR CATALONIA, SPAIN

In this study, the fate and exposure analysis is carried out for the Mediterranean region of Catalonia next to the sea, which significantly influences how to apply the presented method.

7.6.1 Wind Speed Classes

Harthan (2001) 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 is high (47.4%; Cunillera, 2000), with an average for 1996 to 1999 for all 14 stations considered. 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 area's situation as a whole. The formation of wind speed classes on the district level is required (see the overview of districts in Catalonia in Figure 7.5). The districts are assigned the following codes for the classes of wind

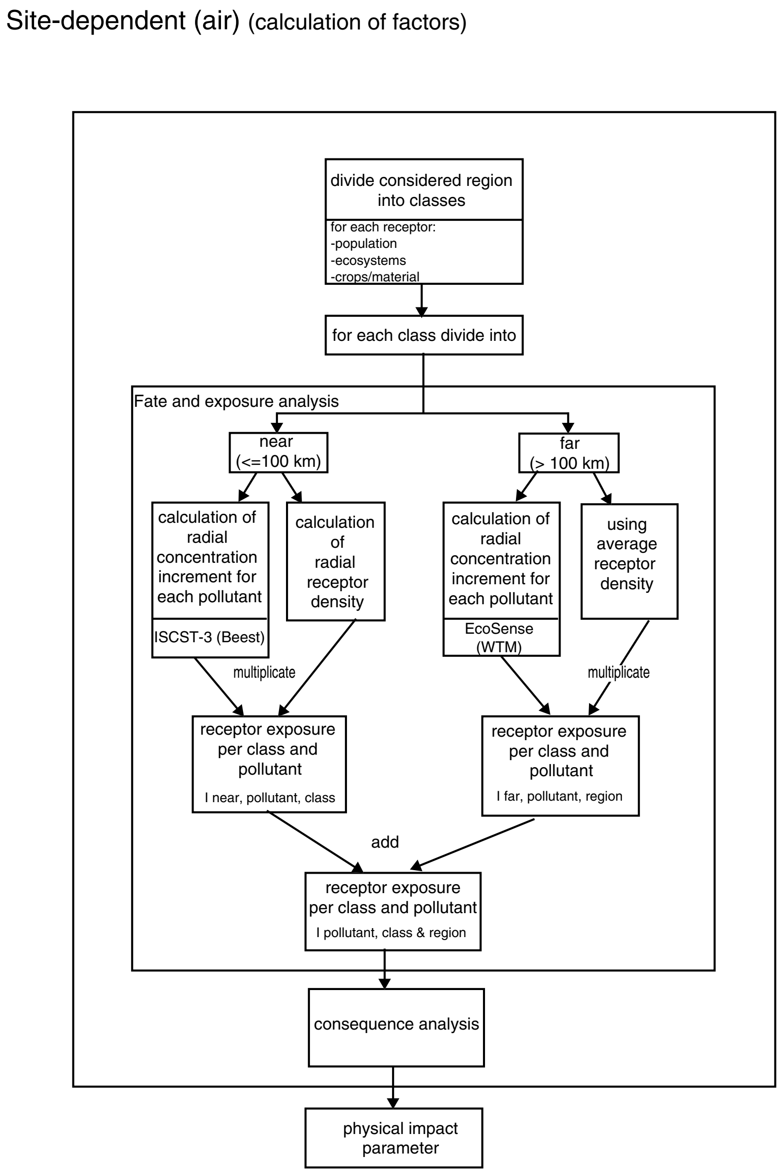

FIGURE 7.4 Site-dependent impact assessment for air emission.

speed: A: 0 to 2 m/s, B: 2 to 3 m/s, and C: 3 to 4 m/s. Because only wind speed data from 14 stations were provided in a detailed manner, average annual wind speed data for 48 stations in 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 7.5 and the portions of districts belonging to each class of meteorological conditions are shown in Table 7.4.

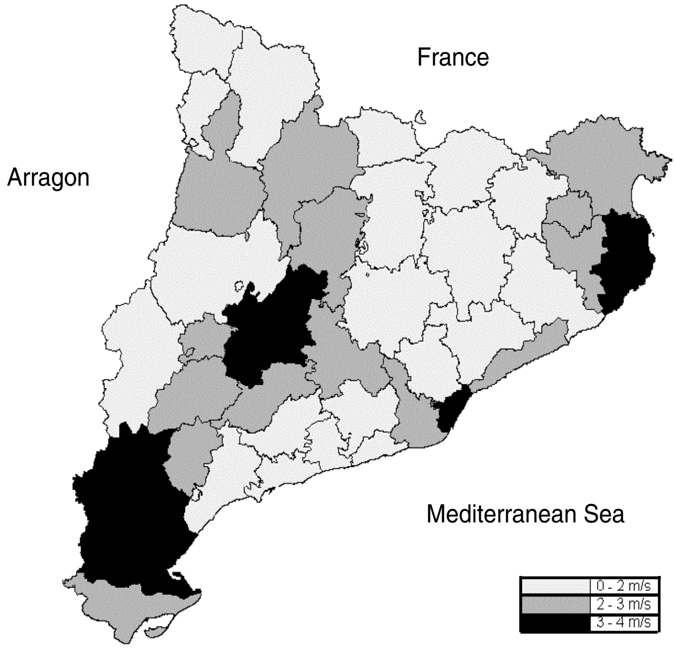

FIGURE 7.5 Classes of average wind speed for all districts in Catalonia (data of 1997 and 1998).

TABLE 7.4
Number and Share of Districts Belonging to Each Class of Wind Speed

Average wind speed class (m/s)	Districts	Share (%)
0 to 2	20	48.8
2 to 3	14	34.1
3 to 4	7	17.1
0 to 4	**31**	**100**

7.6.2 Settlement Structure Classes

As a region in the context of the example, the province level in Catalonia is considered. Furthermore, the districts are taken into account. In this way, four settlement structure classes have been identified to represent statistically the Catalonian situation sufficiently (Figure 7.6); this is characterized by the rapid decrease of the population from the city boundaries toward the countryside. For the purpose of this study, population data for Catalonia are taken from the MiraMon GIS (Pons and Maso 2000).

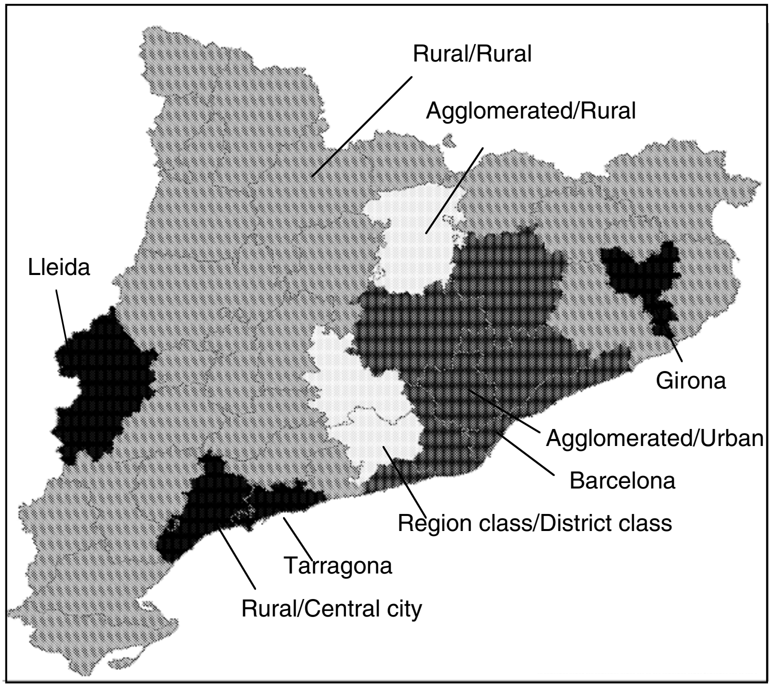

FIGURE 7.6 Settlement classes in Catalonia.

To calculate the radial population density for every municipality, the distance between each municipality must be calculated to assign the municipalities to the respective annuli. These data are also taken from MiraMon, which 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 center of the municipalities are calculated forming the average of all coordinates describing the outer limits. The latitudes and longitudes of the centers for the district outside Catalonia are converted into UTM coordinates using one of the converters available from the Internet.

After the distances between the municipalities are calculated, it is known which municipality lies in which annulus around a considered municipality. When data about the population of each municipality are used, 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 7.7 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.

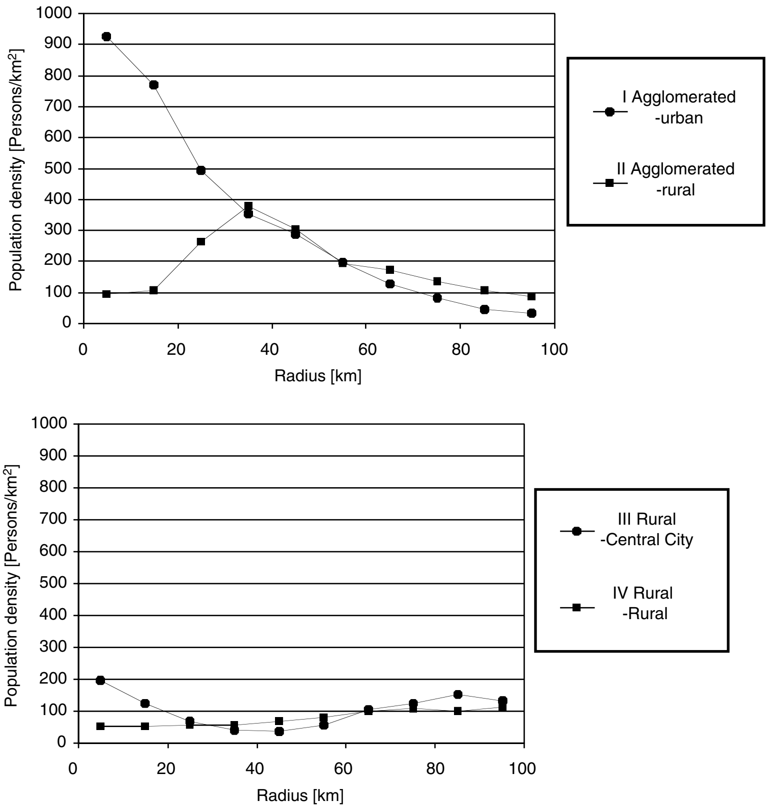

FIGURE 7.7 Radial population density distributions for the settlement structure classes in Catalonia.

The radial population density for the agglomerated region was calculated for the province of Barcelona, Catalonia. For the urban districts of the province, to which the city center of Barcelona belongs, the radial population density is decreasing rapidly from the center toward the outer radius of 100 km. The main reason for this behavior is the concentration of population within greater Barcelona, whereas outside the province the districts are scarcely populated.

The curve of the rural districts within the province of Barcelona shows the typical behavior of a rural region close to a big city. Within the first kilometers the population density is rather low. A peak of population density occurs between 30 and 40 km from a fictive stack in this district. This can be explained by the proximity of Barcelona city, which influences the radial population density for rural districts in the province of Barcelona also.

Within the rural regions of Girona, Lleida and Tarragona are districts classified as "central city" and "rural." For central cities to which the districts of Tarragonès, Baix Camp, Segrià and Gironès belong, the population density also decreases from the center toward the countryside. However, the population density is not as high in the center as for the respective districts in the agglomerated region. In the particular situation of Catalonia, after a minimum of between 40 and 50 kα, the population density reaches a local maximum between 80 and 90 km. A look at Figure 7.6 explains this behavior: all rural regions are located around the agglomerated region — especially the central cities in the rural regions, which are within a distance of 100 km 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-explanatory in rural districts. Toward r = 100 km, the population density increases; this explains the settlement structure of Catalonia. Statistically, a big city (central city) is within a distance of 100 km of each rural district in a rural area. Another look at Figure 7.6 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 km. 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 modeling at this distance.

7.6.3 Short- and Long-Range Exposure

The classes derived in the fate and exposure analysis are given combined class codes in order to enable easy discovery 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 and the effect analysis method chosen.

For the short-range exposure, calculations are carried out for every combination of pollutant, stack height and generic meteorological file. Concentration curves such as those in Figure 7.7 are overlapped with the curves of radial population density (Figure 7.7) for each generic class of population density and integrated over the modeling area (see Expression 7.8). The integration over the modeling area is done by summing up the products of concentration increment, population density and area for all values lying in the chosen grid.

For the long-range exposure, the concentration increment is calculated for eight grid cells of fictitious emission sites covering Catalonia, using the EcoSense software (IER 1998). The runs are carried out for stack heights of 5, 100 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. Multiplying 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, which 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 must be subtracted from the calculated population exposure. Because the short-range population exposure is calculated using the BEEST software, the population exposure within the radius of 100 km from the stack must be subtracted as well. For this purpose, the population exposure of the emission cell as well as the population exposure of part of the adjacent cells is subtracted.

After summing up the population exposure of each grid cell and 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, emission cell and stack height considered. Because the modeling 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 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 is 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 7.5). A linear regression is carried out and an approximation formula is calculated:

$$I_{far} = \exp\,(-0.2043 \bullet \ln \tau)^2 + 2.3916 \bullet \ln\,(-\,7.3174) \qquad (7.12)$$

where

I_{far} is the long-range contribution to the population exposure per mass of emitted pollutant (persons.μg/m^3.yr/kg).

τ is the atmospheric residence time (h).

TABLE 7.5
Diameter and Atmospheric Residence Time τ of Several Particles

Pollutant (diameter)	τ (h)
$PM_{2.5}$	135
PM_{10}	22
PM (d = 4 to 10 mm)	15
PM (d = 10 to 20 mm)	11
PM (d > 20 mm)	3

Source: Nigge, K.M., *Life-Cycle Assessment of Natural Gas Vehicles, Development and Application of Site-Dependent Impact Indicators*, Springer-Verlag, Berlin.

7.6.4 Overall Impact Indicators

The overall impact indicator, I_{total}, is the sum of I_{near} and I_{far}. Table 7.6 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 sulfate 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.yr/kg for nitrate and 0.13 persons.µg/m^3.yr/kg for sulfate, respectively.

Generally speaking, it can be said that the population exposure is smaller the lower the population density is. Another general correlation is that an increasing stack height leads to a decreasing impact indicator. Because Catalonia is quite populated in comparison with other regions in Spain, because every pollutant deposited on the Mediterranean does not lead to a human health effect via inhalation, and because the modeling area of EcoSense is limited to Europe (therefore neglecting harmful effects of Spanish emissions to North Africa, for instance), every molecule or particle going into the long-range transport favors the decrease of the overall population exposure.

Another obvious effect is 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 PM_{10}, $PM_{2.5}$, NO_x and SO_2, it can be seen that the values decrease from $PM_{2.5}$ over SO_2 and PM_{10} 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 a factor of 10 for $PM_{2.5}$ to a factor of 70 for PM_{10}. This can be explained with the fact that $PM_{2.5}$ accounts for a much greater long-range contribution to the population exposure than PM_{10} due to its long atmospheric residence time. Therefore, the lowest value of $PM_{2.5}$ 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 of pollutant using the conversion factors in Table 7.3.

7.6.5 Estimates for Adjacent Regions and Other Stack Heights

Process chains often comprise processes outside Catalonia, so an approximation formula for other regions in Spain is presented in the following expression. It is supposed that the long-range exposure for other regions in Spain does not vary significantly from the values in Catalonia. Therefore, it holds that:

$$I_{far,other\ regions} = I_{Catalonia}. \tag{7.11}$$

TABLE 7.6
Site-Dependent Human Health Impact Indicators for Several Pollutants and Stack Heights[a]

Pollutant Stack height Class		Acetal-dehyde			1,3-Butadiene			NO_x			SO_2			PM 2.5			PM 10		
District	**Wind**	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
		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

[a]Persons * µg/m^3 * year/kg
Notes: District population densities: I: agglomerated – urban; II: agglomerated – rural; III: rural – central city; IV: rural – rural; Cat: Catalonian average.
Wind speeds: A: 0 to 2 m/s; B: 2 to 3 m/s; C: 3 to 4 m/s

With respect to the short-range exposure, it is supposed that significant variations are 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 7.12. Because 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 7.4).

$$I_{near,\ other\ province} = I_{near,Catalonia} \cdot (\rho_{other\ province}/\rho_{Catalonia}) \tag{7.12}$$

Finally, it should be mentioned also that an interpolation can be carried out for emission heights different from the given ones.

7.7 CASE STUDY: SITE-DEPENDENT IMPACT INDICATORS USED FOR THE WASTE INCINERATION PROCESS CHAIN

In this section the impact indicators derived in Section 7.6.4 are applied to the industrial process chain of the case study, the municipal waste incineration (MSWI) plant of Tarragona (SIRUSA), and the related transport processes. Chapter 1 describes the data of the MSWI as well as the related transport processes. A comparison between the results for the applied impact indicators and the results for a midpoint indicator is provided and discussed. Figure 7.8 shows a regional map of Catalonia together with a physical map of Europe so that the localization of the study can be better visualized.

7.7.1 Goal and Scope Definition

Two situations and one scenario of the MSWI process chain are compared to estimate avoided environmental damages on human health.

Scenario 1 is the basis for all comparisons. It describes the SIRUSA incinerator in 1996, including 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.

Scenario 2 is the situation after the installation of an advanced acid gas removal system (AGRS) in 1997. In addition to the features in Scenario 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 Scenarios 1 and 2 are taken from the existing LCA study (STQ, 1998) (see Chapter 2).

