ENVIRONMENTAL LIFE CYCLE ASSESSMENT

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Olivier Jolliet
Myriam Saadé-Sbeih
Shanna Shaked
Alexandre Jolliet
Pierre Crettaz

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Foreword

The pressure that our growing human population is putting on nature and our common global environment is becoming increasingly serious and visible not just in the way that we are impacting the global climate but also in our use of land, water, and nonrenewable resources, our emissions of thousands of chemicals with potential toxic effects on humans and ecosystems, and our contribution to regional environmental problems like acidification and photochemical air pollution. Our responsibility as citizens, enterprises, and political decision-makers to face the challenge and seriously change the way in which we interact with the environment, through our consumption of products and services, becomes more evident every year. Communication about sustainability is widespread and many claims are made about environmental sustainability, but few are substantiated. The field suffers from a lack of factual knowledge about the impacts caused by our use of products and services and about what matters and what is insignificant, and decisions are often made on a poorly informed basis.

We need to be able to put numbers on environmental sustainability in order to support a more qualified debate about what the most sustainable choices are among alternative solutions. Life cycle assessment (LCA) is a central tool to this end. With its system perspective, considering the entire life cycle from the cradle to the grave of the product or technology, and with its broad coverage of environmental impacts, it enables us to reveal and avoid problem shifting in the value chain and between different environmental impacts. It offers us a widely encompassing insight and quantitative information on the environmental sustainability performance of the solutions that we use it to assess. This is why LCA is today the analytical backbone in the European Commission's Strategy for Sustainable Consumption and Production and why it is widely disseminated and used in sustainable building initiatives, sustainability assessment of biofuels and biomaterials, environmental product declarations and ecolabeling, and not least in product development in thousands of companies who use it to focus their ecodesign activities and support decision-making in the development of new and improved product generations.

The successful dissemination of LCA and life cycle–based approaches in many parts of the world is accompanied by a growing demand for skilled professionals who are trained in the use of this methodology, and there is therefore a need for good teaching materials in LCA. I find it difficult to think of a team more qualified to fill this need than that of Professor Olivier Jolliet from the University of Michigan. Professor Jolliet has his professional roots in the European history of LCA method development, where he was involved in research and teaching in the strong Swiss LCA environment already in the early 1990s. During the last decade, he has been an active member of the North American LCA environment, and he has trained hundreds of students over the years. He has played a central role in the international community around life cycle impact assessment method development with a consistent focus on improving the state of the art and disseminating good LCA practice. Together with his team, he has developed several full life cycle impact assessment methods over the years, and he has led various method development working groups under the Society of Environmental Toxicology and Chemistry (SETAC) and the United Nations Environment Programme (UNEP)-SETAC Life Cycle Initiative, which he was instrumental in launching. He is a member of the core team behind the USEtox model for assessment of human and ecotoxicity in LCA and currently leads the UNEP-SETAC Life Cycle Initiative's flagship project on enhancing life cycle impact assessment.

Professor Jolliet is an excellent presenter and pedagogue, and together with his team of coauthors, he has managed to distill these skills into the chapters of this textbook, which in its structure follows the stages of the life cycle assessment methodology. There is a chapter for each stage, offering a thorough introduction to the elements to be performed, accompanied by concrete examples. The book also offers two chapters illustrating the application of LCA to comparative assessments of wastewater treatment technologies and of biobased products. It is a concise and pedagogic book that will serve well for the reader who wants to understand the essentials of LCA, and I give it my strong recommendation.

Michael Hauschild
Professor in Quantitative Sustainability Assessment
Technical University of Denmark

Preface

How can we make sustainability decisions that account for the big picture? Life cycle assessment (LCA) provides such a framework that has been widely adopted in the past decade by governments to inform policy and by proactive companies to identify their most important environmental problems and reduce corresponding impacts.

Building on 25 years of LCA teaching with more than a thousand students worldwide, this textbook meets the needs of professionals and students around the world to teach themselves good practice in LCA and to discover the beauty and limitations of this systems approach.

Several features make LCA a unique sustainability tool worth learning about and applying to many of today's problems:

- First, LCA, in contrast to a regulatory-oriented method, is primarily a voluntary tool that stimulates leading companies and governments to identify their strengths and weaknesses, go beyond basic assumptions and counterproductive opposition between stakeholders, and ultimately find innovative and sustainable solutions toward product and behavior improvement.
- The progress made in LCA since the 1990s has been striking, both in terms of data availability and the scientific quality of impact assessment methods. While the rate of development and innovation will continue to be high in the coming years, LCA has reached a level of maturity such that its studies and approaches are regularly published in the best scientific journal of the fields.
- Most of all, LCA education is fun, because the learners come from a wide range of backgrounds and interests, with one common perspective: approaching sustainability through systems thinking. The beauty of LCA is that despite the 25 years of LCA research, development, and practice, every new study brings its surprises and unexpected results that broaden our understanding.

Our hope and expectation is that this book will constitute the next step toward an open community of LCA learners and teachers who can further share their experiences and lessons.

Additional material is available from the CRC Press website: http://www.crcpress.com/product/isbn/9781439887660; and the Teaching LCA website: http://www.teachinglca.org.

Authors

Pierre Crettaz earned his PhD in environmental engineering at the Swiss Federal Institute of Technology in Lausanne in 2000. Between 2000 and 2005, he worked as a senior scientist at the Swiss Federal Office of Public Health and was responsible for the registration of pesticides with respect to their human health effects. He obtained his MSc in applied toxicology at the University of Surrey in Guildford.

He is currently the head of the biocides section at the Swiss Federal Office of Public Health. He represents Switzerland in different international committees on biocides (European Chemicals Agency, Organisation for Economic Co-operation and Development) and is responsible for the risk assessment of biocidal products such as insecticides, rodenticides, repellents, and disinfectants. Risk management and communication, as well as the efficacy of biocide evaluation, are also part of his key activities.

Alexandre Jolliet is an undergraduate student in the College of Literature, Science, and the Arts at the University of Michigan, where he is studying political science and philosophy with specializations in political economy and development, political philosophy, and ethics. He spends his summers preparing and teaching courses to Spanish secondary school students preparing for their Cambridge English First Certificate in English (FCE) exams at English Summer International Schools in Tarragona, Spain.

Olivier Jolliet is a full professor in life cycle impact and risk modeling at the Department of Environmental Health Sciences in the School of Public Health, University of Michigan. He is also one of the founders of Quantis-International, which provides life cycle assessment (LCA) expertise to companies and governments. His teaching and research aim to (a) enable LCA learners to understand and critically assess the strengths and limitations of an LCA; (b) compare the life cycle risks and benefits of consumer products, foods, and emerging technologies; and (c) model and screen population exposure, intake fractions, and pharmacokinetics of emerging chemicals. He coinitiated the UNEP/SETAC Life Cycle Initiative, a joint initiative of the United Nation Environment Program and the Society for Environmental Toxicology and Chemistry. Dr. Jolliet has been one of the pioneers in applying LCA to food sustainability since the early 1990s, and he has developed comprehensive Life Cycle Impact Assessment Methods, such as the IMPACT2002 method. He is one of the developers of USEtox, the UNEP-SETAC model for comparative assessment of chemicals, and is presently developing an extended method to assess the near-field exposure to chemicals in consumer products. Dr. Jolliet has authored or coauthored 150 peer-reviewed publications and book chapters. He has given more than 40 LCA-related master courses and 20 short courses for professionals, and he has been a primary advisor for 15 PhD students and more than 50 Master's students.