As a third alternative, a future scenario has been created based on the modular model for the waste incineration process described briefly in Chapter 1. In addition to the features in Scenario 2, a fictitious DeNox system is installed to eliminate NO_x emissions. Ammonia for this purpose is transported from the industrial area of

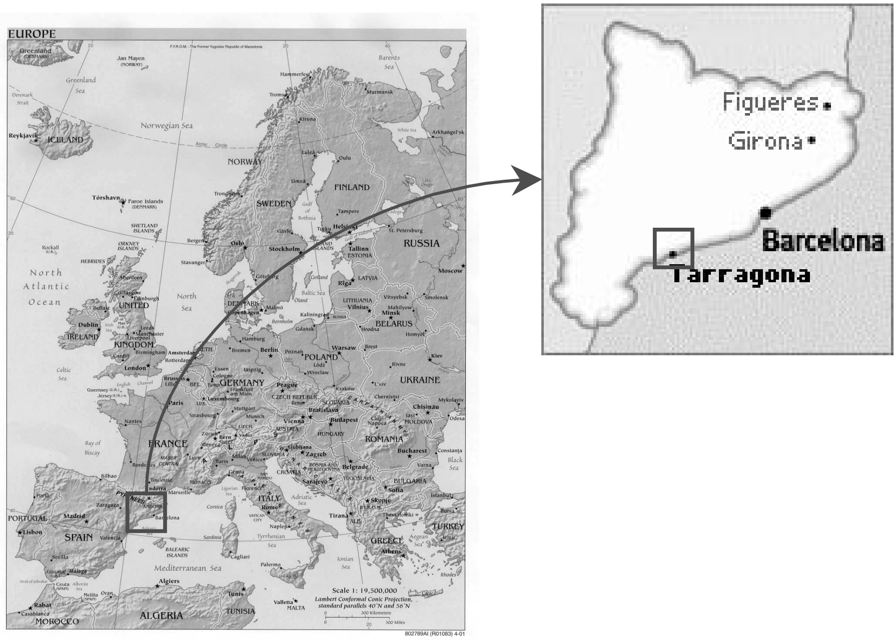

FIGURE 7.8 Physical map of Europe and regional map of Catalonia.

Tarragona to the waste incinerator. It should be stressed that, although this is a scenario with a nonvalidated MSWI model, it is introduced into this case study in order to discuss possible results of further improvement in 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 particular case study, the functional unit presented here is 1 year of treated waste (which equals 153,467 t). However, 1 t of treated waste as well as 1 TJ and 1 kWh of produced electricity could also be chosen.

According to the lessons learned in the application of the methodology of environmental damage estimations for industrial process chains in Chapter 6, the focus in this study was restricted to the transport and incineration processes. Therefore, it is not necessary to carry out a dominance analysis and no cut-off criteria need to be defined. However, to compare the results with midpoint indicators, the two LCIA methods with the human toxicity potential (HTP) explained in Chapter 3 are used: CML (Heijungs et al., 1992) and EDIP (Hausschild and 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. Because only the AoP (area of protection) of human health is taken into account here, no further aggregation needs to be carried out. The options of uncertainty analysis, environmental risk assessment, accidents and eco-efficiency are not considered.

7.7.2 LCI Analysis

In order to obtain the LCI data of this abbreviated waste incinerator process chain, the volume flow (Nm^3/a) of the waste incineration plant is multiplied by the unit emission files for the waste incinerator in the different cases, and the overall transport (tkm) is multiplied with the unit emission files for the transport. These unit emission files provide the emission of different air pollutants per Nm^3 and per tkm, respectively. The previously mentioned data and the unit emission files for SIRUSA can be found in the case study sections of previous chapters; the unit emission files for the transport processes are presented in Table 7.7. The table only lists the pollutants for which impact indicators are available that are considered crucial for environmental problems related to transport. The unit emission data for the MSWI for Scenarios 1 and 2 stem from the existing LCA study (STQ, 1998; see Chapter 2) and, for Scenario 3, from Hagelueken (2001; see Chapter 1).

The most obvious result is that the overall tkm increases significantly from Scenario 1 over Scenarios 2 and 3. Thus, the additional transport processes somehow compensate for the decrease of emissions due to the improvement of flue gas treatment. To what extent transport compensates for the damages avoided through more powerful gas treatment needs to be shown by environmental damage estimations.

7.7.3 LCIA Method

The HTP is calculated according to the CML method (Heijungs et al., 1992) and the EDIP method (Hauschild and Wenzel, 1998; the nomenclature of the latter is changed). Further explanations of the HTP can be found in Chapter 3. Table 7.8 presents the HTP results for the three alternatives.

7.7.4 Dominance Analysis and Spatial Differentiation

No dominance analysis is necessary for this application. The medium is air because the impact factors have been calculated for that medium. The pollutants are those for which site-dependent impact factors have been determined in Section 7.6.4. They correspond largely to the pollutants identified as predominant in Chapter 5. The

TABLE 7.7
Unit Emissions for Transport Processes

Pollutant	Emission (kg/tkm)
Benzo[a]pyrene	4.00×10^{-10}
Cd	2.10×10^{-9}
NMVOC	4.40×10^{-4}
NO_x	3.12×10^{-3}
PM_{10}	2.00×10^{-4}
SO_2	2.08×10^{-4}

Source: Kern, 2000

TABLE 7.8
HTP Results for the Three Alternatives

Process	CML (–/a)	%	EDIP (m^3/a)	%
Scenario 1				
MSWI	912,885	97.9	2.85×10^{12}	98.3
Transport	19,365	2.1	4.88×10^{10}	1.7
Total	932,251	100.0	2.90×10^{12}	100.0
Scenario 2				
MSWI	160,492	88. 0	8.95×10^{11}	94.2
Transport	21,885	12.0	5.51×10^{10}	5.8
Total	182,378	100.0	9.50×10^{11}	100.0
Scenario 3				
MSWI	43,361	64.5	2.31×10^{11}	79.3
Transport	23,912	35.5	6.02×10^{10}	20.7
Total	67,272	100.0	2.91×10^{11}	100.0

processes studied are those that have resulted as the most important in Chapter 6: the incineration and transport processes.

In addition to assigning the corresponding region to each process, in this application the transport process must be differentiated with respect to the districts crossed. This is an example of what is meant by the expression on mobile processes in the mathematical foundation of the methodology in Chapter 6. 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 7.9 describes the transport routes for every process and the districts crossed. Table 7.10 summarizes 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 Catalonia. (For the abbreviation of classes, see Section 7.6.) The emission height for the municipal waste incinerator is 50 m and for the transport processes is 5 m. Using the transport distances in each class from Table 7.11 and the tkm presented in Chapter 1, it can be calculated how many tkm refer to each class of population density and meteorological conditions. Multiplication by the unit emission file of transport leads to the spatially differentiated LCI, the eco-technology matrix.

7.7.5 Fate and Exposure and Consequence Analysis

Table 7.12 shows the damage-assigning matrix for DALY (egalitarian). It should be noted that this matrix only represents an example of the many possibilities according to the decision table described in Chapter 6. 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 7.6. Another damage-assigning matrix has also been established with the values of the impact indicator expressed with the unit

TABLE 7.9
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)

Notes: District population densities: I: agglomerated – urban; II: agglomerated – rural; III: rural – central city; IV: rural – rural; Cat: Catalonian average.
Wind speeds: A: 0 to 2 m/s; B: 2 to 3 m/s; C: 3 to 4 m/s

TABLE 7.10
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	—	—	6	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	6	24	17	—	—	—	—	—	—	**50**

Note: See Table 7.9 for abbreviations of classes.

TABLE 7.11
Eco-Technology Matrix for the Former Situation 1[a,b]

	MSWI	Transport[b]										
Pollutant	**IIIA[g]**	**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	1×10^{-5}	2×10^{-5}	7×10^{-6}	2×10^{-3}	4×10^{-5}	1×10^{-4}	1×10^{-4}	5×10^{-4}	8×10^{-5}	2×10^{-4}	9×10^{-5}
Cd	15	7×10^{-5}	1×10^{-4}	4×10^{-5}	9×10^{-3}	2×10^{-4}	6×10^{-4}	6×10^{-4}	2×10^{-3}	4×10^{-4}	1×10^{-3}	5×10^{-4}
PM_{10}	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
NO_x	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[c]	0	14	22	7	1,929	42	128	118	497	87	234	96
O_3[d]	142,333	101	158	53	13,677	296	906	838	3522	616	1658	684
Nitrate[e]	142,333	101	158	53	13,677	296	906	838	3522	616	1658	684
Sulphate[f]	602,643	7	11	4	912	20	60	56	235	41	111	46

[a] kg/a
[b] $h_{source} = 5$ m
[c] as NMVOC
[d] as NO_x
[e] as NO_x
[f] as SO_2
[g] $h_{stack} = 50$ m.

(persons.μg/m^3.yr/kg). It should be 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 $PM_{2.5}$. It is assumed that these substances are adsorbed on particles of $PM_{2.5}$ and therefore behave in the same terms of fate and exposure. However, the DALY value is specific for each substance because 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 sulfate do not differ because these are secondary pollutants for which country averages have been calculated. Only the value for the sulfate in the first row differs slightly, due to the stack height of the MSWI (50 m), which differs from the one of the transport processes (5 m stack height). The highest values for the primary pollutants appear in Madrid and the lowest in the small town of Huesca. These values are calculated according to the expression described in Section 7.6.5 for the transfer of impact factors to other regions. In comparison to Catalonia, Madrid is densely populated and Huesca is scarcely populated, so these results are obvious.

7.7.6 Damage Profile

Table 7.13 shows the damage matrix for Scenario 1 (former situation) resulting from the multiplication of the eco-technology matrix (Table 7.11) and the damage-assigning matrix (Table 7.12). If the first column is considered, the first box (the first element of the diagonal) represents the damage for the MSWI. 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 to be located in Madrid. Of course, it must be admitted that the impact indicator applied to this cell refers to a stack height of 5 m (transport) rather than to 50 m (MSWI), in which the damage would be twice as high as it is currently. Nevertheless, this provides an impressive demonstration of the importance of spatial differentiation.

Next, the diagonal elements of the damage matrix are added to obtain the damage profile, then the part corresponding to transport is compared with the value of the waste incinerator. In this way 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 Scenario 1 over Scenarios 2 and 3. On the one hand, this is due to the sharp decrease of the overall damage and, on the other, the decrease of damage due to the improvement of flue gas treatment is partly compensated for by the additional transport processes.

From the results of all scenarios 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

TABLE 7.12
Damage-Assigning Matrix for DALY (Egalitarian)[a]

Class	Pollutant										
	As	BaP	Cd	PM 10	Ni	NO_x	SO_2	O_3[c]	O_3[d]	Nitrate[e]	Sulfate[f]
III A[g]	5.4×10^{-4}	3.2×10^{-2}	6.5×10^{-4}	1.7×10^{-4}	1.2×10^{-4}	1.8×10^{-6}	6.1×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.7×10^{-5}
I B[b]	1.5×10^{-3}	8.9×10^{-2}	1.8×10^{-4}	6.4×10^{-4}	3.4×10^{-4}	8.3×10^{-6}	2.1×10^{-5}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
II A[b]	1.3×10^{-3}	7.7×10^{-2}	1.6×10^{-4}	5.4×10^{-4}	3.0×10^{-4}	5.9×10^{-6}	1.8×10^{-5}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
II B[b]	9.1×10^{-4}	5.3×10^{-2}	1.1×10^{-4}	3.4×10^{-4}	2.1×10^{-4}	4.0×10^{-6}	1.2×10^{-5}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
III A[b]	7.9×10^{-4}	4.6×10^{-2}	9.5×10^{-4}	2.9×10^{-4}	1.8×10^{-4}	3.1×10^{-6}	1.0×10^{-5}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
IV A[b]	6.6×10^{-4}	3.8×10^{-2}	7.9×10^{-4}	2.2×10^{-4}	1.5×10^{-4}	2.3×10^{-6}	8.0×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
IV B[b]	4.9×10^{-4}	2.8×10^{-2}	5.9×10^{-4}	1.4×10^{-4}	1.1×10^{-4}	1.6×10^{-6}	5.4×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
HU[b]	2.6×10^{-4}	1.5×10^{-2}	3.1×10^{-4}	3.4×10^{-5}	5.8×10^{-4}	4.4×10^{-7}	1.7×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
ZZ[b]	3.9×10^{-4}	2.2×10^{-2}	4.6×10^{-4}	9.6×10^{-5}	8.7×10^{-4}	1.2×10^{-6}	3.7×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
SO[b]	2.4×10^{-4}	1.4×10^{-2}	2.9×10^{-4}	2.7×10^{-5}	5.5×10^{-4}	3.6×10^{-7}	1.5×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
GU[b]	2.6×10^{-4}	1.5×10^{-2}	3.1×10^{-4}	3.4×10^{-5}	5.8×10^{-4}	4.4×10^{-7}	1.7×10^{-6}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}
M[b]	2.6×10^{-3}	1.5×10^{-1}	3.1×10^{-3}	1.1×10^{-3}	5.8×10^{-4}	1.3×10^{-5}	3.8×10^{-5}	1.0×10^{-6}	4.1×10^{-7}	4.5×10^{-5}	3.6×10^{-5}

[a] year∗year/kg
[b] $h_{source} = 5$ m
[c] as NMVOC
[d] as NO_x
[e] as NO_x
[f] as SO_2
[g] $h_{stack} = 50$ m

TABLE 7.13
Damage Matrix with DALY (Egalitarian) for Scenario 1

	MSWI	Transport										
III A[c]	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[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[b]	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[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[b]	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[b]	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[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[b]	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[b]	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[b]	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[b]	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[b]	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]year/(kg/year)
[b]h_{source} = 5 m
[c] h_{stack} = 50 m.

of pollutants emitted. The more toxic a substance is, the higher is the increase in DALY.

From the results using the endpoint indicators derived in this study on the one hand, and the HTP on the other hand, it seems that the HTP concept underestimates the environmental importance of transport. Although the share of transport in Scenario 1 is 2.1/1.7% for HTP (CML/EDIP), it is between 3.2 and 4.2% for the endpoint indicators derived in this study. Although the share of transport for the endpoint indicators reaches between 17.1 and 19.6% in Scenario 2, 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 Scenario 3: (35.5/20.7%) for HTP and 44.1 to 49.2% for the endpoint indicators, respectively.

The reason for the differences between the results for the HTP and the endpoint indicators is clear. The studied HTP methods consider the fate of the substances, but do not include exposure information. The environmental impact of transport is highly dependent on the location where it takes place, so 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 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 should only be carried out if transport does not increase significantly, which would worsen the overall environmental efficiency of the process chain.

It has been found that the chosen HTP methods underestimate transport in this case study and, therefore, does not identify very well the differences in the environmental impact for the two different processes considered. Figure 7.9 shows the results for all three cases — differentiated in waste incinerator and transport — for the population exposure, DALY (egalitarian) and HTP (EDIP).

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 that account for a low population exposure), the reduction for the endpoint 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 endpoint indicators as derived in this study is beneficial with respect to a gain of information for both purposes: the comparison of different scenarios and the comparison of different processes within one scenario.

7.7.7 Uncertainties in the Applied Framework

Essential for the validity of the presented application of site-dependent impact assessment is the question of 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 should be seen as a balance between the uncertainties introduced on the one hand and the handiness and feasibility of the applied method on the other. In general,

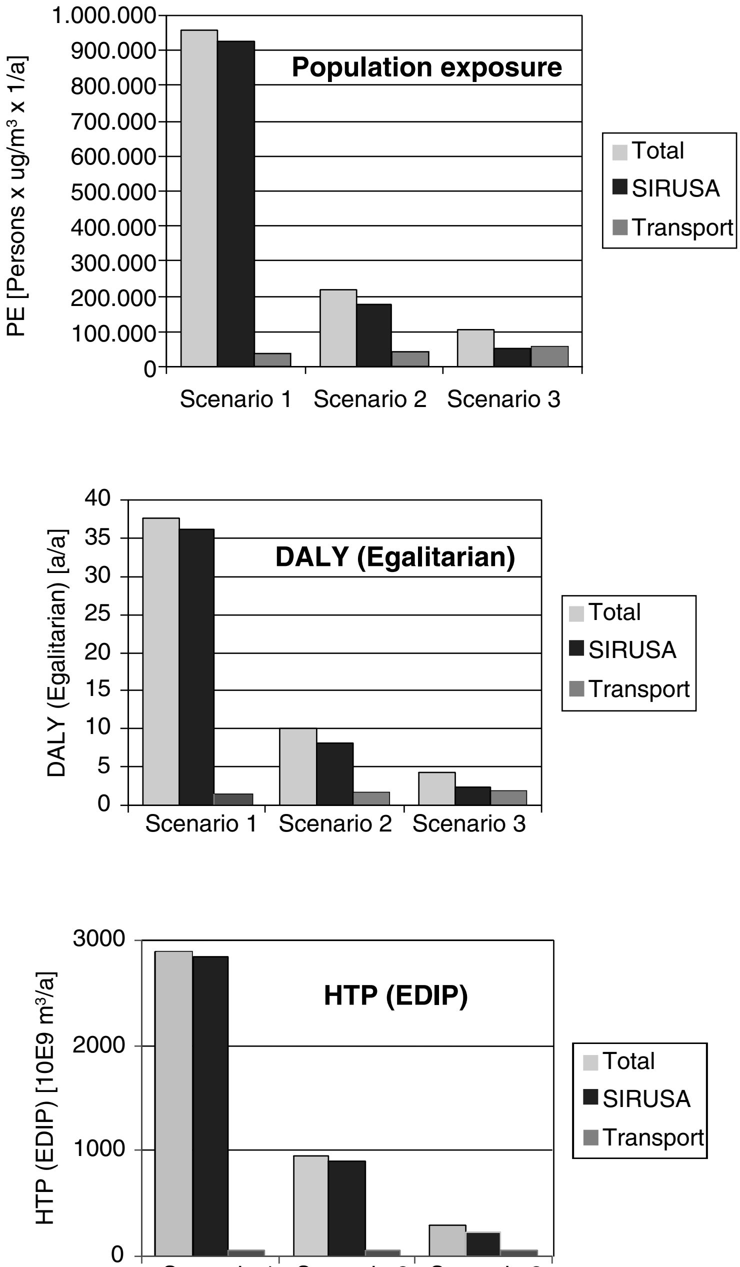

FIGURE 7.9 Results of the case study for all three scenarios for the population exposure — PE, DALY (egalitarian) and HTP (EDIP), where SIRUSA stands for the MSWI process.

it can be said that the larger the amount of data available, the lower the uncertainty behind the damage estimates is.

7.7.7.1 Goal and Scope Definition and Inventory Analysis

In the goal and scope definition, subjective elements exist in the form of the selection of the functional unit, which implies whether benefits are (or are not) 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 sensitivity analysis and the use of scenarios for the different options.

In the inventory analysis, the main types of uncertainties and variability analyzed by Monte Carlo simulation in Chapter 5 are those referring to parameters, including frequency of sampling, method of measurement (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 for all process simulations, the adequate estimation of physical properties that are not well established 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.

7.7.7.2 Dominance Analysis and Spatial Differentiation

The dominance analysis adds a level of uncertainty by reducing 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 should be said that the area under study, Catalonia, is rather small (compared to other administration units like Spain, European Union, or the U.S.). 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 and settlement structure, and according to meteorological conditions, must be done with special care.