Myriam Saadé-Sbeih earned her PhD in environmental sciences from the University of Lausanne, Switzerland, in 2011, after working for 2 years in the Life Cycle System group of Professor Jolliet at the Swiss Federal Institute of Technology, Lausanne. She is currently a Research Fellow at the Graduate Institute of International and Development Studies in Geneva, Switzerland, and since 2012 has been the scientific coordinator of a research program on the transboundary Orontes River basin, funded by the Swiss Development and Cooperation Agency. Her research focuses on human–hydrosystem coevolution in uncertain contexts, especially in the Middle East. She is a Swiss expert for the United Nations Convention to Combat Desertification (UNCCD).

Shanna Shaked earned her PhD in applied physics from the University of Michigan in 2011 after receiving her BS and BA in physics, astronomy, and mathematics from the University of Arizona in 2002. She also received her MA in teaching from Ithaca College in 2013. She is a coauthor of peer-reviewed publications in journals such as *Environmental Science and Technology* and *International Journal of Life Cycle Assessment*, and she has presented her life cycle assessment research on modeling global impacts of products at numerous international conferences, including as an invited member of a panel discussion and as chair of a special session on regionalization in LCA. She also coauthored a chapter on global life cycle impacts in the 2011 *Enyclopedia of Environmental Health.*

She is currently a lecturer in the Departments of Physics & Astronomy and Environmental Science at the University of California, Los Angeles. She serves on the University of California, Los Angeles Physical Sciences Undergraduate Education Committee to revise and improve undergraduate teaching using evidence-based techniques.

She is passionate about using education to help equalize opportunities by making quality education available to all and by helping more people realize and assess the global impacts of their actions.

Symbols

a_{ij}	Amount of technological process *i* used by process *j*
A	Technology matrix
$\tilde{a}_{ij}$	I/O coefficient, monetary output from sector *i* required to produce \$1 of output from sector *j*
Ã	Economic I/O matrix
AE	Accumulated exceedance
b_{kj}	Elementary flow of substance *k* extracted from the environment or emitted in the environment through process *j*
B	Environmental matrix
$\tilde{b}_{ki}$	Emission factor, elementary flow *k* directly extracted from or emitted to the environment per monetary unit of sector *i*
$\tilde{\mathbf{B}}$	"Satellite" environmental matrix
c	Constant (1E12 ecopoint/year)
C	Cost
C_{DALY}	Cost assigned to one year lost
CF_i	Ecofactor of substance *i* (ecological scarcity method)
$CF_{d,i}^{\text{damage}}$	Damage characterization factor for substance *i* in damage category *d*
$CF_{m,i}^{\text{midpoint}}$	Midpoint characterization factor of substance *i* in the midpoint category *m*
C_i^m	Concentration threshold value of substance *i* in medium *m* (critical volumes)
CTU_e	USEtox comparative toxic units, corresponding to potentially affected fraction of species; PAF-cubic meter-days per kilogram for ecosystem impacts
CTU_h	USEtox comparative toxic units, corresponding to cases of cancer and noncancer for human health impacts
E	Matrix of aggregated emission and extraction factors
$\tilde{e}_{kj}$	Total elementary flow *k* extracted from or emitted into the environment per monetary demand of sector *j*
$\tilde{\mathbf{E}}$	Matrix of environmental emissions and resource extractions from each economic sector over the entire production chain
F_{ai}	Actual flow of substance *i* in the reference area (ecological scarcity method)

F_{ci}	Critical maximum acceptable flow of substance i in the reference area (ecological scarcity method)
F_{ni}	Annual flow of substance i in Switzerland (ecological scarcity method)
$G^{\text{per biocultivated ha-year}}$	Gain (or reduction) in impact per biocultivated hectare and year for a given study
GSD^2	Geometric standard deviation
$\mathbf{I}$	Identity matrix
iF	Intake fraction
K_{ow}	Octanol-water partition coefficient
$MDF_{d,m}$	Midpoint-to-damage characterization factor estimating the damage to the area of protection d caused per unit of the midpoint reference substance of category m
N_d	Normalization value of damage category d
$N_{\mathrm{FU}}^{\text{per biocultivated ha-year}}$	Annual production of functional units per hectare cultivated in the bio-based scenario
ODP	Ozone depletion potential
$\mathbf{P}$	Employment matrix per monetary unit invested in a sector
p_i	Parameter value
$POCP$	Tropospheric ozone concentration increase
$P_{total,r}$	Total population
S_i	Impact score
S^m	Critical volume of medium m (critical volumes)
$S_{\text{bio-based}}^{\text{per FU}}$	Total impact score per functional unit of the bio-based scenario
$S_{\text{conventional fossil}}^{\text{per FU}}$	Total impact score per functional unit of the considered study for the conventional fossil scenario of reference
$S_{\text{conventional fossil}}^{\text{subsituted part per FU}}$	Impact score per functional unit associated only with the part of the conventional fossil product that is substituted by the biofuels
$S_{\text{Human Health}}$	Human health damage score due to traffic (cost internalization)
$(S_{\mathrm{B}} - S_{\mathrm{A}})/S_{\mathrm{A}}$	Relative difference in score between Scenarios A and B
S_d^{damage}	Damage characterization score in damage category d
$S_{d-\text{normalized}}^{\text{damage}}$	Normalized score in the damage category d
S_m^{midpoint}	Midpoint impact score of a category
S_{weighted}	Weighted environmental impact score
s_i	Relative sensitivity of the model output to the input parameter i

$(S_B - S_A)$	Difference in sensitivity between Scenarios A and B
$\boldsymbol{t}$	Vector of created jobs
u_i	Emitted or extracted mass of substance i per functional unit as given in the inventory
$u_{i\ \text{total}\ r}$	Total annual global, continental, or national emissions or extractions in the region r
$\boldsymbol{u}$	Emissions and extraction inventory vector
$\tilde{\mathbf{u}}$	Quantities of emitted substances and extracted resources
U_B	Base uncertainty
U_C	Uncertainty factor over completeness
U_G	Uncertainty factor over geographical correlation
U_L	Uncertainty factor over technological correlation
U_R	Uncertainty factor over reliability
U_S	Uncertainty factor over sample size
U_T	Uncertainty factor over temporal correlation
w_d	Weighting factor
$\boldsymbol{x}$	Total output vector, total amount of goods and services in each sector needed to meet the demand $\tilde{\boldsymbol{y}}$
$\tilde{\mathbf{x}}$	Total output vector, total monetary amount of goods and services in each sector needed to meet the demand $\tilde{\boldsymbol{y}}$
$\boldsymbol{y}$	Demand vector
$\tilde{\boldsymbol{y}}_j$	Amount spent in sector j for providing one functional unit (in $/FU)
$\tilde{\mathbf{y}}$	Economic demand vector
$\mathbf{Z}$	National transactions matrix
ΔC	Cost increase
Δp	Relative change in the model input parameter i
ΔS	Relative change in the model output
η	Ecological efficiency
μ	Geometric mean