The problem with choosing the outer boundaries of the classes (in this case provinces and districts) is that information on dispersion conditions (wind speed) as well as on the settlement structure (population density) is not always bound to administrational units. Meteorological conditions are particularly subject to geographic situations such as topography or latitude, while administrational units are generally linked to settlement structures. For instance, the district of Barcelonès comprises the city of Barcelona with a few other municipalities and represents an urban district in an agglomerated region.

However, not all district boundaries delimit the settlement structure that clearly. For example, the Tarragonès district (central city in rural region) comprises the city of Tarragona as well as rural municipalities, which contradicts the definition of each class. Administrational units also depend on history and political decisions because determination of the limits of the municipalities usually is not a recent decision. Thus, the choice of administrational units is problematic in terms of how representative settlement and dispersion conditions are. However, these limits are chosen because data are usually available for these administrational units; population data are available for municipalities, districts and provinces.

7.7.7.3 Fate and Exposure Assessment

Comparing the parameter uncertainty and variability involved in the LCI in relation to the impact pathway analysis (IPA; see Chapter 5) shows that uncertainties in the fate and exposure assessment as well as in the consequence/effect analysis are more important than those in the LCI analysis. This is due to the dose–response and exposure–response functions, weighting schemes, and fate models. Examples of fate models used in this study include dispersion software, multimedia models and long-range transport programs (as included in integrated impact assessment models).

The program BEEST, as well as EcoSense, was primarily developed for the calculation of concentration increments due to power plant emissions. This work, however, applies these programs 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 particularly, in order 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. Because substance data influence the dispersion of pollutants, uncertainties in these data are directly introduced into the results of the damage estimations. Another major source of uncertainty related to the fate models used is the choice of the modeling area and the grid size of the dispersion programs. In the long-range transport program WTM, the overall modeling area is confined to Europe (Eurogrid as implemented in EcoSense); a contribution to the population exposure 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. Because the long-range exposure does not depend very much on local variations, the resolution of 100×100 km seems to be appropriate for the use of Gaussian dispersion models. The grid for BEEST calculations is chosen so that the 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 program, it has been seen that the outcome for I_{near} is highly sensitive to the volume flow and the mass flow chosen. Although EcoSense allows the introduction of any mass or volume flow and still leads to the same results for I_{far}, the volume flow and the mass flow must be determined carefully for BEEST calculations. 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 must be stated, though, that the assumption for the volume flow is based only on a small number of industrial processes and that the used threshold is a political value.

As performed 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, volume flow and temperature are known. Moreover, it should be stated that a large variety of algorithms can calculate the effective emission height for different purposes. Generally speaking, it seems to be difficult to find one procedure of resolving these problems in defining source characteristics. This requires further research.

Another important source of uncertainty for the numerical results of the site-dependent human health impact factor for Catalonia is the fact that the models ISCST-3 in BEEST (short-range) and WTM in EcoSense (long-range) may be in poor accordance in terms of dispersion. As stated earlier, the dispersion results for BEEST are highly sensitive to the source characteristics. Moreover, these characteristics must be chosen so that the results at the outer boundary comply with the results calculated by the EcoSense modeling area that 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 modeling boundary with the BEEST program cannot be determined. This means that, 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 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 increase 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, which is accepted because one may suppose that the population density outside Catalonia is smaller. For this 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 short-range exposure. However, this is only a theoretical reasoning because no change in I_{far} can be observed.

The same problem applies to the stack height. Although the height has a great influence on the outcome for the short-range modeling (because it is assumed that the higher the stack, the more pollutants go into the long-range transport), the results for long-range modeling are quite insensitive to stack height. The longer the atmospheric residence time of the pollutant is, the greater the uncertainties to which problems related to wind speed and stack height lead. A pollutant with a short atmospheric residence time is deposited and decays mostly in the short-range modeling 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 for ERA, additional sources of uncertainty must be considered — in particular, site characteristics and transfer factors for transport among media. The site characteristics include human life-styles because different diets and media composition show 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. Also, a smaller number of years 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 no more data were 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 7.4.1, it must be said that the formation of classes of wind speed (0 to 2 m/s, 2 to 3 m/s and 3 to 4 m/s) has not been undertaken according to statistical reasoning. The limits of these classes are chosen according to the limits defined by the Deutscher Wetterdienst (DWD) and constitute a good compromise between the concept of classes and the meaningfulness of the classes. If broader class limits were chosen, the handiness of the results would increase 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 narrower class limits were chosen, 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 and a statistical reasoning combining several districts in one class would no longer be possible. The class limits chosen here 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.

Neglect of terrain elevations and precipitation is necessary in the site-dependent impact assessment due to the absence of statistical reasoning for this parameter; nevertheless, this leads to uncertainties. In particular, the concentration increment of particles calculated is overestimated because 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 temperature. Therefore, it is valid to choose the country average for all data sets and throughout the whole year. The neglect of wind direction leads to uncertainties, especially on a local level, because wind direction strongly influences 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 modeling. 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 because the biggest municipality in Catalonia has a smaller surface. However, reducing the resolution outside Catalonia to the district level increases the uncertainties. It is assumed that this is a reasonable procedure because it only concerns a limited number of adjacent districts to Catalonia and, with respect to working loads, this is a feasible way.

7.7.7.4 Consequence and Effect Analysis

One of the most important sources of uncertainties relates to the dose–response and exposure–response functions of pollutants further described in Chapter 4. These functions determine the consequence and effect analysis. Therefore, uncertainties due to these functions directly apply to the endpoint-related indicators or damage estimates (physical impacts such as cancer cases, as well as YLD, YOLL, DALY and external environmental costs). If one wants to avoid these uncertainties, the impact indicators can be applied as "pressure on human health." However, in order to take into account the differences in the toxicity of the pollutants and sensitivity to human health, dose–response and exposure–response must be considered. For instance, the EcoSense database offers a variety of dose–response functions that 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 judgment that directly influences the outcome. In order to increase the transparency and reduce the subjective influence by the methodology developer, this work offers several options. For uncertainties about YLD, YOLL and DALY values for several pollutants, see Hofstetter (1998). For more information on uncertainties in the evaluation of external environmental costs, see the EC (2000).

7.8 QUESTIONS AND EXERCISES

1. Why is a differentiation between near (<100 km) and far (>100 km) necessary for the impact assessment in the case of air emissions?
2. How is the fate analysis carried out for the presented site-dependent approach?
3. Explain how the exposure analysis is performed for the presented site-dependent approach.

4. What are the advantages and disadvantages of the presented effect analysis approaches?
5. Describe the procedure on how to apply site-dependent impact assessment in the framework of the overall methodology introduced in Chapter 6.
6. Discuss whether there are limits for the usefulness of end-of-pipe technologies in light of the results in the case study on the installation of different levels of advanced gas treatment systems.
7. What are considered to be the main uncertainties in environmental damage estimations for industrial process chains? What are the next steps ahead to reduce them?
8. Calculate the incremental damage caused in air by the emissions to air of 200 kg of a pollutant with a fate and exposure factor $F_p^{air,air} = 2.5 \times 10^{-5}$ and an effect factor $E_p^{air} = 1.2 \times 10^{-6}$.
9. Calculate the DALY for the egalitarian case and external cost of discount rate = 3% per person for a PM_{10} emission increment of 2×10^3 µg/m^3.
10. The incremental exposure per mass of pollutant for the population of an urban area of Massachusetts with a population density $\rho_2 = 550$ persons/km^2, is $I_{Massachusetts} = 0.18$. Calculate the incremental receptor exposure per mass of pollutant in a farming area of New Hampshire with a population density $\rho_1 = 90$ persons/km^2.

REFERENCES

Beeline Software, Inc. (1998), Beest for Windows. User's Manual. Beeline Software Inc., Asheville, NC.

Crettaz, P., Pennington, D., Brand, K., Rhomberg, L., and Jolliet, O. (2002), Assessing human health response in life-cycle assessment using ED10s and DALYs — part 1, cancer effects, *Risk Anal.*, 22(5), 929–944.

Cunillera, J. (2000), Algorithm for meteorological data obtained from the Meteorological Service of Catalonia, technical paper, Meteorological Service of Catalonia, Barcelona, Spain.

European Commission (EC) (1995), ExternE – Externalities of Energy, 6 vols., EUR 16520-162525. DG XII Science, Research and Development, Brussels.

EC (European Commission) (2000), *Externalities of Fuel Cycles*, vol. 7, methodology update, report of the JOULE research project ExternE (externalities of energy), EUR 19083. DG XII Science, Research and Development, Brussels.

Hagelueken, M. (2001), Effects of different models of municipal solid waste incinerators on the results of life-cycle assessments, student research project, cooperation of TU Berlin and Universitat Rovira i Virgili, Tarragona, Spain.

Harthan, R. (2001), Development of site-dependent impact indictors for human health effects of airborne pollutants in Catalonia, Spain, master thesism cooperation of TU Berlin and Universitat Rovira i Virgili, Tarragona, Spain.

Hauschild, M. and Wenzel, H. (1998), *Environmental Assessment of Products — Scientific Background*, (vol. 2), Chapman & Hall, London.

Heijungs, R., Guinée, J.B., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., and Wegener-Sleeswijk, A. (1992), Environmental life-cycle assessment of products — guide and backgrounds, Technical report, CML, University of Leiden, The Netherlands.

Hofstetter, P. (1998), *Perspectives in Life-Cycle Impact Assessment — a Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere*, Kluwer Academic Publishers, London.

Huijbregts, M.A.J. and Seppälä, J. (2000), Towards region-specific, European fate factors for airborne nitrogen compounds causing aquatic eutrophication, *Int. J. LCA*, 3(5), 65–67.

IER (Institut für Energiewirtschaft und Rationale Energieanwendung) (1998), EcoSense 2.1. Software, University of Stuttgart, Germany.

Israël, G. (1994), Transmission von Luftschadstoffen, educative report, Environmental Engineering Institute, TU Berlin.

Kern, M. (2000) Environmental Damage Estimations for the Process Chain of the Ash Treatment Plant TRISA. Master's thesis, Technical University of Berlin.

Kremer, M., Goldhan, G., and Heyde, M. (1998), Waste treatment in product-specific life-cycle inventories, *Int. J. LCA*, 3(1), 47–55.

Krewitt, W., Mayerhofer, P., Trukenmüller, A., and Friedrich, R. (1998), Application of the impact pathway analysis in the context of LCA, *Int. J. LCA*, 3, 86–94.

Krewitt, W., Trukenmueller, A., Bachmann, T., and Heck, T. (2001), Country-specific damage factors for air pollutants — a step towards site dependent LCIA, *Int. J. LCA*, 6(4), 199–210.

Manier, G. (1971), Untersuchungen über meteorologische Einflüsse auf die Ausbreitung von Schadgasen. Berichte des Deutschen Wetterdienstes Nr. 124, German Meteorological Service (DWD), Offenbach, Germany.

Moriguchi, Y. and Terazono, A. (2000), A simplified model for spatially differentiated impact assessment of air emissions, *Int. J. LCA*, 5, 281–286.

Murray, C.J.L. and Lopez, A.D. (1996), *The Global Burden of Disease*, vol. 1 of *Global Burden of Disease and Injury Series*, WHO, Harvard School of Public Health, World Bank. Harvard University Press, Boston.

Nigge, K.M. (2000), *Life-Cycle Assessment of Natural Gas Vehicles, Development and Application of Site-Dependent Impact Indicators*, Springer-Verlag, Berlin.

Pons, X. and Masó, J. (2000), The MiraMon map-reader — a new tool for the distribution and exploration of geographical information through internet or on CD, presented at 19th ISPRS Congress and Exhibition, Amsterdam.

Potting, J. (2000), Spatial differentiation in life-cycle impact assessment, Ph.D. thesis, University of Utrecht, The Netherlands.

Potting, J. and Blok, K. (1994), Spatial aspects of life-cycle assessment, in: SETAC-Europe, Eds., *Integrating Impact Assessment into LCA*, SETAC publication, Brussels.

Potting, J. and Hauschild, M. (1997), Predicted environmental impact and expected occurrence of actual environmental impact (part 2) — spatial differentiation in life-cycle assessment via the site-dependent characterization of environmental impact from emissions, *Int. J. LCA*, 2, 209–216.

Spadaro, J.V. and Rabl, A. (1999), Estimates of real damage from air pollution - site dependence and simple impact indices for LCA, *Int. J. LCA*, 4, 229–243.

STQ (Servei de Tecnologia Química) (1998), Análisis del ciclo de vida de la electricidad producida por la planta de incineración de residuos urbanos de Tarragona, technical report, Universitat Rovira i Virgili, Tarragona, Spain.

Thompson, M., Ellis, R., and Wildavsky, A. (1990), *Cultural Theory*, Westview Print, Boulder, CO.

Udo de Haes, H.A. (1996), *Towards a Methodology for Life-Cycle Impact Assessment*, SETAC-Europe publication, Brussels.

Udo de Haes, H.A., Joillet, O., Finnveden, G., Hauschild, M., Krewitt, W., and Mueller-Wenk, R. (1999), Best available practice regarding categories and category indicators in life-cycle impact assessment, background document for the Second Working Group on LCIA of SETAC-Europe, *Int. J. LCA*, 4, 66–74; 167–174.

US EPA (1995), Users' guide for the industrial source complex (isc3) — dispersion models, volumes I and II. EPA-454/B-95-003a, EPA-454/B-95-003b. Triangle Park, NC.

VDI (Verein Deutscher Ingenieure) (1992), *Dispersion of Pollutants in the Atmosphere, Gaussian Dispersion Model for Air Quality Management — Part 1, VDI*, Verlag, Düsseldorf, Germany.

Wenzel, H., Hauschild, M., and Alting, L. (1997), *Environmental Assessment of Products. Volume 1 – Methodology, Tools and Case Studies in Product Development*, Chapman & Hall, London.

8 Applications of Environmental Impact Analysis in Industrial Process Chains

With the contributions of
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8.1 INTRODUCTION

In this chapter, the methodologies explained previously will be applied in three other cases. These examples are pilot studies to further show the potential for integration of the tools for the life-cycle and site-specific impact assessment described in the book. We note the academic character of the projects, but we wanted to include them to demonstrate the strategy outlined in Chapter 6 for cases other than waste incineration.

In the first example, life-cycle assessment (LCA) and risk assessment are applied to the case of the landfill activity of mixed household solid waste. The use of the WISARD (waste integrated systems assessment for recovery and disposal) software model (Ecobalance, 1999) and database made it possible to perform the inventory analysis and impact assessment phases of the LCA. In the risk assessment, the ISCST-3 Gaussian dispersion model (U.S. EPA, 1995) was used for the calculation of the considered substance concentrations in the region; the CalTOX™ multimedia exposure model (DTSC, 2002) was used as well to evaluate the human health risk.

The second case shows the life-cycle assessment of a universal remote for television and video using the EIME (environmental information and management explorer) software model (Ecobalance, 1998) for the inventory analysis and impact assessment phases. The limits of site-specific studies are discussed in the context of this case study.

In the third case, an environmental impact analysis of an industrial separation process including life-cycle assessment, environmental risk assessment and impact pathway analysis is carried out. In this case, TEAM™ (tool for environmental analysis and management), an LCA software (Ecobalance, 1997), was applied to the potential impact assessment for the inventory analysis and impact assessment, while CalTOX and the integrated assessment model Ecosense (IER, 1997) were the tools used for the site-specific assessment.

8.2 EXAMPLE 1: LANDFILLING OF MIXED HOUSEHOLD SOLID WASTE (MHSW)

8.2.1 Introduction

Landfilling of mixed household solid waste (MHSW) was chosen as the first example for its similarity to the case study "Waste Incineration." The first applications example stems from the same life-cycle stage as the case study of the previous chapters: "Recycle — Waste Management." A further description of the typical features of that stage for a life-cycle assessment study can be found in Ciambrone (1997).

Landfill traditionally has been the most widely used method of waste treatment. However, the practice of landfilling has shown that the disposal of wastes that have not been pretreated causes emissions corresponding to those of a bioreactor. These emissions are considered high risk whereas landfills are ranked as the worst option in the waste hierarchy according to the pollution-preventing principle described in Chapter 1. In modern landfills the emissions are collected and treated as much as possible by biogas combustion and leaching effluents purifications.

In this example, an LCA is performed for an average Spanish landfill; then, a risk assessment is carried out for the substance that has the highest potential danger according to the human toxicity indicator used in the life-cycle impact assessment (LCIA). A fictitious site has been designated for this example which can be understood as a fully developed exercise for the sequence life-cycle assessment, dominance analysis for human toxicity potential and environmental risk assessment.

8.2.2 LCA for Landfilling of Mixed Household Solid Waste

8.2.2.1 Introduction

A case study of an LCA concerning the treatment of 50,000 t of mixed household solid waste (MHSW) in a medium-size Spanish landfill is performed. The data of the inventory include the consumption of raw materials and energy through the use of containers, collection and transport of wastes and the management of the landfill, and the corresponding emissions to air, water and soil. The following nine environmental impact categories have been considered in the impact assessment phase of the LCA: water eutrophication; depletion of nonrenewable resources; air acidification; greenhouse effect; aquatic eco-toxicity; human toxicity; terrestrial eco-toxicity; depletion of the ozone layer; and photochemical oxidant formation. The software model and database WISARD®[1] (waste integrated systems assessment for recovery and disposal) of Ecobalance Price Waterhouse & Coopers, mentioned in Chapter 2, has been used in the inventory analysis and impact assessment phases of the LCA (Figure 8.1).

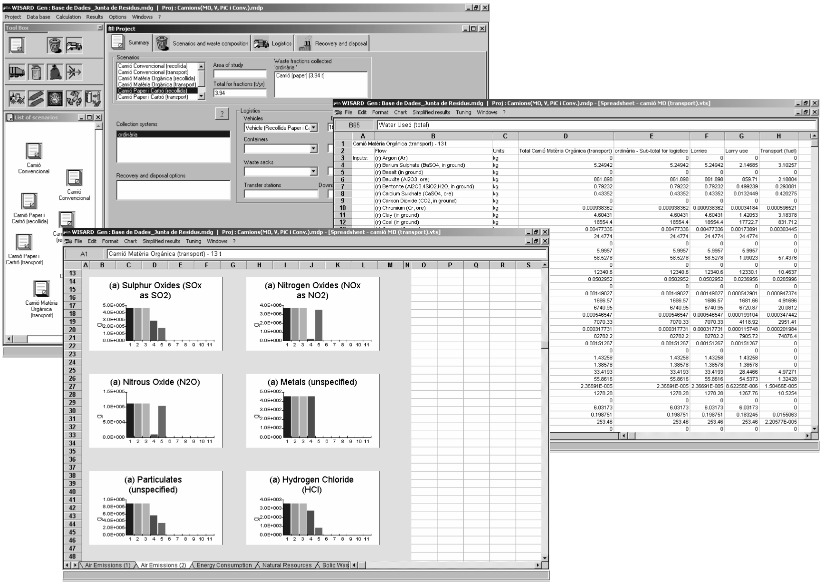

FIGURE 8.1 The software model and database WISARD.

8.2.2.2 Goal and Scope Definition

In this case study, the LCA methodology is applied to the treatment of MHSW in a typical medium-size Spanish landfill in order to evaluate the environmental impact of landfilling through the entire life-cycle associated with this waste management activity. Wastes have been considered the main input to the system; 50,000 t of MHSW produced and landfilled annually in the studied area are considered the functional unit of the LCA. Figure 8.2 shows the flow diagram of the system studied with the corresponding main inputs and outputs considered.

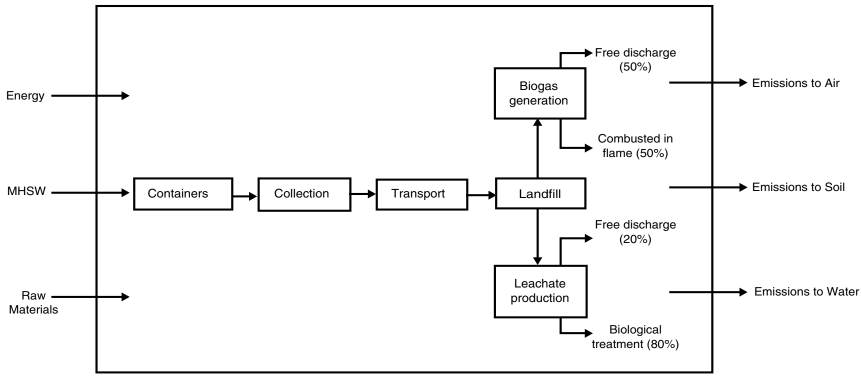

FIGURE 8.2 Flow diagram of landfilling activity.

8.2.2.3 Inventory Analysis

The considered landfill is situated in Spain. To carry out the inventory analysis, real and bibliographic Spanish data have been considered. For the collection data, the following main steps have been taken into account: use and maintenance of waste containers, collection and transport of wastes from the point of generation to the landfill, and management of the wastes in the landfill with partial (50%) collection and flaring of the generated biogas and with partial (80%) collection and biological treatment of the produced leachate.

The following section and Table 8.1 through Table 8.4 summarize the main input data introduced into the software for characterizing, from an environmental point of view, all the elements directly and indirectly implicated in the landfilling activity (containers, vehicles, landfill and MHSW).

Containers:
General — capacity: 1.1 m^3; lifetime: 6 years; and container load rate: 75%
Composition — HDPE: 59 kg; steel: 4 kg; rubber: 1.5 kg; and aluminum: 0.5 kg
Cleaning — frequency: 1 per week; water: 30 l; and cleaning product: 0.07 kg

Collection and transport vehicles:
General — load: 21.5 m^3; compacting ratio: 6.5%; lifetime: 185,000 km; collection: daily
Vehicle body composition — steel: 13 t; PP: 500 kg; aluminum: 250 kg; glass: 50 kg
Cleaning — frequency: 15 times/1000 km; water: 300 l; cleaning product: 5 kg
Diesel consumption — collection: 116 l/100 km; transport: 50 l/100 km
Average distances — collection: 9 km; transport: 18 km
Oil consumption — motor oil: 108 l/10,000 km; hydraulic oil: 165 l/10,000 km

Landfill:
General — tonnage stored: 4167 t/month; total storage capacity: 2,000,000 m^3
Construction — clay: 19,000 t; sand and gravel: 12,000 t; HDPE: 150 t; diesel: 30,000 l

TABLE 8.1
Characterization and Composition of MHSW

	Weight (%)	Density (kg/m³)	Moisture (%)	Biogas (kg/kg)
Organics	37.62	600	77.50	0.63
Paper/cardboard	21.01	183	22.32	0.29
Glass	8.21	300	1.84	0
Plastics	15.81	20	22.50	0
Metals	5.31	83	7.70	0
Others	12.04	94	20.93	0.18

TABLE 8.2
Vehicle Emission Factors during Collection and Transport[a]

Flow	During collection	During transport
Ammonia (NH_3)	1.2	0.5
Carbon dioxide (CO_2)	306,240	132,000
Carbon monoxide (CO)	1,983	855
Methane (CH_4)	19	8
Nitrogen oxides (NO_x)	4,420	1,905
Nitrous oxide (N_2O)	131	57
Non methane hydrocarbons	726	313
Particulates	415	179
Sulfur oxides (SO_x)	88	38

[a]Grams per 100 km.

TABLE 8.3
Main Biogas Air Emissions of MHSW[a]

Flow	Biogas free discharge	Biogas combusted in flare
Carbon dioxide (CO_2, biomass)	126,977	199,150
Carbon monoxide (CO)	73.8	89.4
Hydrocarbons (CxHy)	176.3	97.0
Methane (CH_4)	39,332	—
Nitrogen oxides (NO_x)	—	10.6
Sulfur oxides (SO_x)	—	3.7

[a]Grams per tonne.

TABLE 8.4
Main Leachate Composition of MHSW[a]

Flow	Leachate free discharge
Ammonia (as N)	280.4
BOD_5 (biochemical oxygen demand)	1224.6
Chlorides (Cl^-)	1010.4
Metals	149.6
Nitrate (NO^{3-})	1.4
Nitrite (NO^{2-})	0.3
Phosphates (as P)	0.3
Potassium (K^+)	345.1
Sodium (Na^+)	618.7
Sulphates (SO_4^{2-})	117.2

[a]Grams per tonne.

Operation (per month) — sand: 350 t, electricity: 2250 kWh; diesel: 4500 l; oil 100 l
Biogas — free discharge: 50%; combusted in flare: 50%
Leachate — total production: 55 l/t; free discharge: 20%; biologically treated: 80%
Final capping — clay: 13,500 t; top soil: 6000 t; diesel: 13,500 l

After introducing the previously presented input data in the software model (WISARD), the eco-balance or inventory of the landfilling activity studied (50,000 t of MHSW) has been automatically calculated. The complete inventory was integrated by 307 environmental loads (inputs and outputs): energy and raw materials consumed and emissions to air, water and soil. The main inputs and outputs of the system that contribute more than 5% in any of the environmental impact indicators subsequently considered are shown in Table 8.5.

8.2.2.4 Impact Assessment

To carry out this phase of the LCA case study, the following nine impact categories have been considered: water eutrophication (WE), depletion of nonrenewable resources (DNRR), air acidification (AA), greenhouse effect (GE), aquatic eco-toxicity (AE), human toxicity (HT), terrestrial eco-toxicity (TE), depletion of the ozone layer (DOL) and photochemical oxidant formation (POF). The specific environmental impact indicators used in each of the environmental impact categories mentioned are shown in Table 8.6.

The environmental loads (inputs and outputs) previously inventoried have been classified under their corresponding environmental impact indicator, following the classification criteria specified in these indicators. Characterization factors have then been used to prioritize the environmental loads or, in other words, to quantify the potential contribution of each environmental load in the different impact indicators. These characterization factors are pre-established for each impact indicator. Finally, the corresponding potential contributions have been determined by multiplying the mass of the environmental loads in the inventory by these factors (for example, 54,533 g trichloroethane × 1200 g equiv. of 1-4-dichlorobenzene/g trichloroethane = 65,439,600 g equiv. of 1-4-dichlorobenzene).

The inventoried environmental loads are classified under their corresponding impact indicators with their respective characterization factors in Table 8.7. The total potential contribution of each environmental load in each impact category, as well as its corresponding contribution percentage for the four stages of the landfilling activity (containers, collection, transport and landfilling), are presented.

8.2.2.5 Interpretation

As can be seen in Table 8.8 and Figure 8.3, landfilling is the stage that contributes more to WE, AA, GE, AE, HT, DOL and POF, indicators being the transport of wastes to the landfill, the main contributor to DNRR and TE indicators.

Direct and indirect consumption of scarce natural nonrenewable resources contributes significantly to DNRR indicator. During the construction of vehicle bodies for collecting and transporting wastes to the landfill, these scarce raw materials are

TABLE 8.5
Inventory of Landfilling Activity[a]

Flow	Units	TOTAL	Containers	Collection	Transport	Landfilling
			Inputs			
(r) Natural gas (in ground)	kg	26,749	9,526	3,896	5,024	8,303
(r) Oil (in ground)	kg	194,463	8,935	64,523	58,819	62,186
(r) Phosphate rock (in ground)	kg	16,148	0	1	1	16,147
(r) Zinc (Zn, ore)	kg	289	18	90	180	0
			Outputs			
(a) Aromatic hydrocarbons (unspecified)	g	783,231	5	37	74	783,115
(a) Cadmium (Cd)	g	24	1	7	12	4
(a) CFC 12 (CCl_2F_2)	g	46,624	0	0	0	46,624
(a) Ethylene (C_2H_4)	g	1,727,757	687	1,249	1,316	1,724,505
(a) Hydrocarbons (except methane)	g	2,512,440	6,211	899,164	801,240	805,825
(a) Lead (Pb)	g	635	34	193	380	27
(a) Mercury (Hg)	g	10	2	3	5	1
(a) Methane (CH_4)	g	989,662,857	105,419	2,198,955	2,063,482	985,295,000
(a) Nitrogen oxides (NO_x as NO_2)	g	8,504,624	122,639	2,944,386	2,629,864	2,807,735
(a) Sulfur oxides (SO_x as SO_2)	g	1,209,893	83,348	251,866	333,398	541,280
(a) Trichloroethane (1,1,1-CH_3CCl_3)	g	54,533	0	0	0	54,533
(a) Zinc (Zn)	g	2,899	184	898	1,791	26
(w) Ammonia (NH_4^+, NH_3, as N)	g	2,834,934	166	5,243	4,590	2,824,935
(w) Cadmium (Cd^{++})	g	171	10	5	5	151
(w) Chromium (Cr III)	g	1,074	1	3	5	1,065
(w) Mercury (Hg^+, Hg^{++})	g	65	62	0	1	2

[a]50,000 t MHSW.
(r): natural resources; (a): emission to air; (w): emission to water.

TABLE 8.6
Environmental Impact Categories and Indicators Used in WISARD

Acronym	Category	Indicator — source (year)
WE	Water eutrophication	CML (water) — CML Leiden University (1992)
DNRR	Depletion of nonrenewable resources	EB (R*Y) — Ecobalance USA (1998)
AA	Air acidification	ETH (air acidification) — Swiss Federal Institute of Technology, Zurich (1995)
GE	Greenhouse effect	IPCC (direct, 20 years) — International Panel on Climate Change (1998)
AE	Aquatic ecotoxicity	USES 1.0 — CML Leiden University (1996)
HT	Human toxicity	USES 1.0 — CML Leiden University (1996)
TE	Terrestrial ecotoxicity	USES 1.0 — CML Leiden University (1996)
DOL	Depletion of the ozone layer	WMO (average) – World Meteorological Organization (1998)
POF	Photochemical oxidant formation	WMO (average) — United Nations Economic Commission for Europe (1991)

TABLE 8.7
Impact Assessment of Landfilling Activity[a]

Impacts	Characterization factors	TOTAL	Containers (%)	Collection (%)	Transport (%)	Landfilling (%)
Water eutrophication (g eq. PO_4)	*	1,206,626	0.2	0.5	0.7	98.6
(w) Ammonia (NH_4^+, NH_3, as N)	0.42	1,190,672	0.0	0.2	0.2	99.6
Depletion of nonrenewable resources (yr^{-1})	*	28,995	8.6	27.9	40.8	22.7
(r) Zinc (Zn, ore)	40.29	11,626	6.4	31.2	62.4	0.0
(r) Oil (in ground)	0.0557	10,832	4.6	33.2	30.2	32.0
(r) Natural gas (in ground)	0.117	3,130	35.6	14.6	18.8	31.0
(r) Phosphate rock (in ground)	0.115	1,857	0.0	0.0	0.0	100.0
Air acidification (g equiv. H^+)	*	228,251	2.3	31.6	29.9	36.2
(a) Nitrogen oxides (NOx as NO_2)	0.0217	184,883	1.4	34.6	30.9	33.0
(a) Sulfur oxides (SOx as SO_2)	0.0313	37,809	6.9	20.8	27.6	44.7
Greenhouse effect (direct, 20 years) (g equiv. CO_2)	*	64,802,966,673	0.1	0.6	0.6	98.7
(a) Methane (CH_4)	64	63,338,422,828	0.0	0.2	0.2	99.6
Aquatic eco-toxicity (g equiv. 1-4-dichlorobenzene)	*	1,157,851	7.8	7.7	12.4	72.1
(w) Cadmium (Cd^{++})	4,500	769,211	6.1	2.8	2.7	88.4
(a) Mercury (Hg)	16,000	161,391	17.6	25.1	48.5	8.7
(w) Chromium (Cr III)	84	90,228	0.1	0.3	0.5	99.2
Human toxicity (g equiv. 1-4-dichlorobenzene)	*	173,132,057	2.1	9.3	16.8	71.8
(a) Trichloroethane (1,1,1-CH_3CCl_3)	1,200	65,439,600	0.0	0.0	0.0	100.0
(w) Ammonia (NH_4^+, NH_3, as N)	17	48,193,881	0.0	0.2	0.2	99.6
(a) Lead (Pb)	67000	42,548,314	5.4	30.4	59.9	4.3
Terrestrial eco-toxicity (g eq. 1-4-dichlorobenzene)	—	6,271,415,948	13.3	27.0	46.7	13.1

-- continued

TABLE 8.7 (CONTINUED)
Impact Assessment of Landfilling Activity[a]

Impacts	Characterization factors	TOTAL	Containers (%)	Collection (%)	Transport (%)	Landfilling (%)
(a) Cadmium (Cd)	130,000,000	3,159,589,505	3.9	30.3	49.8	16.0
(a) Zinc (Zn)	660,000	1,913,290,536	6.4	31.0	61.8	0.9
(w) Mercury (Hg^{+}, Hg^{++})	8,200,000	532,756,761	95.5	0.5	1.0	3.0
Depletion of the ozone layer (average) (g equiv. CFC-11)	*	40,031	0.0	0.4	0.3	99.3
(a) CFC 12 (CCl_2F_2)	0.82	38,231	0.0	0.0	0.0	100.0
Photochemical oxidant formation (average) (g equiv. ethylene)	*	11,687,352	0.8	3.6	3.2	92.5
(a) Methane (CH_4)	0.007	6,927,640	0.0	0.2	0.2	99.6
(a) Ethylene (C_2H_4)	1	1,727,757	0.0	0.1	0.1	99.8
(a) Hydrocarbons (except methane)	0.416	1,045,175	0.2	35.8	31.9	32.1
(a) Aromatic hydrocarbons (unspecified)	0.761	596,039	0.0	0.0	0.0	100.0

[a]50,000 t MHSW.
(r): natural resources; (a): emission to air; (w): emission to water.

TABLE 8.8
Stages' Contribution (%) in Each Impact Indicator

Impacts	Containers	Collection	Transport	Landfilling
WE	0.2	0.5	0.7	**98.6**
DNRR	8.6	27.9	**40.8**	22.7
AA	2.3	31.6	29.9	**36.2**
GE	0.1	0.6	0.6	**98.7**
AE	7.8	7.7	12.4	**72.1**
HT	2.1	9.3	16.8	**71.8**
TE	13.3	27.0	**46.7**	13.1
DOL	0.0	0.4	0.3	**99.3**
POF	0.8	3.6	3.2	**92.5**

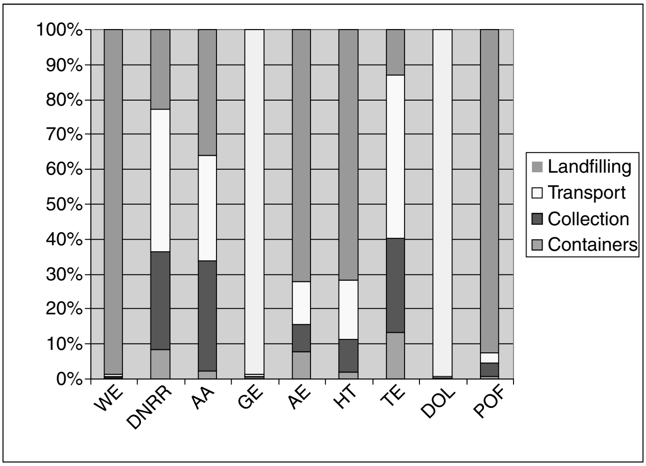

FIGURE 8.3 Stages' contribution (%) in each impact indicator.

directly and indirectly consumed (e.g., zinc, oil, natural gas, etc.), transport and collection of wastes being the main contributors to this indicator.

Free discharge of biogas, estimated at 50% of the total generated in the landfill, and its associated methane air emissions is the main contaminant contributor to the GE indicator. Biogas is also composed by other contaminants such as CFC 12, which is the main contributor to the DOL indicator, and ethylene, hydrocarbons and aromatic hydrocarbons are the main contributors to the POF indicator. Also, free discharge of biogas (estimated at 50%) and leachate (estimated at 20%) have other associated atmospheric and water emissions; the presence of trichloroethane in biogas and ammonia in leachate are the main contaminant contributors to the HT indicator and cadmium and chromium in lechate are the main contaminant contributors to the AE indicator.

The combustion in a flare of the biogas collected (estimated at 50% of the total generated) and the consequent air emissions of nitrogen and sulfur oxides are the main contributors to the AA indicator. Also, significant nitrogen and sulfur emissions are produced during collection and transport of wastes to the landfill.

The potential risk of leachate free discharges into soil (estimated at 20% of the total generated) significantly increases the risk of WE; ammonia is the main contaminant contributor to this indicator. In the TE indicator, the main contributors are cadmium and zinc air emissions generated primarily during the construction of vehicle bodies but also during diesel oil production and its combustion during collection and transport of wastes. Mercury in water, generated during the container manufacturing process, is also an important contaminant contributor to this indicator.