Acronyms

ADEME	French Environmental Protection Agency	Agence de l'Environnement et de la Maîtrise de l'Energie
AERM	Rhine-Meuse water agency	Agence de l'eau Rhin-Meuse
BEA	Bureau of Economic Analysis	
BOD5	Biological oxygen demand	
CAS	Chemical Abstracts Service	
CEDA	Comprehensive Environmental Data Archive	
CF	Carbon footprint	
CFC	Chlorofluorocarbon	
CFL	Compact fluorescent lamp	
CH_4	Methane	
CIRAIG	International Reference Centre for the Life Cycle of Products, Processes and Services	Centre international de référence sur le cycle de vie des produits, procédés et services
CML	Institute of Environmental Sciences, Leiden University	Centrum voor Milieukunde
CMLCA	Chain Management by Life Cycle Assessment	
CO_2	Carbon dioxide	
COD	Chemical oxygen demand	
CRA	Comparative risk assessment	
CST95	Critical surface time 95	
CTU	Comparative toxic unit	
DALY	Disability-adjusted life year	
DOC	Dissolved organic carbon	
E3IOT	Environmentally extended input–output table for Europe	
ED	Ecosystem diversity	
EDGAR	Emissions database for global atmospheric research	
EDIP	Environmental design of industrial products	
EIA	Environmental impact assessment	
EINECS	European Inventory of Existing Commercial Substances	
EINES	Expected increase in number of extinct species	

EIO-LCA	Economic Input–Output Life Cycle Assessment	
ELCD	European Reference Life Cycle Database	
ELU	Environmental load unit	
EPA	U.S. Environmental Protection Agency	
EPFL	Swiss Federal Institute of Technology, Lausanne	Ecole polytechnique fédérale de Lausanne
EPS	Environmental priorities strategies	
EQ	Ecosystem quality	
ESU	Energy-Materials-Environment	Energie-Stoffe-Umwelt
ETH Zürich	Swiss Federal Institute of Technology, Zurich	Eidgenössische Technische Hochschule Zürich
EU	European Union	
EXIOBASE	Multiregional environmentally extended supply and Use/ Input–Output database	
EXIOPOL	New environmental accounting framework using externality data and input output tools for policy analysis	
FAO	Food and Agriculture Organization	
FAQDD	Quebec Action Fund for Sustainable Development	Fonds d'action québécois pour le développement durable
FU	Functional unit	
Gabi	Life cycle assessment software	Ganzheitliche Bilanz
GHG	Greenhouse gas	
GJ	Gigajoule	
GW	Global warming	
GWP	Global warming potential	
H/C ratio	Hydrogen to carbon ratio	
HH	Human health	
HHV	Higher heating value	
I/O	Input–output approach	
ILCD	International Reference Life Cycle Data System	
IPCC	Intergovernmental Panel on Climate Change	
IRR	Internal rate of return	

ISO	International Organization for Standardization	
IVAM	Environmental Science Department of the University of Amsterdam	Interfacultaire Vakgroep Milieukunde
JRC	Joint Research Centre	
KNCPC	Korea National Cleaner Production Center	
kWh	Kilowatt-hour	
LCA	Life cycle assessment	
LCC	Life cycle costing	
LCI	Life cycle inventory	
LCIA	Life cycle impact assessment	
LED	Light-emitting diode	
LHV	Lower heating value	
LIME	Life cycle Impact assessment Method based on Endpoint modeling	
LPW	Lumens per watt	
LRV	Swiss Regulation on Air Pollution Control	Luftreinhalte-Verordnung
LUCAS	LCIA method Used for a CAnadian-Specific context	
MFA	Material flow analysis	
MIET	Missing inventory estimation tool	
MJ	Megajoules	
MRIO	Multiregional input–output	
N_2O	Nitrous oxide	
Nm^3	Normal cubic meters	
NMVOCs	Nonmethane volatile organic compounds	
NOx	Nitrogen oxides	
NPP	Net primary production	
NREL	National Renewable Energy Laboratory	
OFEFP	Swiss Federal Office for the Environment, Forests, and Landscape	Office Fédéral de l’Environnement des Forêts et du Paysage
PAH	Polycyclic aromatic hydrocarbon	
PC	Polycarbonate	

PDF	Potentially disappeared fraction of species over one square meter and in one year	
PET	Polyethylene	
PM	Particulate matter	
PM_{10}	Particles ≤ 10 μm in aerodynamic diameter	
$PM_{2.5}$	Particles ≤ 2.5 μm in aerodynamic diameter	
R&D	Research and development	
RA	Risk assessment	
RE	Resource and ecosystem services	
ReCiPe	LCIA method as a “recipe” to calculate life cycle impact category indicators	
RIVM	Dutch National Institute of Public Health	
RMIT	Royal Melbourne Institute of Technology	
SETAC	Society of Environmental Toxicology and Chemistry	
SFA	Substance flow analysis	
SIAAP	Interdepartmental Syndicate for the Sanitation of Greater Paris	Syndicat Interdépartemental pour l’Assainissement de l’Agglomération Parisienne
SimaPro	Life cycle assessment modeling software	
SLCA	Social life cycle assessment	
SME	Small and medium enterprises	
SO_2	Sulfur dioxide	
SO_x	Sulfur oxides	
SPOLD	Society for the Promotion of Life Cycle Assessment	
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin	
tDM	Ton dry matter	
TOC	Total organic carbon	
TPM	Total particulate matter	
TRACI	Tool for the Reduction and Assessment of Chemical and other environmental Impacts	

TREIC	Tracking environmental impacts of consumption
UCTE	Union for the Coordination of Transmission of Electricity
UNEP	United Nations Environment Programme
URL	Uniform resource locator
USD	US dollar
USES-LCA	Uniform system for the evaluation of substances-life cycle assessment
USEtox	UNEP-SETAC toxicity model
WBCSD	World Business Council for Social Development
WF	Water footprint
WIOD	World input–output database
WMO	World Meteorological Organization
WRI	World Resources Institute
WSI	Water stress index
WULCA	Water use in life cycle assessment
WWTP	Wastewater treatment plant

1 Introduction

Olivier Jolliet, Gabrielle Soucy, Shanna Shaked, Myriam Saadé-Sbeih, and Pierre Crettaz

1.1 PRIORITIES FOR THE ENVIRONMENT

Environmental issues are playing an ever-increasing role in the decision-making process at every level: political, economic, industrial, and individual. More than just a passing trend, the increasing attention given to environmental problems stems from a basic observation: because of its limited capacity to absorb the effects of human activities, the environment sets a limit to society's development. This limit has already been reached in many regions of the planet (UNEP 2012).

Sustainability concepts are constantly discussed in the headlines, but it is considerably harder to take action. To ensure a sustainable future, statements and studies must be followed by meaningful actions that effectively reduce environmental impact, and which can even improve the situation. For an action to be efficient, three conditions must be fulfilled:

- Technological solutions must be available.
- Different solutions must be prioritized and best practices selected, accounting for environmental efficiency, cost, and resulting economic constraints.
- Actions should be optimized to further reduce impacts.

Life cycle assessment (LCA) is a decision-making tool which specifically addresses this need of selecting and optimizing available technological solutions. Doing so is fundamental when financial resources are limited, as Barlow (1993, p. 10) stated in a slightly provocative way: "The problem is not one of how to tackle the individual problem—the engineering is either available or can be developed to deal with that. Rather the problem is how to decide priorities. The world just cannot afford to do everything." LCA is a complement to technological developments, since it highlights which processes should be improved in priority order.

LCA is particularly relevant from a sustainability perspective, because it covers the entire life cycle of a product or service, avoiding that local improvements only result in shifting the environmental impact elsewhere. LCA differs from other environmental methods by linking environmental performance to functionality, quantifying the pollutant emissions and the use of raw materials based on the function of the product or system.

The expression *life cycle assessment* conveys the breadth of this approach, encompassing all the impacts of a product from the design stage to its final disposal. LCA is called *ecobalance* in some other languages (e.g., *Oekobilanz* in German), emphasizing the quantified balance and inventory of polluting emissions and resource extraction.

1.2 CRITICAL APPROACH, OBJECTIVES, AND BOOK STRUCTURE

1.2.1 Being Critical

Even though LCA has many advantages, it is not devoid of shortcomings. Analogous to economic accounting for the estimation of a product's actual cost, ecological accounting requires a certain number of assumptions that must be logical and coherent. Some applications of LCA have been harshly criticized, suggesting that a given LCA method was selected to obtain the results expected by the sponsor of the study.

For this environmental tool to have the greatest robustness, one must be able to identify the potential biases of a study. The objective of this book is to explain how to identify the critical aspects of an LCA and how to use consistent criteria to realize and evaluate an LCA independently of individual interests.

1.2.2 Objectives

This book aims to enable the reader to

- Understand LCA methodology
- Become familiar with existing databases and methods based on the latest results of international research
- Be able to analyze and criticize a completed LCA
- Be able to apply LCA methodology to a simple case study

1.2.3 Book Structure

First, Chapter 2 presents the general principles and characteristics of LCA, along with the first application to a simple example. Subsequent chapters detail each phase of the LCA methodology: the goal and scope definition (Chapter 3), the emissions and extractions inventory (Chapter 4), the analysis of their environmental impacts (Chapter 5), and the interpretation of results (Chapter 6). Finally, the last chapters (7–9) provide various detailed examples of LCA application and analysis. At each step, problems and solutions are illustrated with concrete examples from diverse fields such as water management, the automotive industry, electronics, and packaging.

1.3 BACKGROUND AND STANDARDIZATION

This section outlines the historical development of LCA (Table 1.1) and the corresponding standards for good practice determined by the International Organization for Standardization (ISO; Table 1.2). The 1972 publication of *Limits to Growth* (Meadows and Club of Rome 1972) by the then recently founded Club of Rome spread the concept of a limit to growth and development based on predictions of limited resource availability. Twenty years later, this idea of a limit was confirmed, but in the form of the environment's inability to absorb all polluting emissions (Meadows et al. 1992).

The 1973 energy crisis strongly stimulated the conducting of energy balance assessments, in which energy consumption was tracked for a process or system.

TABLE 1.1
Historical Dates of LCA Development

1972	Publication of *Limits to Growth* by the recently founded Club of Rome, which broadly increased awareness of the limited availability of resources and development based on simulations. Early LCA studies on bottle packaging in the United States.
1973	Energy crisis: Generalization of the balance approach (mainly energy balance).
1977	First life cycle impact assessment method: Swiss ecoscarcity.
1984 and 1991	Ecological balance of packaging materials I (Bus 1984; Habersatter and Widmer 1991): Comprehensive packaging LCA that also provided data on energy and materials, acting as a precursor to existing LCA databases.
1992	Club of Rome's founder claimed that the first limitation encountered was typically environmental pollution rather than lack of resources.
1992	CML guide to LCA published by the University of Leiden (the Netherlands) (Heijungs et al. 1992). Rather than focusing simply on air, water, or soil damage, this guide organized environmental impacts into effect-oriented categories, such as acidification and climate change.
1993	LCA "code of practice" published by SETAC (Society of Environmental Toxicology and Chemistry) (SETAC 1993). SETAC is one of the main international scientific organizations involved in developing structural aspects of LCA through various SETAC working groups.
1996	Creation of *International Journal of Life Cycle Assessment.*
1997–2006	ISO (International Organization for Standardization) published a series of ISO 14000 norms on LCA, in response to the demand to internationally harmonize various methodologies used in LCA. The most recent ISO 14040/14044 standards were published in 2006 (ISO 2006).
2002	Launch of the Life Cycle Initiative, a collaboration between UNEP (United Nations Environmental Program) and SETAC.
2003	Ecoinvent life cycle inventory database released by the ecoinvent center within the Swiss Federal Institutes of Technology.
2008	USEtox toxicity model by UNEP-SETAC Life Cycle Initiative.
2008	Marked interest by large global distributors and manufacturers in life cycle approaches and evaluation of their products.
2009	Extension of LCA application to non-OECD countries—conference on life cycle management in South Africa.
2010	ReCiPe released as a successor to the impact assessment methods Eco-indicator 99 and CML 2002 (Goedkoop et al. 2009).
2011	Global guidance principles for LCA databases by UNEP-SETAC Life Cycle Initiative.
2012	European impact assessment method (Hauschild et al. 2012) in conjunction with a new database proposed by the Joint Research Center (JRC) (Sala et al. 2012) (http://www.mdpi.com/ 2071–1050/4/7/1412).
2012	IMPACTWorld+ released as a successor to IMPACT2002+.
2013–present	Flagship project of the Life Cycle Initiative on global guidance for LCA methods with initial focus on climate change, water use, land use, particulate matter impacts, and LCIA framework.

TABLE 1.2
Selected ISO 14000 Standards on Environmental Management and LCA

Key Standards on Environmental Management	
ISO 14001	Environmental management systems—Requirements with guidance for use (2004)
ISO 14004	Environmental management systems—General guidelines on principles, systems and support techniques (2004)
ISO 14020–25	Environmental labels and declarations (1999–2006)
ISO 14031	Environmental management—Environmental performance evaluation—Guidelines (1999)
ISO 14040	Environmental management—Life cycle assessment—Principles and framework (2006)
ISO 14050	Environmental management—Vocabulary (2009)
Additional Standards Specific to LCA	
ISO 14044	Environmental management—Life cycle assessment—Requirements and guidelines (2006)
ISO 14046	Environmental management—Water footprint—Principles, requirements and guidelines (2014)
ISO 14048	Environmental management—Life cycle assessment—Data documentation format (2002)
ISO 14049	Environmental management—Life cycle assessment—Examples of application of ISO 14041 to goal and scope definition and inventory analysis (2000)

To cover a broader set of environmental impacts, the need for the accounting of pollutant emissions to air, water, and soil also became apparent. This led to methodological developments, initially within the packaging industry, which were eventually applied to all economic sectors, as it turned out that the product often had a much larger impact than its packaging.

Eventually, the LCA combined these various types of accounting into a function-based analysis. Three organizations were and are involved in the development and standardization of LCA: the Society of Environmental Toxicology and Chemistry (SETAC), the United Nations Environment Program (UNEP), and the ISO. Starting in the early 1990s, SETAC offered a scientific exchange platform for LCA developments, still continuing today through conference presentations and working groups. Since 2002, the Life Cycle Initiative has been an important institutional framework for the development of LCA methods and their use in industry (website provided in Appendix I). Launched by SETAC and UNEP, this initiative aims to develop and disseminate practical tools for evaluating solutions, risks, advantages, and disadvantages associated with products and services throughout their life cycle. The first phase spanned the period from 2002 to 2007, developing consensus on life cycle approaches. This was followed by a second phase from 2007 to 2011, with the aim of spreading awareness and use of life cycle approaches throughout the world. The third phase (2012 to date) is developing consensus on impact indicators and providing guidance for organizational LCA, which considers the life cycle impacts of a given company or organization, including the supply, use, and disposal of its products and services.

ISO produces international standards for most technological fields. ISO standards are adapted to industrial applications and result from a consensus between experts from various backgrounds, including industry, technology, economy, and academia. During the 1980 and 1990s, ISO published over 350 standards relating to environmental issues. In particular, the ISO 14000 series on environmental management systems was edited, updating and providing a framework for businesses to manage the environmental impact of their activities and to measure their environmental performance.