As a final comment, it should be mentioned that impact assessment interpretation is a particularly difficult task in a landfilling activity, mainly because of the temporary dependence of its environmental consequences. In landfilling, the chemical life of wastes can be estimated approximately 30 years before being considered an inert waste, which implies that biogas and leachate emissions will vary in quantity and composition during this period. In this case study, any emission to air, water and soil produced along the chemical life of wastes (30 years) has been directly assigned to the functional unit considered (50,000 t of MHSW).

8.2.3 Selection of Pollutant for Site-Specific Impact Assessment in Landfilling Example

As pointed out in the introduction to this example, a dominance analysis is to be carried out for the human toxicity indicator. Table 8.7 indicates that the air emissions of 1,1,1-trichloroethane ($C_2H_3Cl_2$) contribute most to the human toxicity indicator. Therefore, the next section presents an exposure risk assessment for 1,1,1-tricholoroethane from landfilling of MHSW.

8.2.4 Risk Assessment of the 1,1,1-Trichloroethane Emissions from Landfilling of Mixed Household Solid Waste

8.2.4.1 Introduction

Pollutants emitted to the atmosphere are transported through it and may subsequently impact environmental media (i.e., soil, water and vegetation) near the plant, resulting in a number of potential sources for human exposure. Because the landfill's emissions of trichloroethane were the main contaminant contribution to the human toxicity indicator according to the LCA applied in Example 1.1(Table 8.7 and Section 8.2.3), the aim of this exercise is to calculate the incremental lifetime risk due to the 1,1,1-trichloroethane ($C_2H_3Cl_3$) emission of the landfill of MHSW for the residents living in the surroundings of the plant. In order to obtain this, the air 1,1,1-trichloroethane concentrations in the vicinity of the landfill were quantified by application of a Gaussian dispersion model (ISCST-3). Then, human health risks due to 1,1,1-trichloroethane emissions from the landfill were calculated by application of a multimedia exposure model (CalTOX).

8.2.4.2 Air Dispersion Models

Pollutants emitted to the air are dispersed depending on the meteorological conditions, e.g., wind speed and solar radiation, and the characteristics of the region, e.g., elevations of the terrain and land use. Accordingly, they occur in the atmosphere in areas even farther from the emission source.

The atmosphere is the starting medium of the environmental fate and transport of pollutants of the landfill air emissions. The pollutants are dispersed in the air depending on the meteorological and topographic conditions in the location of the emission source and can be transported in the atmosphere over large distances. However, a portion of the pollutants is deposited in the surrounding area of the emission source and accumulates in other environmental media such as soil, surface water or vegetation. If air concentrations of pollutants cannot be determined with measurements, they can be calculated using air dispersion models, which simulate the atmospheric dispersion using meteorological and topographic information of the considered region. In the current exercise, the air dispersion of emitted 1,1,1-trichloroethane was modeled for the surroundings of the landfilling of MHSW. In this risk assessment, the ISCST3 model, described in Chapter 4, was used to estimate air concentration dispersion of the 1,1,1-trichloroethane emissions.

8.2.4.3 Data for the ISCST3 Model

ISCST3 is based on a Gaussian plume model (see Chapter 4). It is most common to compute ambient air concentrations and surface deposition fluxes at specific receptors near a steady-state emission source. The model is capable of simulating air dispersion of pollutants from point, area, volume, and line sources. A full description of the ISCST3 model and its algorithms can be found in the ISCST3 User's Guide (U.S. EPA, 1995).

The results of the air dispersion model rely on four basic data sets: 1) meteorological conditions; 2) facility characteristics; 3) location of buildings near the emission sources; and 4) location of receptors (distance to the emission source and elevation on the terrain). To calculate the air dispersion of contaminants, the ISCST3 model requires hourly meteorological data. They include values of 1) wind speed and flow vector; 2) ambient air temperature; 3) atmospheric stability class; and 4) rural and urban mixing height.

In order to calculate the dry and wet deposition fluxes to the ground, additional information is needed: 1) friction velocity; 2) Monin–Obukhov length; 3) surface roughness length; and 4) precipitation code.

The meteorological data used in this exercise contained hourly values of wind speed and wind direction, ambient temperature, precipitation, and solar radiation. All further parameters can be calculated using this information. Figure 8.4 shows the wind rose (distribution of the flow vector) corresponding to the meteorological data used in this case study. It can be observed that wind blowing from the north is the most frequent and that wind blowing from the east is strongest. The studied landfill of MSHW is situated in a zone with a high percentage of calm hours. About 27% of all hourly wind speed values did not exceed 0.1 m/s and the average wind speed was 2.75 m/s.

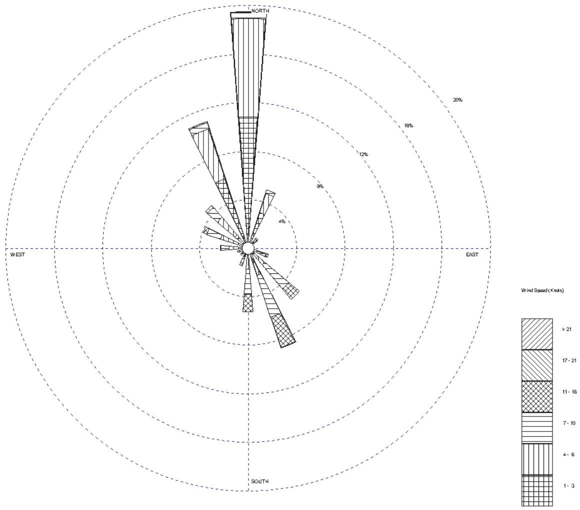

FIGURE 8.4 Wind rose (wind speed and flow vector).

The studied area was defined with an extension of 10 × 10 km, locating the plant in the middle. A set of topographic data for the studied area, including the elevations of the terrain, was used. Figure 8.5 shows the topography of the study area. It can be observed that in the eastern direction, 3 km away from the plant the terrain presents the most considerable elevation in the area. In the southern direction, 1 km away from the plant the terrain presents an elevation that, because it is closer to the plant, can be suspected of affecting the air dispersion of the pollutant emissions.

A network of 2602 Cartesian receptors (10 × 10 grid) was established to model the dispersion of pollutants in the entire studied area. The measuring points were set 200 m apart, representing a total area of 100 km^2. Each receptor was assigned an elevation based on the topographic map of the region.

The characteristics of the emission source have a high importance to the resulting air dispersion concentrations of a pollutant. In addition to the dimensions of the sources — location, surface, height — and the physical characteristics of the emission flow, the concentrations of pollutants in the emission flow are required by the ISCST3 model. The landfill is located in the middle of the topographic map; its dimensions are 173 × 173 × 70 m. The emissions of 1,1,1-trichloroethane, as explained in the application of the LCA to the landfill of MHSW, were 54,533 g per year.

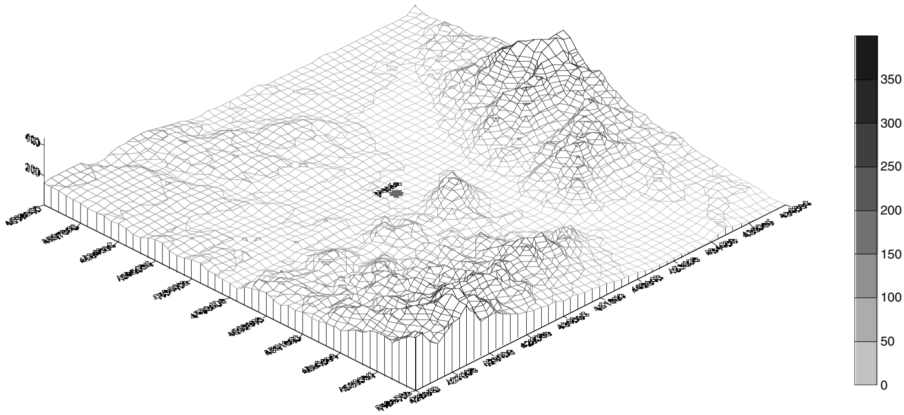

FIGURE 8.5 Topographic map of the study area.

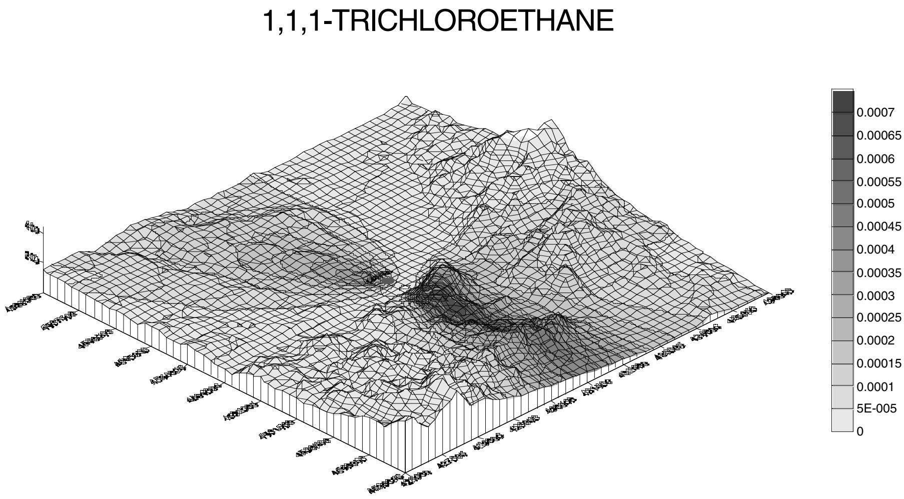

FIGURE 8.6 Air 1,1,1-trichloroethane concentration (μg/m^3)

8.2.4.4 Air Dispersion Results of 1,1,1-Trichloroethane

This section of the study presents the results of the air dispersion modeling for the emission of 1,1,1-trichloroethane. The atmospheric distribution of the 1,1,1,-trichloroethane emissions is illustrated in Figure 8.6, which shows the average concentration of 1,1,1-trichloroethane at ground level in the vicinity of the landfill.

The main wind in the study area is blowing from the north (most frequent; see the wind rose in Figure 8.4). Therefore, it was expected that the highest air concentration of the pollutants would occur south of the landfill. This prediction was confirmed by the modeling results. The annual average concentrations southward were substantially higher than in other parts of the region. However, it can be observed that air dispersion in the south was influenced by topography, more concretely by the elevation present 1 km away in the south. On the other hand, it can be observed that the wind blowing from the southeast influenced the air dispersion to the northwest direction.

In order to assess the air concentrations of 1,1,1-trichloroethane, medium concentrations for the entire study area were estimated. The medium air concentration of the pollutant was 9.69×10^{-2} ng/m^3; it was estimated with the annual averages in all 2584 Cartesian receptors representing the study area.

8.2.4.5 Exposure Calculation Model

For the evaluation of the exposure of the population living in the area, a multiple pathway exposure, transport and transformation model (CalTOX) was used. In Chapter 4 this model is compared with other multimedia model EUSES. This model includes modules for the distribution of substances in the environment, exposure scenario models for humans and the environment, human risk estimation and efforts to quantify and reduce uncertainty in multimedia. It has been designed to assist in assessing human exposure, define soil clean-up goals at (uncontrolled) hazardous waste sites and improve the quality of risk assessment information, especially as required for regulatory decisions. The models and data sets have been compiled as Microsoft Excel 4.0 (or higher) spreadsheets. They are available together with information and documentation via the Internet through various websites.

Each pathway can be included or excluded separately in the calculations, depending on the scope of the study. The exposure model defines air, drinking water, food, and soil as the four main sources for human exposure to a substance via different pathways such as inhalation, ingestion, or dermal contact.

Figure 8.7 shows the eight compartments implemented in CalTOX: air, surface water, ground water, sediment, surface soil, root-zone soil, vadose-zone soil and plants. Usually a chemical equilibrium between the phases is assumed. Unidirectional transport is assumed only from soil to water and from upper to lower soil zones, mainly because of the much higher diffusion speed in these directions compared to the other direction. The equations used in CalTOX to estimate exposure and risk are taken from the U.S. Environmental Protection Agency Risk Assessment Guidance for Superfund (U.S. EPA, 1989) and from the California Department of Toxic Substances Control (DTSC, 1992a,1992b). They are based on conservation of mass.

The release of a substance can be continuous (to air, water and surface soil) or a batch process with an initial concentration within deeper soil zones. The exposure model of CalTOX calculates daily human doses for various pathways (e.g., inhalation of air, ingestion of soil, milk, meat, etc.) based on daily intake rates and predicted concentrations in the respective exposure medium. Finally, risk values based on the doses are calculated.

8.2.4.6 Spatial Scale, Time Scale and Assessable Substances

There is only one spatial scale because CalTOX is intended to be used site specifically (i.e., to assess a specific existing site rather than a big area such as an entire

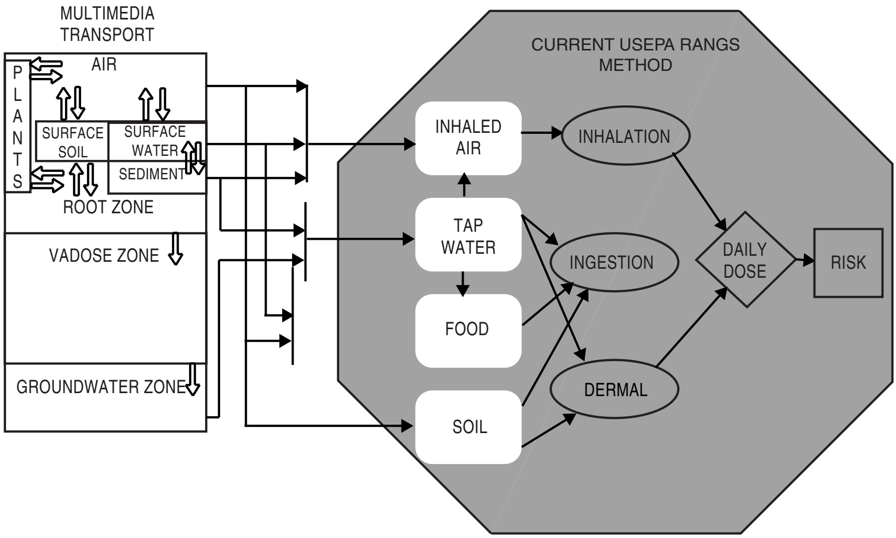

FIGURE 8.7 The structure of the CalTOX model with the multimedia transport, the inter-media transfer and its exposure pathways (DTSC, 2002a).

country). Although it is rather flexible, the assessed area should not be smaller than 1000 m^2 with a maximum water fraction of 10% of the surface area. Theoretically, there is no upper limit for the area, but one should keep in mind that "site specific" means specific for an actual site but not for a country or a continent. However, this limit is set by the adjustments to the input data. The greater the area is, the more average (hence less specific) every value necessarily must be and the less site specific is the entire assessment.

To obtain good results, the time scale should be defined as rather long, preferably from 1 year to decades. If shorter periods are assessed properly, time-averaged landscape properties must be employed. Obviously, CalTOX has not been designed to assess acute exposure at an actual site. The original idea was to have an already contaminated site and to assess the risk for human beings living or working at or near a site that led to subchronic or chronic exposure only.

The substances assessable with CalTOX, listed in descending order of reliability, are: nonionic organic chemicals, radionuclides, fully dissociating inorganic and/organic chemicals, solid-phase metal species, and partially dissociated inorganic and organic species (the latter only if partition coefficients are well adjusted to the pH of the considered landscape). Generally, only very low concentrations that do not exceed the solubility limit in any phase can be applied. Surfactants or volatile metals cannot be assessed.

8.2.4.7 Input Data for CalTOX

Such complex models need a good range of input data in order to obtain trustworthy results:

Data describing the substance:

- Physicochemical properties
- Measured emissions into the compartments
- Background concentrations of the contaminant
- Toxicological properties consisting of cancer and noncancer potency for human beings because only human risk is considered in CalTOX

Data characterizing the area:

- Geographical data such as size of the contaminated area
- Meteorological and hydrological data, e.g., the average depth of surface water, the annual average precipitation, wind speed or environmental temperature
- Soil properties, such as the organic carbon fractions
- Data about the human population, e.g., the average body weight or daily intake rates for different kinds of food

In the present study, the objective of the application of CalTOX is to evaluate the health of the population due to the 1,1,1-trichloroethane emissions from a landfill applied to a determined area. First, the physicochemical properties of 1,1,1-trichloroethane as well as the risk parameters were obtained from the database of CalTOX, and are shown in Table 8.9 and Table 8.10, respectively. Then the geographical

TABLE 8.9
Value for Some Physicochemical Properties of 1,1,1-Trichloroethane

Compound name	1,1,1-Trichloroethane
Molecular weight (g/mol)	133.4
Octanol–water partition coefficient	2.7×10^2
Melting point (K)	242.75
Vapor pressure (Pa)	16,515.975
Solubility (mol/m^3)	9.89505247
Henry's law constant (Pa-m^3/mol)	1651.5975
Diffusion coefficient in pure air (m^2/d)	0.67392
Diffusion coefficient in pure water (m^2/d)	8.6386×10^{-5}
Organic carbon partition coefficient (L/kg)	110

TABLE 8.10
Value for Risk Parameters of 1,1,1-Trichloroethane

Compound name	1,1,1-Trichloroethane
EDF substance ID	71-55-6
Inhalation dose ADI for noncarcinogenic effects	0.28
Ingestion dose ADI for noncarcinogenic effects	0.5
Dermal dose ADI for noncarcinogenic effects	0.5
Total dose ADI for noncarcinogenic effects	0

TABLE 8.11
Geographical, Meterological and Population Data Describing the Study Area

Property	Value
Area [km^2]	100
Inhabitants	580,245
Area fraction of water	0.01
Area fraction of natural soil	0.46
Area fraction of agricultural soil	0.46
Wind speed [m/s]	2.8
Average annual precipitation [mm/a]	455.4
Environmental temperature [°C]	15.7
Egg intake [kg/(kg*d)]	3.85^{-4}
Grain intake [kg/(kg*d)]	3.11^{-3}
Fruit and vegetable intake [kg/(kg*d)]	7.39^{-3}
Daily intake of fish [kg/d]	0.08
Daily intake of leaf crops (including fruits and cereals [kg/d]	0.674
Daily intake of meat [kg/d]	0.04
Daily intake of dairy products [kg/d]	0.069
Milk intake [kg/(kg*d)]	2.78^{-3}
Body weight [kg]	67.52

parameters and population data from the nearest residential area within 10 km around the plant were defined (Table 8.11).