The ISO 14040 series (14040 to 14049) is devoted to LCA (Table 1.2). The first standard (ISO 14040) establishes the guidelines to perform an LCA. ISO 14044 replaced ISO 14041, 14042, and 14043 in 2006 to describe the phases of inventory, impact assessment, and interpretation (Finkbeiner et al. 2006). ISO 14046:2014 provides guidelines for LCA-based water footprint assessment of products, processes, and organizations. Examples of its application are presented in ISO 14047 and 14049, and ISO 14048 describes data documentation format. For carbon footprints, ISO and the Greenhouse Gas Protocol from the World Resources Institute (WRI) and World Business Council for Social Development (WBCSD) provide more detailed recommendations (WRI and WBCSD 2011).

1.4 USE OF THE LCA TOOL

While the method and its use mainly expanded in Europe and Japan until the mid-2000s, LCA is also increasingly used in North America and developing countries. This is largely due to the interest of major international distributors that want to better assess the sustainability of their products.

In terms of the application of LCA (Cooper and Fava 2006), it can be used industrially for a variety of purposes, including support of a corporate strategy (63% of respondents), research and development (62%), and the design of products or processes (52%). LCA is also used in education (46%) as well as for labels and product descriptions (11%). The importance of LCA studies is increasing as companies increasingly apply them to their own products and require LCA data from their suppliers. LCA has been increasingly used to perform meta-analyses, reviewing all available LCAs about a given topic (e.g., solid waste treatment options; use of biomass) (see Chapter 9). A more detailed analysis of the most prominent LCA application domains is presented in Section 7.3.

2 General Principles of Life Cycle Assessment

Olivier Jolliet, Gabrielle Soucy, Shanna Shaked, Myriam Saadé-Sbeih, and Pierre Crettaz

This chapter defines the life cycle assessment (LCA), its goals, and key phases. It explains the main characteristics of LCA and compares them with other environmental analysis tools. A real-life example illustrates the approach by presenting a comparison between different types of cups used in stadiums. At the end, two exercises encourage the reader to apply and practice the topics covered in the chapter.

2.1 DEFINITION OF THE FOUR LCA PHASES

LCA evaluates the environmental impact of a product or service (sometimes referred to just as a product for brevity); the assessment is based on a particular function and considers all life cycle stages. It helps identify where environmental improvements can be made in a product's life cycle and aids in the designing of new products. Primarily, this tool is used to compare various products, processes, or systems, as well as the different life cycle stages of a particular product.

According to the definitions provided in the International Organization for Standardization (ISO) standards and by the Society of Environmental Toxicology and Chemistry (SETAC), an LCA consists of a goal and scope definition, inventory analysis, impact assessment, and interpretation of results (Figure 2.1). These four phases are defined as follows:

1. In the *goal and scope definition* (Chapter 3), the problem is described and the objectives and scope of the study are defined. A number of crucial elements are determined at this point: the function of the system, the functional unit on which the emissions and the extractions will be based, and the system boundaries. The base scenario and the alternatives are described in detail.
2. In the *inventory analysis* (Chapter 4), the polluting emissions to air, water, and soil are quantified, as well as the extractions of renewable and nonrenewable raw materials. The resource use required for the function of the system is also determined here.
3. The *impact assessment* (Chapter 5) evaluates the environmental impacts due to the inventoried emissions. It can be broken down into the following steps (Jolliet et al. 2004):
   a. *Selection* of the impact categories, category indicators, and characterization models

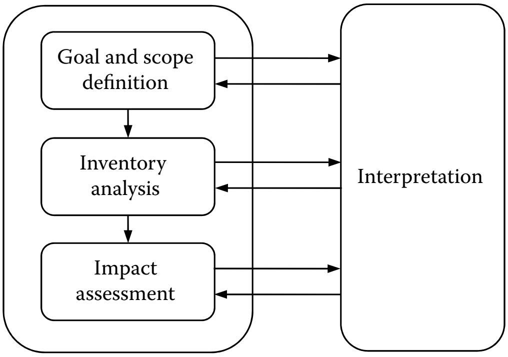


FIGURE 2.1 The four iterative phases of life cycle assessment.

   b. *Classification* of the emissions that contribute to each environmental impact category (global warming, human toxicity, ecotoxicity, resource use, etc.)
   c. *Midpoint characterization* weights and aggregates the emissions into midpoint impact categories
   d. *Damage characterization* aggregates impact categories into damage categories (damage to human health, ecosystem quality, resources, etc.)
   e. An additional *normalization* step may be carried out to show the contribution of the studied product as a fraction of the global impact in a given impact category
   f. Finally, the impact assessment can be completed with a socially based weighting to account for the relative importance—or, ideally, damage—of the midpoint impacts
4. The *interpretation* (Chapter 6) is where the results obtained so far are interpreted and the uncertainties are evaluated. The key parameters and improvement options can be identified using sensitivity studies and uncertainty propagation, and a critical analysis evaluates the influence of the chosen boundaries and hypotheses. Finally, the environmental impacts can be compared with economic or social impacts.

   These four phases of LCA will be examined in detail in the following chapters, but it is important to note that they have not all reached the same level of maturity.
   a. The goal and scope definition phase is well developed.
   b. New developments are required to further the inventory phase, particularly in *allocation*, where it is necessary to allocate emissions and extractions from one product to another coproduct (Section 4.5). Also, data availability and reliability need to be improved.
   c. Major progress has recently been made in impact assessment. Analysis frameworks are now well defined in most impact categories. Midpoint characterization factors are now available for classical categories such as global warming and acidification, and a first consensus at international

level has been reached for toxicity impacts. Further work on spatialization is ongoing. Initial damage characterization methods are available but require additional refinements. Many questions remain undecided regarding weighting and the final evaluation of damages.

d. Few studies include an uncertainty analysis at the interpretation phase, but there have been recent developments in uncertainty methods that can be applied to LCA. This gap should be filled as a priority in future LCA practice.

Chapters 7 and 8 provide a complementary overview, identifying the key point and critical issues of each LCA phase and illustrating them through a comprehensive application of LCA to urban sewage sludge treatment.

LCA has been increasingly used to perform meta-analyses, reviewing all available LCAs about a given topic (e.g., solid waste treatment options and use of biomass) to provide ranges of main environmental life cycle indicators. Chapter 9 presents an example of such a meta-analysis applied to the life cycle environmental impacts of bio-based products.

2.2 PERFORMING AN LCA

2.2.1 Iterative Method

It is strongly recommended to perform an LCA in two steps:

1. The *preliminary evaluation* or *screening* is a quick and simple analysis where the order of magnitude of each life cycle stage contribution is evaluated. An initial sensitivity study gives an indication of the key processes and impacts so that less time is spent on aspects which have a negligible overall contribution.
2. Secondly, a more detailed analysis is performed by repeating in greater depth the goal and scope definition, inventory, and impact assessment phases. The information gathered in the preliminary evaluation is used to identify emissions, processes, and stages with the greatest environmental impacts that need to be further explored in priority order. The final interpretation phase includes a detailed sensitivity study and uncertainty analysis. The study may be finalized by a comparison of environmental impacts with socioeconomic performance.

2.2.2 Calculations by Hand and Using Software

Preliminary calculations, generally for energy consumption and CO_2 emissions, can be made by hand or by using a spreadsheet. When more substances in the inventory are considered, it is recommended to use software specifically designed for LCA, while still carrying out energy and CO_2 balances by hand as a check (see Section 4.2.2). The most commonly used software programs are described in Section 6.7 and Appendix II.

2.3 CHARACTERISTICS SPECIFIC TO LCA AND COMPARISON WITH OTHER ENVIRONMENTAL ANALYSIS TOOLS

2.3.1 Characteristics Specific to Life Cycle Assessment

Characteristics specific to LCA are as follows:

- LCA focuses on the environmental impacts of a product or service. When choosing among various alternatives, the final decision is made by combining the results of the LCA with other aspects such as costs, social implications, economic performance, and technical feasibility. These aspects are evaluated with other complementary analysis tools, such as life cycle costing (Section 6.8.1).
- LCA links these environmental impacts to the system's function, which facilitates a comparison between alternatives.
- Quantified balances are made over the entire life cycle of a product or service, from cradle to grave, from raw material acquisition to waste management.
- LCA accounts for all major environmental issues known today (global warming, resource extraction, impacts of toxic substances on humans and ecosystems, land use, etc.).

A partial LCA can also be conducted by considering only the direct activities of a company from gate to gate. Due to global supply chains, however, this does not yield a complete overview of the considered system and can lead to biased results. If a company creates a product, it is essential to account for the entirety of the product's life cycle activities to avoid suggesting local environmental improvements that effectively export pollution to other parts of the life cycle. These life cycle activities take place upstream from the business in question as well as downstream, including, for example, the type of material that needs to be extracted, the electricity required during the use stage, or the disposal required in waste management.

2.3.2 Comparison with Other Environmental Analysis Tools

LCA provides information that can be used by governments, businesses, and consumers when making a decision. Other decision-making tools are available, each with its specific role and set of complementary information. Figure 2.2 illustrates the positioning of LCA in relation to other environmental instruments and other more general approaches to sustainable development.

In general, actions and policies are directly influenced by overarching concepts and procedural environmental methods. Such procedural tools include *ecolabeling*, which labels a set of products or services within a specific functional category as environmentally friendly, and *environmental audits*, which assess the environmental performance of an individual business and provide follow-up suggestions. An *environmental impact assessment* (EIA) focuses on predicting the impact of a planned installation at a precise location; thus, it corresponds more to a legal procedure than a quantitative tool.

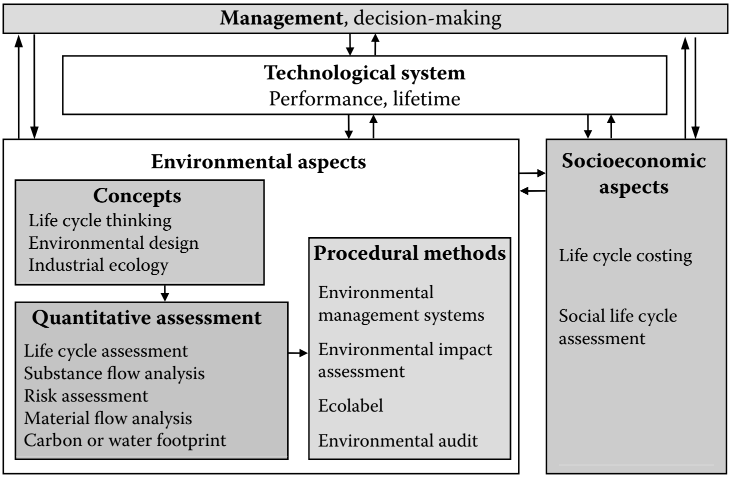


FIGURE 2.2 Relationships between various tools to assess the sustainability of a good or service, inspired by Wrisberg et al. (2002).

These procedural methods are often based on environmental quantitative assessment tools, the most commonly used of which are described as follows:

- A *life cycle assessment* quantifies a large number of resource uses, substance flows, and environmental accumulations to estimate multiple environmental impacts associated with a given function.
- A *substance flow analysis* (SFA) quantifies the flows and environmental accumulation of either a single substance, such as mercury, or a group of substances, such as inorganic nitrate compounds.
- A *risk assessment* (RA) studies the risk or the probability of severe impacts occurring from an installation (such as a nuclear power plant) or the risks of using a chemical substance.
- A *material flow analysis* (MFA) tracks the flow of material in the economy of a given region. This is usually a raw material, such as paper, glass, concrete, or plastics.
- A *carbon footprint* (CF) determines the direct and indirect emissions of greenhouse gases due to a product, a human activity, or a business.
- A *water footprint* (WF) determines the impacts associated with water as an area of concern, including water usage and environmental exposures related to water quality.

These tools cover a range of objects, scales, and effects (Table 2.1), and some basic elements can be common to several tools.

2.3.2.1 Comparison between Substance Flow Analysis and LCA

SFA focuses on the transfers among various environmental media (e.g., air, water, or soil) of a single substance associated with a given region or industry, whereas LCA

TABLE 2.1
Main Characteristics of Environmental Analysis Tools

Tool	Object of Study	Scale and Scope	Considered Substances and Impacts	Basis for Comparison	Basic Elements
Life cycle assessment (LCA)	Product or service	Global or regional Entire life cycle	Many substances Multiple impacts on humans and ecosystems	Function of the product or service	Mass balance Multimedia model Effects assessment
Substance flow analysis (SFA)	Polluting substance	Regional or global Substance cycle	Single substance No impact	Given time and region	Mass balance Multimedia model
Risk assessment (RA)	Installation or chemical substance	Local or regional Selected stage	Relevant substances Toxicity	Maximum level of risk	Multimedia model Effects assessment
Material flow analysis (MFA)	Raw material or compound	Regional or national Material life cycle	Single or multiple material No impact	Given time and region	Mass balance Material flow tracking
Carbon footprint (CF)	Product, activity, or company	Global Entire life cycle	Greenhouse gases Climate change	Product function, activity, or company	Mass balance Global warming potential
Water footprint (WF)	Product, activity, or company	Local or regional Most important life cycle stages	Water consumed and water-related exposure Water quantity and quality-related impacts	Product function, activity, or company	Water balance Consumption Competition Adaptation
Environmental impact assessment (EIA)	New localized activity	Local scale Local activity	Highly variable	Local carrying capacity	Highly variable

considers the environmental fate of a large number of substances associated with a product function, and then estimates impacts by accounting for the multiple effects of these substances. Both methods rely on mass balances to perform calculations and validate the emissions and extraction inventory.

2.3.2.2 Comparison between Environmental Impact Assessment and LCA

The comparison between LCA and EIA is presented in Figure 2.3, depicting the complementary temporal and spatial scales over which each tool is usually applied. LCA is applied to a product or service, from resource extraction (cradle) to the end of life (grave), on a global scale if needed, whereas the analysis in an EIA is carried out on a specific site. For a site-specific evaluation, an EIA better considers conditions particular to the area, such as the number of inhabitants close to the site, the distance between the site and residential neighborhoods, and the presence of certain ecosystems. The ideal tool capable of covering local and global scales from cradle to grave does not yet exist because of the amount of data and computational resources needed, but one prototype of a multiscale multimedia model is currently in development at the University of Michigan (Wannaz et al. 2012).