It must be taken into account that 1,1,1-trichloroethane does not occur naturally in the environment. It is found in many common products such as glue, paint, industrial degreasers, and aerosol sprays. Since 1996, 1,1,1-trichloroethane has not been produced in the U.S. because of its effects on the ozone layer.

An expected most important exposure pathway might be through inhaled air because the landfill is the major source. No information is available to show that 1,1,1-trichloroethane causes cancer. The International Agency for Research on Cancer (IARC) has determined that 1,1,1-trichloroethane is not classifiable as to its human carcinogenicity.

8.2.4.8 Exposure Results of 1,1,1-Trichloroethane

Figure 8.8 represents the daily human doses through eight types of exposure that were calculated based on the simulated air concentration due to the 1,1,1-trichloroethane emissions of the landfill of MSHW. As expected, the main exposure pathway is air inhalation. Depending on the exposure level, adverse health effects other than cancer can be associated with all chemical substances. The hazard ratio result of the actual human exposure due to the emission of the 1,1,1-tricholoroethane is 1.7×10^{-10}; thus human exposure does not exceed the defined exposed level.

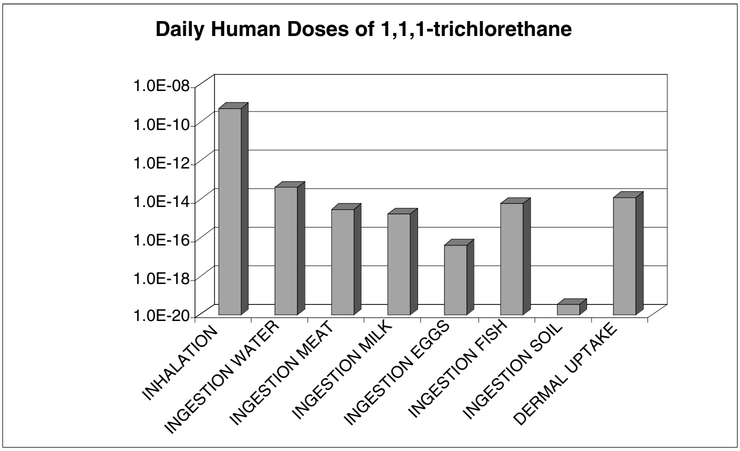

FIGURE 8.8 Daily human dose through different exposure types (mg/kg/d).

8.2.5 Conclusions from the Landfilling Example of MHSW

A comprehensive LCA in the context of integrated solid waste management software was carried out. It is evident that in the case of landfilling, the site-specific component is relevant to evaluate if potential impacts really correspond to actual impacts. To further study this question in the example the main pollutant contributing to the human health indicator was chosen to carry out an exposure risk assessment.

It could be shown that the potential impact does not correspond to any unacceptable risk for the neighborhood. Nevertheless, the value of LCA clearly lies in its highlighting of the existing emissions that must all be considered according to the precautionary principle and in its holistic approach by assessing several pollutants and the related impact categories at once along the studied life cycle. Further work is necessary and should include performing similar studies for other media such as water and soil with adequate models and taking into account the accident risks.

8.3 Example 2: LCA of Universal Remote for Operating a Television and Video

8.3.1 Introduction

A case study of an LCA of a universal remote for operating a television and a video is performed. The inventory data include the consumption of raw materials and energy through manufacturing, distribution and use of the remote, and the corresponding emissions to air, water and soil in each of these different stages. The following 11 environmental impact categories have been considered in the impact assessment phase: raw material depletion (RMD); global warming (GW); ozone depletion (OD); air toxicity (AT); photochemical ozone creation (POC);

air acidification (AA); water toxicity (WT); water eutrophication (WE); energy depletion (ED); water depletion (WD); and hazardous waste production (HWP). The software model and database EIME®[1] (environmental information and management explorer mentioned in Chapter 2) of Schneider Electric Industries, Ltd., has been used in the inventory analysis and impact assessment phases (Figure 8.9).

8.3.2 Goal and Scope Definition

In this case study, the LCA methodology was used to identify and quantify aspects with a major environmental impact through the entire life-cycle of a universal remote for operating a television and a video (one for all). The final purpose was to identify the most relevant design aspects that could be improved from an environmental point of view. One unit of the universal remote (97 g) has been considered as the functional unit of the LCA.

The following main steps have been taken into account in the environmental assessment: production of materials and components required for the remote, assembly or end product manufacturing process, distribution to retailers, and consumption of alkaline batteries during use. The remote's end-of-life stage was not considered due to the lack of accurate information about this stage; thus any environmental load associated with this final stage was not considered in this LCA. (The inventory of disposal for electronic products is difficult to carry out due to product complexity and the need of simulating different release scenarios of environmentally critical substances into the environment from landfills, incinerators and metal refineries in case of recycling). Figure 8.10 shows the flow diagram of the system studied with the corresponding main inputs and outputs considered.

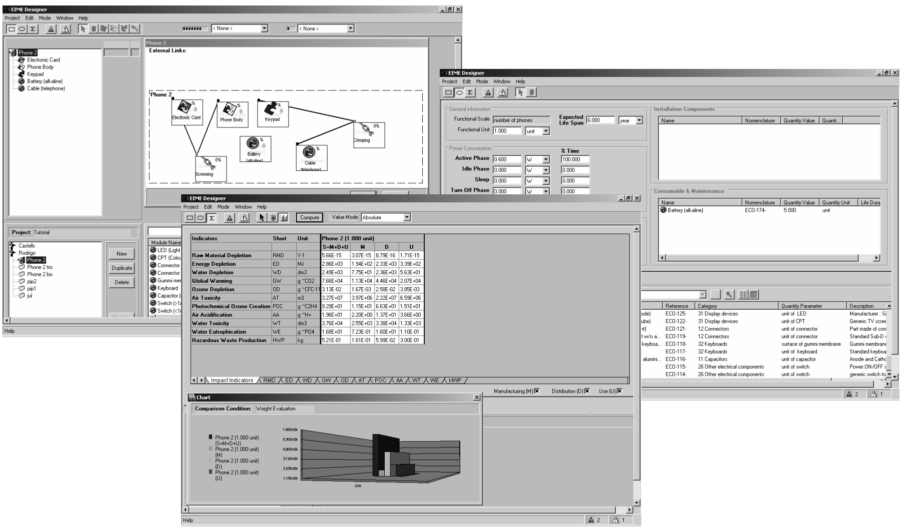

FIGURE 8.9 The software model and database EIME.

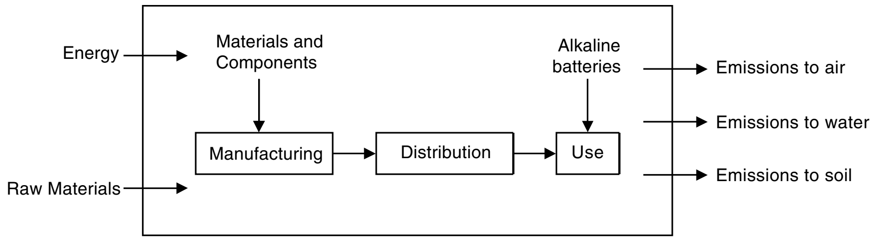

FIGURE 8.10 Flow diagram of the universal remote.

8.3.3 Inventory Analysis

To carry out the inventory analysis, real and bibliographic European data have been considered. For the collection data, the three main steps identified in the preceding section have been taken into account.

8.3.3.1 Input Data for the Software EIME

The following section summarize the main input data introduced in the software for characterizing, from an environmental point of view, all the elements directly and indirectly implicated during manufacturing, distribution and use of the universal remote.

Manufacturing:
Universal remote total weight (1 unit): 97 g (Figure 8.11 and Table. 8.12)
Processes required for manufacturing 1 unit of the universal remote:
Printed circuit board (PCB) etching process: 0.2 MJ elec/PCB and dissolution of 1.5 g of Cu/PCB
Wave soldering process: 0.7 MJ elec/PCB, 1.1 g of Pb and 1.7 g of Sn
Reflow process: 1.4 MJ elec/PCB, 0.0015 g of Pb/pat and 0.003 g of Sn/pat

Distribution:
750 km in a heavy-duty truck (empty return)
250 km in a light-duty truck (empty return)

Use:
Battery consumption: 2 alkaline batteries of 1.5 V/1.5 years
Battery weight: 24 g/unit
Universal remote lifetime (expected): 10 years

FIGURE 8.11 One unit of the universal remote (97 g).

TABLE 8.12
Components and Materials

Components and materials	Units	Total weight (g)
(c) Capacitor (ceramic)	1	0.020
(c) Capacitor (electrolytic)	1	0.660
(c) Diode (plastic body)	1	0.008
(c) Integrated circuit (Si)	1	0.500
(c) LED (display device)	1	0.250
(c) Radio frequency transistor	1	0.303
(c) Radio frequency transmitter	1	0.250
(c) Resistances	7	0.063
(m) Buttons (silicone rubber)	1	19
(m) Laminate for PCB (FR4)	1	15.746
(m) Plastic box (ABS)	1	60
(m) Spirals (steel)	2	0.200

(c): component; (m): material.

8.3.3.2 Inventory of the Universal Remote

After introducing the previously presented input data in the software model (EIME), the ecobalance or inventory of the universal remote (1 unit) has been automatically calculated. The complete inventory was integrated by 187 environmental loads (inputs and outputs): energy and raw materials consumed, hazardous wastes produced, and emissions to air, water and soil. The main inputs and outputs of the system that contribute more than 5% to any of the environmental impact indicators subsequently considered are shown in Table 8.13.

8.3.4 Impact Assessment

To carry out this phase of the LCA case study, the following 11 impact categories have been considered: RMD, GW, OD, AT, POC, AA, WT, WE, ED, WD and HWP. The specific environmental impact indicators used in each of the environmental impact categories mentioned earlier are shown in Table 8.14.

The environmental loads (inputs and outputs), previously inventoried, have been classified under their corresponding environmental impact indicators by following the classification criteria specified in these indicators. Then, to characterize the environmental loads or, in other words, to quantify the potential contribution of each environmental load in the different impact indicators, characterization factors have been used. These factors are pre-established in each impact indicator. Finally, the corresponding potential contributions have been determined by multiplying the mass of the environmental loads per these characterization factors (for example, 29.5 or 3.0×10^1 g of methane $\times$ 56 g equiv. of carbon dioxide/g methane = 1652 or 1.7×10^3 g equiv. of carbon dioxide).

The inventoried environmental loads are classified under their corresponding impact indicators with their respective characterization factors in Table 8.15. The total potential contribution of each environmental load in each impact indicator is presented, as well as the corresponding contribution percentage for the three stages considered (manufacturing, distribution and use of the universal remote).

8.3.5 Interpretation

As can be seen in Figure 8.12 and Table 8.16, the remote's manufacturing stage contributes more than 60% to each of the environmental impact indicators considered, this stage being the environmentally most relevant within the remote life-cycle. During the use stage, consumption of alkaline batteries (two batteries per 1.5 years during the total remote lifetime of 10 years) contributes between 2 and 36% to each of the indicators. During the distribution stage, transport of the remote from the manufacturing site to retailers in a heavy- and a light-duty truck (750 and 250 km, respectively) contributes less than 1.5% to each of the indicators, this stage being the environmentally less relevant within the remote life-cycle.

Due to the environmental relevance of the manufacturing stage in the total remote life-cycle, in the following paragraphs the manufacturing stage of the universal remote will be studied and discussed in more detail. Table 8.17 shows the environ-

TABLE 8.13
Inventory of the Universal Remote

Flow	Units	TOTAL	Manufacturing	Distribution	Use
Inputs					
(r) Gold (Au, ore)	kg	7.0×10^{-21}	7.0×10^{-21}	0	0
(r) Silver (Ag, ore)	kg	9.4×10^{-21}	9.4×10^{-21}	1.4×10^{-27}	8.2×10^{-26}
(r) Tin (Sn, ore)	kg	1.9×10^{-18}	1.6×10^{-18}	0	2.7×10^{-19}
(r) Zinc (Zn, ore)	kg	5.6×10^{-17}	5.6×10^{-24}	2.1×10^{-27}	5.6×10^{-17}
(r) Water used (total)	L	1.1×10^{2}	6.4×10^{1}	3.1×10^{-2}	4.2×10^{1}
(e) E Total primary energy	MJ	$1.5 \times 10_{2}$	1.3×10^{2}	3.4×10^{-1}	2.0×10^{1}
Outputs					
(a) Cadmium (Cd)	g	3.3×10^{-3}	5.4×10^{-4}	6.4×10^{-7}	2.8×10^{-3}
(a) Carbon dioxide (CO_2, fossil)	g	7.6×10^{3}	6.2×10^{3}	2.4×10^{1}	1.3×10^{3}
(a) CFC 11 ($CFCl_3$)	g	2×10^{-4}	1.2×10^{-4}	0	1.8×10^{-8}
(a) Ethylene (C_2H_4)	g	6.7×10^{-1}	6.5×10^{-1}	1.3×10^{-4}	1.8×10^{-2}
(a) Halon 1301 (CF_3Br)	g	1.1×10^{-4}	7.7×10^{-5}	1.5×10^{-6}	2.9×10^{-5}
(a) Hydrocarbons (except methane)	g	9.0	7.3	8.1×10^{-2}	1.6
(a) Hydrocarbons (unspecified)	g	9.5×10^{-1}	5.6×10^{-1}	5.1×10^{-6}	4.0×10^{-1}
(a) Methane (CH_4)	g	3.0×10^{1}	2.7×10^{1}	2.5×10^{-1}	2.9
(a) Nitrogen oxides (NO_x as NO_2)	g	1.8×10^{1}	1.5×10^{1}	2.9×10^{-1}	2.8
(a) Sulfur oxides (SO_x as SO_2)	g	4.5×10^{1}	3.5×10^{1}	1.1×10^{-2}	1.1×10^{1}
(w) Ammonia (NH_4^+, NH_3, as N)	g	3.7×10^{-1}	3.6×10^{-1}	6.4×10^{-4}	9.5×10^{-3}
(w) Copper (Cu^+, Cu^{++})	g	2.3×10^{-2}	2.3×10^{-2}	1.1×10^{-7}	2.1×10^{-4}
(w) Chlorinated matter (as Cl)	g	2.3×10^{-2}	2.3×10^{-2}	8.5×10^{-6}	7.1×10^{-4}
(w) Iron (Fe^{++}, Fe_3^+)	g	9.9×10^{-2}	1.6×10^{-2}	1.4×10^{-5}	8.3×10^{-2}
(w) Nitrates (NO_3^-)	g	7.8×10^{-1}	7.7×10^{-1}	1.1×10^{-3}	1.1×10^{-2}
(w) Phenol (C_6H_5OH)	g	1.2×10^{-4}	1.0×10^{-4}	1.7×10^{-6}	1.1×10^{-5}
(w) Phosphates (as P)	g	2.7×10^{-2}	1.5×10^{-3}	2.3×10^{-8}	2.5×10^{-2}
(w) Phosphorus (P)	g	2.3×10^{-1}	2.3×10^{-1}	3.1×10^{-6}	6.8×10^{-6}
(w) Suspended matter (unsp.)	g	3.9	3.6	2.8×10^{-4}	4.0×10^{-1}
(hw) Waste (hazardous)	kg	3.4×10^{-2}	3.3×10^{-2}	7.4×10^{-6}	2.5×10^{-4}

TABLE 8.14
Environmental Impact Categories and Indicators Used in EIME

Acronym	Category	Indicator — source (year)
RMD	Raw material depletion	Ec (R*Y) — Ecobalance USA (1994)
GW	Global warming	IPCC (20 years) — International Panel on Climate Change (1995)
OD	Ozone depletion	WMO (high) — World Meteorological Organization (1991)
AT	Air toxicity	CVCH (air) — Swiss Federal Office of Environment, Forests and Landscape (1991)
POC	Photochemical ozone creation	POCP (high) — United Nations Economic Commission for Europe (1991)
AA	Air acidification	CML (AA) — CML Leiden University (1992)
WT	Water toxicity	Spanish legislation — RD927/98 (1998)
WE	Water eutrophication	CML (water) — CML Leiden University (1992)
ED	Energy depletion	Total consumption of energy
WD	Water depletion	Total consumption of water
HWP	Hazardous waste production	Total production of hazardous wastes

mental contribution (in %) of each material, component and process implicated in the remote manufacturing process to each of the 11 environmental impact indicators considered.

As can be seen in Table 8.17 and according to the database used, the greatest environmental impact of the universal remote is associated with the integrated circuit (with silicon content), due mainly to the energy needed for its manufacture. This component contributes very significantly (between 63 and 93%) to 9 of the 11 environmental impact indicators considered: GW, OD, AT, POC, AA, WT, ED, WD and HWP.

The wave soldering process has the greatest contribution to the RMD indicator due to its high consumption of lead and tin (solder alloy). Other components and materials made with scarce natural nonrenewable resources also have a significant contribution to this impact indicator (for example, integrated circuit with silicon content, laminate for PCB with copper content, etc.). During the wave soldering process, the solder flux with volatile organic compound (VOC) content contributes significantly to the POC indicator.

The etching process of the PCB has the greatest contribution to the WE indicator because, according to the database used, a nitrogenous compound (dicyanodiamide) is used in the PCB manufacturing process and is consequently partially released to residual water during the etching process. The etching process of the PCB also has a significant contribution to the WT indicator due to use of the etching agent with copper dissolved into it and its release to residual water.