2.3.2.3 Comparison between Risk Assessment and LCA

LCA has some common elements with RA, particularly in the evaluation of human toxicity and ecotoxicity. Both methods consider the transfer of pollutants among air, water, soil, and food, and they both use dose–response models to quantify the impacts of human and ecosystem exposure to substances. RA differs from LCA in that it is a regulatory-oriented approach, in which hazard-based indicators are based on conservative assumptions and include safety factors to ensure that exposure levels are substantially below the no-observable-effect levels. Such indicators, however, may not provide a consistent basis for relative comparison across chemicals. Rather than maximum estimates, LCA aims to assess comparative risk and the

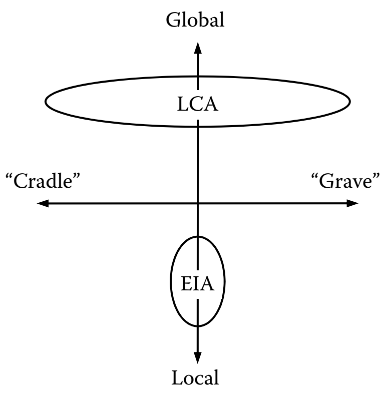


FIGURE 2.3 Comparison between environmental impact assessment and life cycle assessment, based on production cycle covered and localization scale. "Cradle" refers to the beginning of the life cycle (raw material extraction), and "grave" refers to the end of the life cycle (waste disposal).

average contribution of a product or service to a number of environmental impacts (Pennington et al. 2006). Local maximum permissible concentrations or acute toxicity events are thus not usually addressed by an LCA.

2.3.2.4 Comparison between Material Flow Analysis and LCA

MFA and LCA both use mass balance modeling. An MFA merely tracks material flows in a region, whereas an LCA uses these flows in modeling the economic system and unit processes, calculating the emissions and extractions of raw materials related to these material flows.

2.3.2.5 Comparison between Carbon Footprint and LCA

The CF is simply the global warming component of the LCA, and can thus be applied to a product, activity, or company. While an LCA can estimate how various scenarios can shift impacts among different impact categories, a CF focuses solely on the greenhouse effect category.

> In summary, an LCA quantifies material flows throughout the life cycle of a product or service, from which the impacts can be estimated for a comprehensive set of environmental impact categories. LCA is the only method to relate multiple environmental impacts to the function of a product or service.

The life cycle concept is not limited to environmental impacts; the results of an environmental LCA can be combined with those of an economic analysis (Section 6.8.1), a technical analysis (life cycle engineering, Lundquist et al. 2000), or a social analysis (social LCA, Section 6.8.5), thereby integrating the different aspects of sustainability.

2.4 SIMPLE APPLICATION: COMPARING DIFFERENT TYPES OF CUPS

This section presents a comparison among different types of cups to illustrate how an LCA is carried out. The basic hypotheses for this example were adapted from Bättig (2002), where single-use cups are compared with multiuse cups. Chapter 8 presents a more elaborate case study demonstrating the application of LCA, comparing different options for sewage sludge treatment.

2.4.1 Goal and Scope Definition of Cup Case Study

The main objective of this LCA is to compare the environmental impacts of different types of cups used in stadiums during sporting or cultural events.

The functional unit used as a basis for comparison must be common to all scenarios and represent the considered function (Section 3.3). Since the purpose of the cup is to contain a certain volume of drink, the corresponding functional unit is one use of a 300 mL cup. Therefore, the various substance emissions and resource extractions listed in the inventory will be calculated for one use of one cup.

TABLE 2.2
Processes Included within the System Boundaries

Single-Use Cup	Multiuse Cup
Cup manufacturing	Cup manufacturing
Transportation (from production site to stadium)	Transportation (from production site to stadium)
Cleaning of the stadium	
	Transportation (to and from washing facility)
	Washing of the cup
Elimination (incineration)	Elimination (incineration)

The system processes considered are presented in Table 2.2. For a single-use cup, this includes the manufacturing of the cup, its transportation to the stadium where the event takes place, the cleaning of the stadium using air blowers, and the elimination of the cup. For a multiuse cup, the stadium cleaning is assumed to be unnecessary, since cups are collected and reused rather than left on the stadium floor, but washing the cup and its transportation to and from the washing facility must be included. The production and use of detergent for washing the cup are not considered here. The manufacturing and elimination of the infrastructure for cup production are excluded because their impact per cup produced over the entire lifetime of the production infrastructure is considered negligible.

Some of the key parameters in this study are the transportation variables and the number of times a cup is reused, both of which can vary considerably depending on event logistics and user behavior. The important transportation variables are the distance traveled, the mode of transportation, and the size of the load, all of which are necessary data for the calculation of the impacts of any transportation. The number of cup uses is also an important parameter, since any process that occurs only once in a cup's life (such as raw material extraction, manufacturing, and elimination) has its impacts distributed over each use. For a multiuse cup made of polycarbonate (PC) or polypropylene that can be reused 150 times, the impacts of the one-time processes (shaded in Table 2.2) should be divided by 150 to yield the impact contribution of one use of a cup (assuming no losses). The actual number of reuses, however, is much lower due to losses during the event or in transit (cups may be damaged, discarded, etc.). This can be accounted for by introducing a loss percentage.

To examine the influence of these parameters on the total impacts, the material, number of uses, and transportation parameters are varied as follows:

- A paper cup, used once
- A polyethylene (PET) cup, used once
- A PC cup, used 150 times, without accounting for transportation between the stadium and the washing facility
- A PC cup, used 150 times, with 5% losses at every event, and 50 km transportation distance to be washed (round trip to the cleaning facility by car loaded with 1000 cups)

Note that the last scenario derives from a sensitivity study carried out by the authors of this book and was not presented in the original study.

2.4.2 Inventory Analysis of Cup Case Study

The inventory quantifies the pollutant emissions to water, air, and soil, as well as the extractions of raw material from the environment, over all processes in the life cycle of each scenario. It does so by first quantifying the main intermediary flows required per cup use (e.g., key transportation distances and amounts of paper or PET used per functional unit), and then finding the pollutant emissions and resource extraction factors associated with each of these flows (see Section 4.1 for more details on the inventory). Table 2.3 shows an excerpt of this inventory for each scenario.

The PC multiuse cup, without losses or transport to washing facility systematically has the lowest emissions and extractions per functional unit. For the remaining three scenarios, it is not possible to define a ranking from the inventory results alone; the single-use paper cup has the highest emissions of cadmium in air and hexavalent chromium in water, while the single-use PET cup requires the extraction of a large amount of crude oil. When losses and transportation to the washing facility are taken into account, the PC cup emits more CO and N_2O in air than the other scenarios.

Therefore, it is difficult, based solely on the inventory results, to draw conclusions about the relative impacts of the different scenarios, or about the processes and emissions that contribute most to these impacts. This is precisely the purpose of the impact assessment phase, as demonstrated in the following section.

2.4.3 Impact Assessment of Cup Case Study

The impact assessment phase estimates the impacts of the inventory's emissions and extractions on various areas of protection (human health, ecosystem quality, natural resources, etc.). Different impact assessment methods can be used for this evaluation, each of which uses different models to calculate environmental impacts by category. In this example, the Eco-indicator 99 method is used, which is described in more detail in Section 5.5, along with other impact assessment methods.

Based on Eco-indicator 99 calculations, the single-use PET cup has the highest damage score for all three damage categories considered: human health, ecosystem quality, and resource degradation (Figure 2.4). The multiuse PC cup (assuming no losses or transportation to the washing facility) results in the least impact. The other two scenarios fall in the middle, with the paper cup having slightly less impact than the PC cup with losses and transportation for the three damage categories considered.

Since the relative ranking of scenarios is identical in all three damage categories, we simplify the remaining discussion by using the total aggregated impact. (Section 5.2.3 explains how different categories can be combined into a single score.) As defined in the Eco-indicator 99 method (Section 5.5.3), the total impact is calculated as a weighted sum of each area of protection. Here, we express this total impact score by contribution from each process (Figure 2.5).