TABLE 8.15
Impact Assessment of the Universal Remote[a]

Impacts	Characterization factors	TOTAL	Manufacturing (%)	Distribution (%)	Use (%)
Raw material depletion (yr^{-1})	*	1.29×10^{-14}	63.9	0.0	36.1
(r) Gold (Au, ore)	591,000	4.1×10^{-15}	100.0	0.0	0.0
(r) Zinc (Zn, ore)	64.9	3.6×10^{-15}	0.0	0.0	100.0
(r) Tin (Sn, ore)	1,800	3.4×10^{-15}	85.7	0.0	14.3
(r) Silver (Ag, ore)	79,400	7.4×10^{-16}	100.0	0.0	0.0
Global warming (g equiv. CO_2)	*	9.3×10^{3}	83.7	0.4	15.9
(a) Carbon dioxide (CO_2, fossil)	1	7.6×10^{3}	82.6	0.3	17.1
(a) Methane (CH_4)	56	1.7×10^{3}	89.5	0.8	9.7
Ozone depletion (g eq. CFC-11)	*	2.0×10^{3}	73.7	1.3	25.0
(a) Halon 1301 (CF_3Br)	17.2	1.8×10^{-3}	71.9	1.4	26.7
(a) CFC 11 ($CFCl_3$)	1	1.2×10^{-4}	100.0	0.0	0.0
Air toxicity (m^3)	*	3.6×10^{6}	71.7	0.4	27.9
(a) Sulfur oxides (SO_x as SO_2)	33,300	1.5×10^{6}	77.4	0.0	22.6
(a) Carbon dioxide (CO_2, fossil)	125	9.4×10^{5}	82.6	0.3	17.1
(a) Nitrogen oxides (NO_x as NO_2)	33,300	5.9×10^{5}	82.7	1.6	15.7
(a) Cadmium (Cd)	100,000,000	3.3×10^{5}	16.1	0.0	83.9
Photo. ozone creation (g equiv. ethylene)	*	1.1×10^{1}	82.3	0.7	17.0
(a) Hydrocarbons (except methane)	0.808	7.3	81.1	0.9	18.0
(a) Methane (CH_4)	0.03	8.9×10^{-1}	89.5	0.8	9.7
(a) Hydrocarbons (unspecified)	0.799	7.6×10^{-1}	58.2	0.0	41.8
(a) Ethylene (C_2H_4)	1	6.7×10^{-1}	97.3	0.0	2.7
Air acidification (g equiv. H^+)	*	1.9	78.6	0.4	21.0
(a) Sulfur oxides (SO_x as SO_2)	0.0313	1.4	77.4	0.0	22.6
(a) Nitrogen oxides (NO_x as NO_2)	0.0217	3.9×10^{-1}	82.7	1.6	15.7
Water toxicity (liters)	*	1.9×10^{3}	81.0	0.2	18.8

-- *continued*

TABLE 8.15 (CONTINUED)
Impact Assessment of the Universal Remote[a]

Impacts	Characterization factors	TOTAL	Manufacturing (%)	Distribution (%)	Use (%)
(w) Copper (Cu^+, Cu^{++})	20,000	4.6×10^2	99.1	0.0	0.9
(w) Ammonia (NH_4^+, NH_3, as N)	1,000	3.7×10^2	97.2	0.2	2.6
(w) Iron (Fe^{++}, Fe^{3+})	3,330	3.3×10^2	16.4	0.0	83.6
(w) Chlorinated matter (unspecified, as Cl)	10,000	2.3×10^2	96.9	0.0	3.0
(w) Suspended matter (unspecified)	40	1.6×10^2	89.9	0.0	10.0
(w) Phenol (C_6H_5OH)	1,000,000	1.2×10^2	88.7	1.5	9.8
Water eutrophication (g equiv. PO_4)	*	1.1	91.2	0.0	8.7
(w) Phosphorus (P)	3.06	7.0×10^{-1}	100.0	0.0	0.0
(w) Ammonia (NH_4^+, NH_3, as N)	0.42	1.5×10^{-1}	97.2	0.2	2.6
(w) Phosphates (as P)	3.06	8.2×10^{-2}	5.5	0.0	94.5
(w) Nitrates (NO_3^-)	0.095	7.4×10^2	98.5	0.1	1.3
Energy depletion (MJ)	*	1.5×10^2	86.5	0.2	13.2
(e) E total primary energy	1	1.5×10^2	86.5	0.2	13.2
Water depletion (m^3)	*	1.1×10^2	60.7	0.0	39.2
(r) Water used (total)	1	1.1×10^2	60.7	0.0	39.2
Hazardous waste production (kg)	*	1.4×10^{-1}	97.6	0.0	2.4
(hw) Waste: slags and ash (unspecified)	1	1.0×10^{-1}	97.1	0.0	2.9
(hw) Waste (hazardous)	1	3.4×10^{-2}	99.2	0.0	0.8

[a] 1 unit of 97 g.
(r): natural resources; (e): energy; (a): air emission; (w): water emission; (hw): hazardous waste.

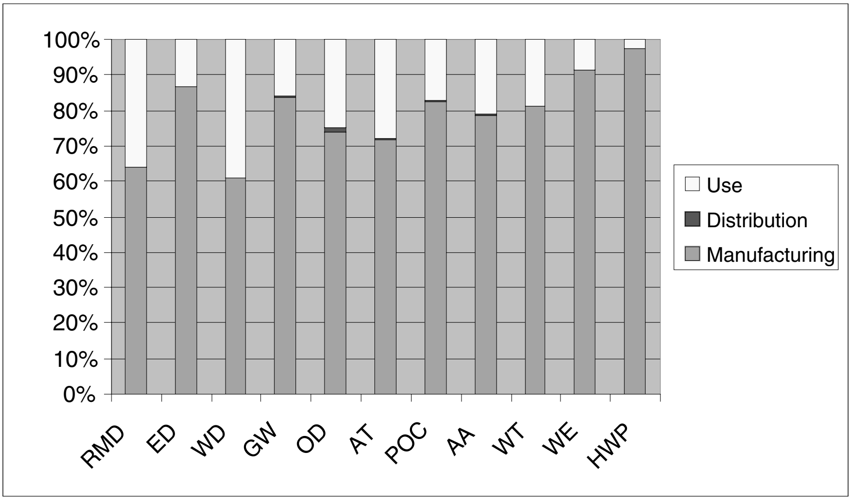

FIGURE 8.12 Stages' contribution (%) in each impact indicator.

TABLE 8.16
Stages' Contributions (%) in Each Impact Indicator

Impacts	Manufacturing	Distribution	Use
RMD	**63.9**	0.0	36.1
GW	**83.7**	0.4	15.9
OD	**73.7**	1.3	25.0
AT	**71.7**	0.4	27.9
POC	**82.3**	0.7	17.0
AA	**78.6**	0.4	21.0
WT	**81.0**	0.2	18.8
WE	**91.2**	0.0	8.7
ED	**86.5**	0.2	13.2
WD	**60.7**	0.0	39.2
HWP	**97.6**	0.0	2.4

8.3.6 Limitations of Site-Specific Impact Assessment in the Remote Life Cycle

The LCA carried out for the universal remote is an excellent illustration of the limitations of site-specific impact assessment. The identified impacts in Table 8.15 are mainly related to the manufacturing, but this does not mean that the impact occurs where the manufacturing is carried out. Conversely, in the case of this universal remote LCA the manufacturing stage only sums up the total life-cycle

TABLE 8.17
Materials′, Components′ and Processes′ Contributions (%) to Each Impact Indicator

Materials, components and processes	RMD	GW	OD	AT	POC	AA	WT	WE	ED	WD	HWP
(c) Capacitor (ceramic)	0	0	0	0	0	0	0	0	0	0	0
(c) Capacitor (electrolytic)	1	0	0	0	0	1	0	0	0	0	0
(c) Diode (plastic body)	0	0	0	0	0	0	0	0	0	0	0
(c) Integrated circuit (Si)	7	**89**	**84**	**87**	**63**	**87**	**59**	**26**	**88**	**91**	**93**
(c) LED (display device)	2	0	0	0	0	1	0	0	0	2	0
(c) Radio frequency transistor	2	0	0	0	0	0	0	0	0	0	0
(c) Radio frequency transmitter	2	0	0	0	0	1	0	0	0	2	0
(c) Resistances	0	0	0	0	0	0	0	0	0	0	0
(m) Buttons (silicone rubber)	0	1	9	1	0	1	1	0	1	0	0
(m) Laminate for PCB (FR4)	3	2	1	3	2	3	3	1	2	1	1
(m) Plastic box (ABS)	0	3	0	3	3	2	3	1	4	1	1
(m) Spirals (steel)	0	0	0	0	0	0	0	0	0	0	0
(p) PCB etching	0	0	0	0	0	0	31	**72**	0	1	1
(p) PCB reflow	3	3	3	3	2	3	1	0	3	1	3
(p) PCB wave soldering	**80**	2	2	2	29	2	0	0	2	1	1

(c): component; (m): material; (p): process.

impacts of its components assembled in the manufacturing process. The components with their own life cycles represent a huge amount of industrial processes that probably occur in many places around the world.

This means that this example falls under the category of “full LCAs of complex products with a huge number of industrial processes” (see Figure 6.1 and descriptive text in Section 6.1 for further explanation). An integrated approach of life-cycle and site-oriented risk assessment as promoted in this book seems to be highly difficult for a universal remote, at least when taking into account the capacity of the impact assessment models currently available.

8.4 EXAMPLE 3: ENVIRONMENTAL IMPACT ANALYSIS OF AN INDUSTRIAL SEPARATION PROCESS, APPLYING LCA AND ERA

8.4.1 INTRODUCTION

An environmental impact analysis of the isopentane separation process from a naphtha stream was developed. The system consists of a distillation column, which requires electricity and steam. The evaluation covered the distillation process and units of utility production. In this academic case study, the environmental assessment was carried out in two ways: potential impact and site-specific impact assessments. By this, the most important aspects about LCA, impact pathway analysis (IPA) and environmental risk assessment (ERA) approaches were analyzed.

This example is in the heart of chemical engineering. The LCA study carried out is a cradle-to-gate study. This means that one industrial process, situated in the petrochemical complex of the Tarragona region (Spain), with its associated environmental loads due to raw material and energy consumption is considered. Two related sub-processes, electricity generation and steam production, both by on-site small fossil-based thermal plants, are simulated more in detail. The data for the other materials are taken from the LCA software database. Based on the LCA results one process is to be further assessed by site-specific impact assessment. In this example we will try to carry out a more generic IPA first and then a more detailed ERA to deal with open questions not solvable with the IPA software Ecosense.

8.4.2 Goal and Scope Definition

In this case study, the environmental impact analysis is applied to the separation process from a naphtha stream in order to evaluate the environmental effects of all process stages, including its requirements of electric power and steam. The system separates isopentane from naphtha about 16 t/h production. In this sense, 16 t/h production has been considered the functional unit for the LCA. The separation plant consisting of a distillation column, the electricity generation and the steam production were the three main steps taken into account. Figure 8.13 shows the flow diagram of the studied system with the inputs and outputs considered.

8.4.3 Potential Impact Assessment

In order to carry out the inventory analysis, the aforementioned three main stages were considered: steam production, electricity generation and separation of isopentane. The required information in the collection data step was obtained from the real process. Nevertheless, in the case of steam generation, some values were taken from the TEAM database. TEAM is the software used (see Chapter 2) to carry out the LCA for the process and to evaluate the total amount of a specific pollutant in any stage. This software calculates the environmental loads produced by a functional unit starting from given process data.

TEAM consists of an integrated group of software tools to model and analyze any system. A modeling tool is used to describe physical operations. It allows a large database to be built and the LCI to be calculated for any system (inventories conducted for a product life-cycle or other consistent systems). The analysis tools for applying LCIA methods (i.e., potential impacts) based on the inventories results allow further evaluation of the system under study. The scheme created in TEAM to obtain the inventory and potential impacts is represented in Figure 8.14.

8.4.3.1 Inventory

The main data considered and introduced in the TEAM software for creating the inventory are summarized in Table 8.18. The process has two different inputs for

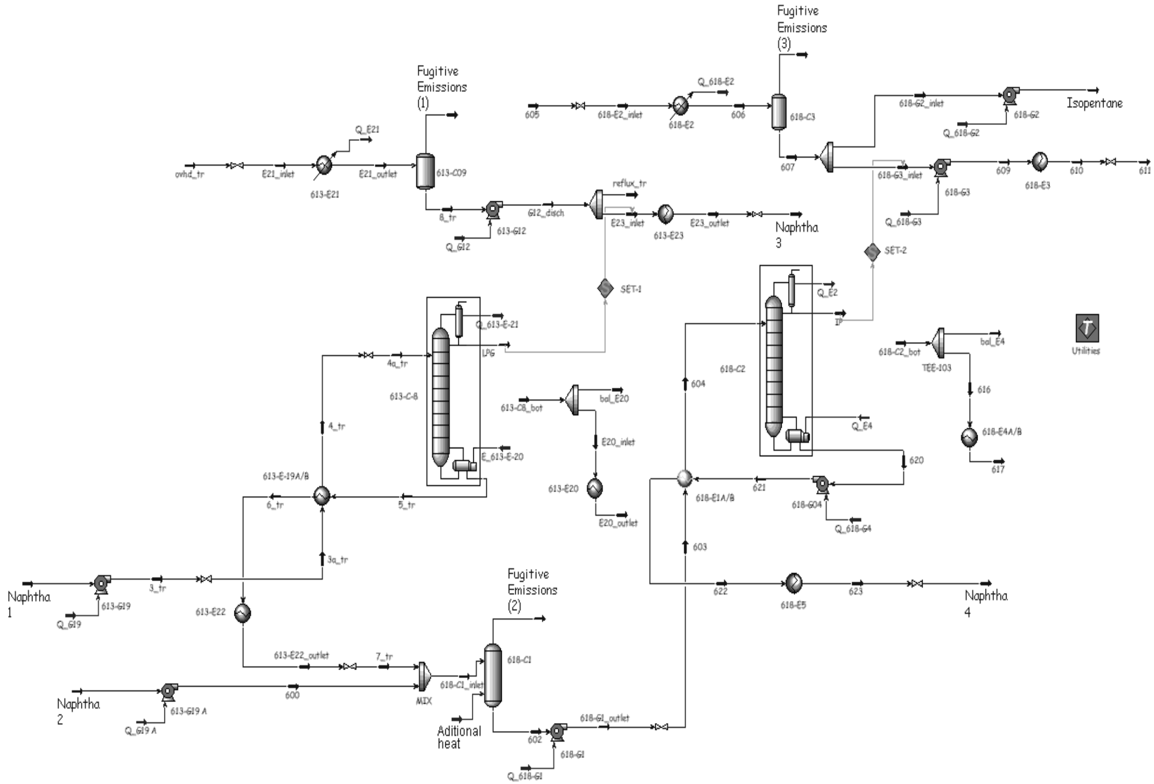

FIGURE 8.13 Flow diagram of the separation process.

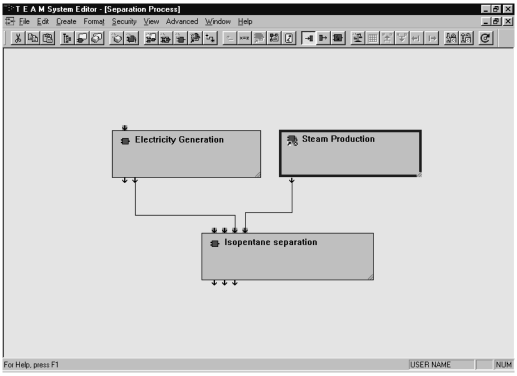

FIGURE 8.14 Diagram of the separation process obtained from TEAM.

naphtha (corresponding to #1 and #2). Likewise, naphtha #3 and #4 represent outputs from the system. Overall energy requirement is 7.2 MJ/h, but the inventory was obtained taking into account that 30% of the total energy is used to produce steam.

Table 8.19 shows the inventory corresponding to the separation process. The comments in parentheses indicate the source, compounds under consideration in a specific group, properties or specific names for the acronyms. As can be seen, volatile organic carbons (VOCs; as fugitive emissions) are the contribution for releases that can be attributed to the separation process, corresponding to 79% of the total amount of generated polluting agents (on a mass basis).

8.4.3.2 Impact Assessment

To carry out this phase of the LCA case study, the following five impact categories have been considered: AA, WE, HT, GE and TE. The specific environmental impact indicator used in each of the environmental impact categories mentioned before is shown in Table 8.20.

The previously inventoried environmental loads are classified in different impact categories. Specific characterization factors are related to each indicator to evaluate the potential contribution of each environmental load. These factors depend on the method used; multiplying them by the environmental loads is possible to obtain the corresponding potential contributions.

Using the factors provided from TEAM, the results obtained can be seen in Table 8.21.

8.4.3.3 Interpretation of the LCA

Looking at the inventory and the impact assessment results (Table 8.19 and Table 8.21) we see that the important fugitive emissions (VOC) have not resulted in any potential impact in the LCIA phase. This is due to the fact that photochemical oxidant formation was not chosen as an impact category for reconsideration. Evidently, by this, the study is also an example on how important information can be lost from the LCI to the LCIA phase by subjective choices of the LCIA indicators.

TABLE 8.18
Main Data (Inputs and Outputs) of the Process

Flow	Inputs	Account (kg/h)	Account (kg/kg Isopentane)	Account total (kg/year)
Raw material	Naphtha (# 1)	2.83×10^4	1.76	2.24×10^8
	Naphtha (# 2)	7.15×10^4	4.45	5.66×10^8
	Coal	1.14×10^2	7.11×10^{-3}	9.06×10^5
	Air	2.01×10^3	1.25×10^{-1}	1.59×10^7
	Water	1.95×10^3	1.21×10^{-1}	1.54×10^7
	Outputs			
Product and by-products	Isopentane	1.61×10^4	1.00	1.27×10^8
	Naphtha (# 3)	5.10×10^2	3.17×10^{-2}	4.04×10^8
	Naphtha (# 4)	8.24×10^4	5.12	6.52×10^8
Emissions	CO_2	2.15×10^2	1.34×10^{-2}	1.70×10^6
	SO_2	1.00	6.23×10^{-5}	7.93×10^3
	NOx	4.37×10^{-1}	2.72×10^{-5}	3.46×10^3
	Particulate matter	1.67	1.04×10^{-4}	1.32×10^4
	As	1.82×10^{-1}	1.13×10^{-5}	1.44×10^3
	Hg	3.39×10^{-4}	2.11×10^{-8}	2.69
	Ni	1.06×10^{-3}	6.60×10^{-8}	8.40
	Pb	1.89×10^{-1}	1.17×10^{-5}	1.50×10^3
	Pb	1.42×10^{-1}	8.84×10^{-6}	1.13×10^3
	VOC (as fugitive emissions)	8.37×10^2	5.21×10^{-2}	6.63×10^6
Solid waste	Inorganic	6.36	3.96×10^{-4}	5.04×10^4
Water waste	COD	3.82×10^{-5}	2.37×10^{-9}	3.02×10^{-1}

8.4.4 Site-Specific Environmental Impact Analysis

The current LCIA results presented in Table 8.21 are visibly predominated by the electricity generation sub-process. This becomes clear when looking back to the inventory in Table 8.19. Therefore, electricity generation is chosen for further site-specific environmental impact analysis. First, an impact pathway analysis is carried out with the integrated assessment model Ecosense for the priority air macro-pollutants. Next an exposure risk assessment is performed for a relevant air emission of a micropollutant that is not directly covered by the integrated assessment model used, but that could signify a risk for the neighborhood of the industrial separation process. For this particular risk assessment study, mercury — not PCDD/Fs — is

TABLE 8.19
Inventory of Process for Separating Isopentane from Naphtha

Flow	Units	TOTAL	Steam production	Isopentane separation	Electricity generation
Inputs					
(r) Coal (in ground)	kg	113.64	0	0	113.64
(r) Natural gas (in ground)	kg	2.64×10^{-2}	2.64×10^{-2}	0	0
(r) Oil (in ground)	kg	2.12×10^{-2}	2.12×10^{-2}	0	0
(r) Air	kg	215.09	0	0	215.09
Naphtha (# 1)	kg	72,543.60	0	72,543.6	0
Naphtha (# 2)	kg	28,713.10	0	28,713.1	0
Water used (total)	liter	1,968.56	2.48×10^{-1}	0	1,968.32
Water: unspecified origin	liter	2.48×10^{-1}	2.48×10^{-1}	0	0
Outputs					
(a) Carbon dioxide (CO_2, fossil)	g	218,278	140.29	0	218,138
(a) Carbon monoxide (CO)	g	4.58×10^{-2}	4.58×10^{-2}	0	0
(a) Hydrocarbons (unspecified)	g	1.93	1.93	0	0
(a) Lead (Pb)	g	144.07	0	0	144.07
(a) Mercury (Hg)	g	3.44×10^{-1}	0	0	0.34
(a) Metals (unspecified)	g	9.17×10^{-1}	9.17×10^{-4}	0	0
(a) Nitrogen oxides (NO_x as NO_2)	g	444.75	1.38	0	443.38
(a) Particulates (unspecified)	g	1.81	1.19×10^{-1}	0	1.69
(a) Sulfur oxides (SO_x as SO_2)	g	1,015.88	1.28	0	1,014.6
(a) Fugitive emissions (VOC)	g	837,000	0	837,000	0
(w) BOD5 (biochemical oxygen demand)	g	9.17×10^{-4}	9.17×10^{-4}	0	0
(w) COD (chemical oxygen demand)	g	9.17×10^{-4}	9.17×10^{-4}	0	0

-- *continued*

TABLE 8.19 (CONTINUED)
Inventory of Process for Separating Isopentane from Naphtha

Flow	Units	TOTAL	Steam production	Isopentane separation	Electricity generation
		Outputs			
(w) Hydrocarbons (unspecified)	g	9.17×10^{-3}	9.17×10^{-3}	0	0
(w) Suspended matter (unspecified)	g	9.17×10^{-4}	9.17×10^{-4}	0	0
Isopentane	kg	16,335.00	0	16,335	0
Naphtha (# 3)	kg	517.44	0	517.44	0
Naphtha (# 4)	kg	83,602.70	0	83,602.7	0
Waste (municipal and industrial)	kg	9.17×10^{-4}	9.17×10^{-4}	0	0
Waste (total)	kg	9.49×10^{-4}	9.49×10^{-4}	0	0
Waste: mineral (inert)	kg	2.75×10^{-5}	2.75×10^{-5}	0	0
Waste: slags and ash (unspecified)	kg	4.58×10^{-6}	4.58×10^{-6}	0	0

(r): natural resources; (a): air emission; (w): water emission.

TABLE 8.20
Environmental Impact Categories and Indicators Used in TEAM

Acronym	Category	Indicator
AA	Air acidification	CML
WE	Water eutrophication	CML
HT	Human toxicity	CML
GE	Greenhouse effect	IPPC (direct, 100 years)
TE	Terrestrial ecotoxicity	USES 2.0

chosen for further analysis (see studies for PCDD/Fs in Chapters 4 and 5). Actions should be initiated as soon as possible to reduce human-generated releases due to the documented, significant adverse impacts on human health throughout the world (UNEP, 2003).

TABLE 8.21
Isopentane Separation Process Impact Assessment Obtained with TEAM™ Software

Impacts	Characterization factors	TOTAL
CML air acidification (g equiv. H^+)		4.14×10^1
(a) Nitrogen oxides (NO_x as NO_2)	2.17×10^{-2}	9.67
(a) Sulfur oxides (SO_x as SO_2)	3.12×10^{-2}	3.17×10^1
CML eutrophication (g equiv. PO_4)		5.78×10^1
(a) Nitrogen oxides (NO_x as NO_2)	1.30×10^{-1}	5.78×10^1
(w) COD (chemical oxygen demand)	2.20×10^{-2}	2.02×10^{-5}
CML human toxicity (g)		2.47×10^4
(a) Carbon monoxide (CO)	1.20×10^{-2}	5.50×10^{-4}
(a) Lead (Pb)	1.60×10^2	2.31×10^4
(a) Mercury (Hg)	1.20×10^2	4.13×10^1
(a) Nitrogen oxides (NO_x as NO_2)	7.80×10^{-1}	3.47×10^2
(a) Sulfur oxides (SO_x as SO_2)	1.20	1.22×10^3
IPCC greenhouse effect (direct, 100 years) (g equiv. CO_2)		2.18×10^5
(a) Carbon dioxide (CO_2, fossil)	1.00	2.18×10^5
USES 1.0 terrestrial ecotoxicity (g equiv. 1-4-dichlorobenzene)		6.06×10^6
(a) Lead (Pb)	1.10×10^4	1.5×10^6
(a) Mercury (Hg)	1.30×10^7	4.47×10^6

(a): air emission; (w): water emission.

8.4.4.1 Impact Pathway Analysis Applied to the Energy Generation in the Industrial Separation Process

The site-specific environmental assessment has been carried out using the Ecosense model, which was developed to support the assessment of priority impacts resulting from the exposure to airborne pollutants, namely, impacts on health, crops, building materials, and ecosystems as described in Chapter 4.

To cover different pollutants and different scales, Ecosense provides two air transport models completely integrated into the system.

All input data required to run the windrose trajectory model (WTM) are provided by the Ecosense database. A set of site-specific meteorological data must be added by the user to perform the ISCST-2 model. The results of the air dispersion model rely on four basic data sets: 1) meteorological conditions; 2) facility characteristics; 3) location of buildings near the emission sources; and 4) location of receptors (distance to the emission source and elevation in the terrain). The geographic data are crucial for this type of assessment. In this case the Tarragona region in Spain was selected as the area of study and information about elevation, population density, and the meteorological situation was provided.

The physical impacts and, as far as possible, the resulting damage costs are calculated by means of pollutant short-range and long-range transport and conversion models and the exposure–response functions for several receptors (human and ecosystems). These can be selected by the user for each individual grid cell (for the case under study, the grid cell corresponds to the Tarragona region in Spain), taking into account the information on receptor distribution and concentration levels of air pollutants from the reference environmental database (IER, 1997). Table 8.22 lists the results of the most important impacts for human health, crops and ecosystem.

According to this table, no impacts or damages are caused by heavy metal emissions even though the inventory shows these loads in the process (Table 8.19) because the heavy metal emissions are smaller than those manageable by the software (Ecosense). Therefore, in the next section, environmental risk of the heavy metal mercury emissions will be assessed to further analyze the environmental impact of those substances.

8.4.4.2 Fate and Exposure Analysis with Risk Assessment of Mercury for the Electricity Generation in the Industrial Separation Process

An environmental risk procedure implies the inherent capacity of the substances to cause negative effects and the exposition or interaction of these substances within receptors (ecosystems or humans). These aspects are closely related to the fate analysis and the distribution in the environment.

In order to know the fate and future exposition of mercury from the electricity generation in the separation process, the software CalTOX described in Chapter 4 and Example 1 (Section 8.2) was applied to mercury for the area around 1000 m from the emission source for 1 year of continuous emission.

The results of the fate and exposure assessment for mercury emissions are shown in Table 8.23 and the following. According to this model, the concentrations on the compartments are constant for the considered region; however, the exposure is changing according to distance from the point of emission. (The exposure changes proportionally at the distance of the emission.) Table 8.24 shows the exposure media of mercury for the studied region. The daily human doses through several exposure types were calculated based on the modeled compartments' concentration of emissions. The main exposure pathway is air inhalation. Also important is exposure through fish ingestion.

Depending on the level of exposure, in principle adverse health effects can be associated with all substances. In this sense, risk characterization is a dose–response analysis that compares the current human exposure with a defined level of exposure. On the other hand, hazard ratio expresses noncarcinogenic effects as a proportion of an exposure intake rate and a reference dose related to the selected exposure pathways and chronic exposure duration (the hazard ratio for mercury is 8.25×10^{-5}).

TABLE 8.22
Impacts and Damage Assessment for Energy Generation in Process

Receptor	Impacts Unit	Impacts Value	Damages Unit	Damages Value
<u>**Human health**</u>				
Impact: chronic YOLL				
Dose–response functions				
Pollutant: TSP	Years	6.69	mUS$	5.62×10^5
Pollutant: nitrates	Years	3.71×10^{-2}	mUS$	3.71×10^4
Pollutant: sulfates	Years	8.82×10^{-2}	mUS$	7.47×10^3
Impact: acute mortality				
Dose–response functions				
Pollutant: TSP	Cases	6.41×10^{-2}	mUS$	2.00×10^5
Pollutant: nitrates	Cases	3.57×10^{-4}	mUS$	3.54×10^5
Pollutant: sulfates	Cases	1.16×10^{-3}	mUS$	2.69×10^3
Pollutant: NO_x	Cases	5.34×10^{-2}	mUS$	1.66×10^5
Pollutant: SO_2	Cases	1.14×10^{-1}	mUS$	2.64×10^2
<u>**Ecosystem**</u>				
Dose–response functions from: UN-ECE 1997				
Pollutant: SO_2				
Impact: SO_2 exceedance area	km^2 exceedance area	0.00	mUS$	NA
Impact: REW SO_2 exceedance area	km^2 exceedance area	9.44×10^{-5}	mUS$	NA
Impact: RCW SO_2 ecosystem area	km^2 exceedance area	1.25×10^{-3}	mUS$	NA
Pollutant: NO_x				
Impact: NO_x exceedance area	km^2 exceedance area	0.00	mUS$	NA
Impact: REW NO_x exceedance area	km^2 exceedance area	0.00	mUS$	NA
Impact: RCW NO_x ecosystem area	km^2 exceedance area	4.97×10^{-4}	mUS$	NA

REW: relative exceedance weighted; RCW: relative concentration weighted; RDW: relative deposition weighted; NA: not available; m = 10^{-3}.
Based on data from IER (1997).

TABLE 8.23
Time Average Concentration in On-Site Environmental Media

Compartment	Unit	Mercury
Air	mg/m^3	3.13×10^{-10}
Total leaf	mg/kg(total)	1.92×10^{-12}
Ground-surface soil	mg/kg(total)	2.96×10^{-9}
Root-zone soil	mg/kg(total)	2.13×10^{-9}
Vadose-zone soil	mg/kg(total)	1.23×10^{-9}
Ground water	mg/L(water)	6.99×10^{-14}
Surface water	mg/L	2.39×10^{-12}
Sediment	mg/kg	2.09×10^{-9}

8.4.5 Interpretation of the Environmental Impact Analysis for the Industrial Separation Process

The LCI indicates the relevance of fugitive emissions (VOC) in the case of an industrial separation process of isopentane from naphtha. The LCIA in relation to the LCI allows the conclusion that the energy generation sub-process is the most relevant process of those considered. Therefore, this process is selected for further analysis if the potential environmental impacts correspond to actual impacts, i.e., damages.

The IPA shows that main damages are produced by the particles, NO_x and the secondary pollutants nitrates and sulfates. This is in line with the results obtained for the MSWI case study (see Chapter 5). The ERA for mercury points out that there is a very low risk for human health impacts in the neighborhood due to mercury exposure based on the fictitious data used for the industrial separation process example. However, this risk is higher than 10^{-6} and therefore not really acceptable according to guidelines mentioned in Chapter 4. A further reduction of the mercury emissions in electricity generation process is thus recommended.

Altogether, this comprehensive environmental impact analysis gives a much more complete picture of the environmental implications of the industrial separation process than each tool applied independently. Moreover, since the databases are common and a lot of work is needed for their collection, the subsequent application of the different analytical tools seems to be a way to get ahead in the future.

8.5 CONCLUSIONS FROM THE APPLICATIONS

Three examples have been presented in this chapter to evaluate the principle applicability of the strategy outlined in Chapter 6 to integrate LCA and ERA where possible in industrial processes other than the municipal solid waste incineration:

TABLE 8.24
Exposure Media Concentrations for Mercury

Exposure	Air (gases)	Air (dust)	Ground soil	Root soil	Ground water	Surface water
Indoor air (mg/m^3)	1.91×10^{-8}	1.31×10^{-14}	8.18×10^{-15}	2.02×10^{-12}	1.05×10^{-18}	5.61×10^{-16}
Bathroom air (mg/m^3)	0	0	0	0	1.35×10^{-16}	7.21×10^{-14}
Outdoor air (mg/m^3)	1.91×10^{-8}	1.31×10^{-14}	NA	NA	NA	NA
Tap water (mg/L)	0	0	0	0	3.50×10^{-14}	7.97×10^{-11}
Exposed produce (mg/kg)	1.92×10^{-12}	1.32×10^{-18}	9.27×10^{-10}	2.02×10^{-17}	2.49×10^{-11}	5.67×10^{-8}
Unexposed produce (mg/kg)				2.28×10^{-8}	2.41×10^{-11}	5.49×10^{-8}
Meat (mg/kg)	3.73×10^{-9}	2.55×10^{-15}	2.63×10^{-10}	1.94×10^{-18}	2.39×10^{-12}	5.44×10^{-9}
Milk (mg/kg)	1.10×10^{-9}	7.51×10^{-16}	8.83×10^{-11}	8.08×10^{-19}	9.95×10^{-13}	2.27×10^{-9}
Eggs (mg/kg)	0	0	0	0	0	0
Fish and seafood (mg/kg)	0	0	0	NA	0	3.20×10^{-6}
Household soil (mg/kg)	0	0	1.36×10^{-7}	1.12×10^{-7}	0	0
Swimming water (mg/L)	0	0	NA	0	NA	1.59×10^{-10}

NA: not available.

- Example 1 is a direct continuation of the MSWI case study presented through Chapters 1 to 7. In the same way as for waste incineration, site-oriented impact assessment makes sense for landfilling. The site-specific study is carried out as a demonstration project for one pollutant only. More air pollutants could easily be checked by the same scheme; for emissions to water and soil other models need to be used.
- Example 2, on the other hand, shows the clear limitations of site-orientated impact assessment for electronic products and other complex product systems in line with the differentiation made in the introductory section of Chapter 6.
- Example 3 demonstrates the applicability of the strategy to an industrial process in the area of chemical engineering. The subsequent application of the different analytical tools gives a much more complete picture of the environmental implications of the industrial separation process than each tool applied independently.

This means the cases show the principle feasibility and the existing limitations of the integration of life-cycle and risk assessment and clearly indicate that the same basis of data can be used. However, the examples presented here must be put into real applications within a decision-making context to be effective. Some further adaptations — especially simplifications of the links between the different assessment tools — are highly recommended in order to facilitate nonacademic applications. We need to move from theory to practice in this area.

Integrated product policy (IPP) and the new chemicals policy are currently major areas of debate in the EU and have the potential to foster the application of the integrative approach of LCA and ERA presented in this book. Looking into the white paper strategy for a future chemicals policy and into the conclusions of the Council of the European Union on the IPP Green Paper, in several places it is pointed out that interaction is needed, i.e., an integrated approach for exchanging information on the chemical/product, preventing products containing harmful chemicals, avoiding processes applying and generating hazardous substances and, consequently, avoiding emissions of chemicals.

A good starting point could also be environmental risk assessments in which the point of departure for the assessment is usually the chemical, i.e., more or less upstream in the life-cycle, whereas LCA considers the functional unit, i.e., the function that the product delivers, which is further downstream. A weak point in many ERAs is the estimation of the use and disposal emissions of the chemicals. LCA methodology may potentially improve these estimates. Simultaneously, risk assessment methodology may assist in generating upstream information in life-cycle assessments — often a weak point in many LCAs. As proposed and shown in this book, data sharing for an integrated life-cycle and risk assessment seems to be the way to proceed.

REFERENCES

Ciambrone, D.F. (1997). *Environmental Life Cycle Analysis,* Lewis Publishers, CRC Press, Boca Raton, FL.

DTSC (1992a). Guidance for site characterization and multimedia risk assessment for hazardous substances release sites, vol. 2, chap. 2, documentation of assumptions used in the decision to include and exclude exposure pathways, a report prepared for the State of California, Department of Toxic Substances Control, Lawrence Livermore National Laboratory, Livermore, CA, UCRL-CR-103462.

DTSC (1992b). Guidance for site characterization and multimedia risk assessment for hazardous substances release sites, vol. 2, chap. 3, guidelines for the documentation of methodologies, justification, input, assumptions, limitations, and output for exposure models, a report prepared for the State of California, Department of Toxic Substances Control, Lawrence Livermore National Laboratory, Livermore, CA, UCRL-CR-103460.

DTSC (2002). CalTOX version 2.3 (www-pages), Department of Toxic Substances Control of the California EPA, State of California, [http://www.dtsc.ca.gov/ScienceTechnology/caltox.html], July 2002.

Ecobalance (1998). EIME – Environmental Information and Management Explorer, Software, Paris, France.

Ecobalance (1997). TEAM – Tool for Environmental Analysis and Management – Version 2, Software, Paris, France.

Ecobalance (1999). WISARD – Waste-Integrated Systems Assessment for Recovery and Disposal, Software, Paris, France.

IER (1997). Ecosense, Version 2.0, Software, University of Stuttgard, Germany.

UNEP (2003). Global Mercury Assessment, Report, Geneva, Switzerland.

UN-ECE United Nations Economic Commission for Europe (1991). Protocol to the 1979 convention on long-range transboundary air pollution concerning the control of emissions of volatile organic compounds or their transboundary fluxes. Report ECE/EB AIR/30. UN-ECE, Geneva, Switzerland.

U.S. EPA (1989). Risk assessment guidance for superfund volume in human health evaluation manual (part A), Office of Emergency and Remedial Response, EPA/540/1-89/002.

U.S. EPA (1995). Users' Guide for the Industrial Source Complex (ISC3) — Dispersion Models Volumes I-II. EPA-454/B-95-003a, EPA-454/B-95-003b. Triangle Park, NC.