TABLE 2.3
Excerpt from the Inventory of Pollutant Emissions and Resource Extractions for Each Scenario (per Functional Unit, i.e., One Use of One 300 mL Cup)

Substance	Unit	Paper	PET	PC	PC with Transportation and Losses
Emissions to Air					
Benzo[*a*]pyrene	g	3.4×10^{-9}	3.2×10^{-9}	4.9×10^{-10}	3.0×10^{-9}
Cd	g	4.4×10^{-7}	2.4×10^{-7}	1.1×10^{-8}	3.9×10^{-8}
CH_4	g	0.0277	0.0358	0.0016	0.0279
CO	g	0.013	0.148	0.006	0.277
CO_2	g	9.2	18.2	0.9	18.5
Hg	g	2.3×10^{-7}	2.6×10^{-7}	1.2×10^{-8}	1.2×10^{-7}
N_2O	g	2.0×10^{-4}	1.1×10^{-4}	1.0×10^{-5}	9.1×10^{-4}
NH_3	g	2.0×10^{-4}	9.0×10^{-6}	1.0×10^{-6}	6.0×10^{-6}
NMVOC	g	0.029	0.294	0.011	0.172
NO_x	g	0.044	0.161	7.0×10^{-3}	0.089
Particles	g	0.014	0.028	1.0×10^{-3}	0.011
Pb	g	2.7×10^{-6}	2.4×10^{-6}	3.0×10^{-7}	1.3×10^{-5}
SO_x	g	0.044	0.185	7.0×10^{-3}	0.074
Emissions to Water					
Al	g	0.00125	0.00155	0.00011	0.00078
As	g	2.5×10^{-6}	3.1×10^{-6}	2.1×10^{-7}	1.7×10^{-6}
BOD	g	0.0554	0.0096	0.0003	0.0025
Cr (VI)	g	1.3×10^{-6}	3.7×10^{-10}	4.1×10^{-11}	2.0×10^{-10}
Cu	g	6.6×10^{-6}	7.8×10^{-6}	5.2×10^{-7}	4.3×10^{-6}
NH_4^+	g	6.3×10^{-5}	2.6×10^{-4}	1.0×10^{-5}	4.7×10^{-4}
Ni	g	6.5×10^{-6}	8.0×10^{-6}	5.2×10^{-7}	4.5×10^{-6}
Pb	g	9.7×10^{-6}	1.1×10^{-5}	1.0×10^{-6}	5.2×10^{-6}
Emissions to Soil					
As	g	5.0×10^{-8}	3.1×10^{-8}	9.0×10^{-10}	5.8×10^{-9}
Cd	g	2.5×10^{-9}	1.7×10^{-9}	1.0×10^{-10}	7.0×10^{-10}
Cr	g	6.2×10^{-7}	3.9×10^{-7}	1.1×10^{-8}	7.2×10^{-8}
Cu	g	1.4×10^{-8}	8.6×10^{-9}	2.0×10^{-10}	1.2×10^{-9}
Hg	g	3.9×10^{-10}	2.4×10^{-10}	6.0×10^{-12}	3.7×10^{-11}
Ni	g	2.1×10^{-8}	1.3×10^{-8}	3.0×10^{-10}	1.9×10^{-9}
Pb	g	6.4×10^{-8}	3.9×10^{-8}	9.0×10^{-10}	5.7×10^{-9}
Zn	g	2.0×10^{-6}	1.3×10^{-6}	3.0×10^{-8}	2.0×10^{-7}
Resource Extraction					
Coal	g	0.78	1.17	0.07	0.55
Natural gas	dm^3	2.14	3.28	0.14	1.31
Copper ore	g	3.0×10^{-4}	2.2×10^{-4}	1.8×10^{-5}	1.0×10^{-4}
Lead ore	G	3.5×10^{-4}	3.6×10^{-4}	5.5×10^{-5}	4.1×10^{-4}
Crude oil	G	2.74	9.87	0.36	6.37

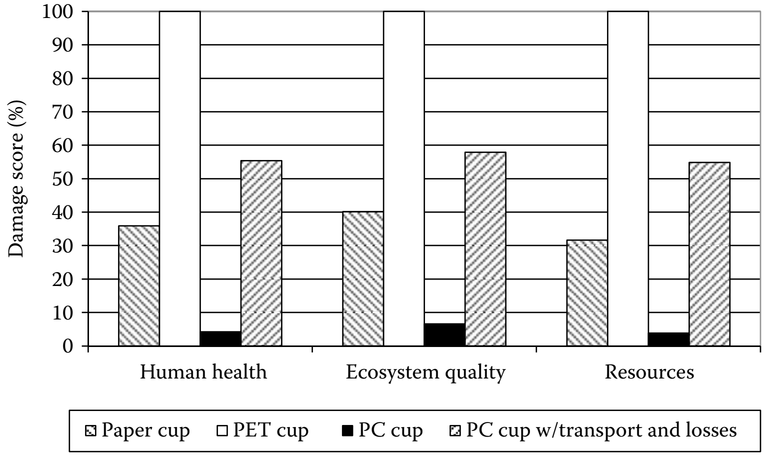


FIGURE 2.4 Proportional damage scores for the different cups, relative to the PET (polyethylene) cup. "PC cup" refers to the multiuse polycarbonate cup. Damage calculated using Eco-indicator 99.

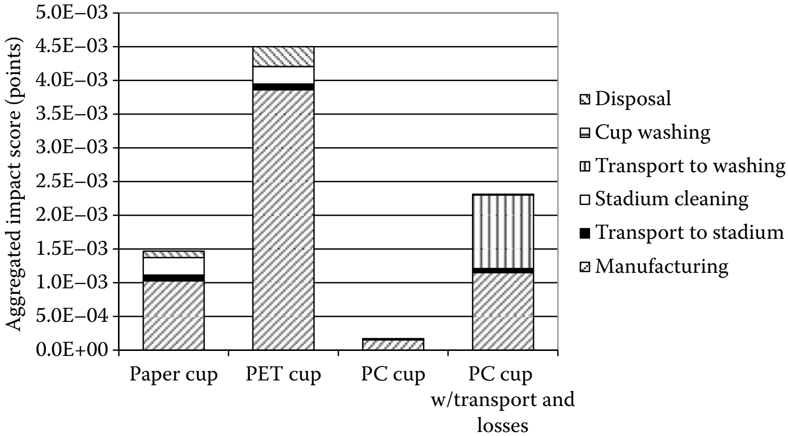


FIGURE 2.5 Total aggregated environmental impacts of the different cup scenarios and distribution by life cycle stage. The Eco-indicator 99 method was used to compare total impacts of a paper cup, a single-use polyethylene (PET) cup, and multiuse polycarbonate (PC) cups.

For all cups, the manufacturing stage (including raw material extraction) accounts for most of the total impact. The impacts of transportation to the stadium and cleaning of the stadium are limited. Assuming a 5% loss in the multiuse PC cup greatly increases each cup's manufacturing impact to a level equivalent to that of the single-use paper cup. Moreover, a 50 km round-trip between the stadium and the washing facility would almost double the impact of the PC cup with losses. The stadium cleaning stage is small in single-use cup scenarios (and assumed not to occur in the multiuse scenarios). The elimination at the end of the cup's life has a small contribution in all four scenarios.

2.4.4 Interpretation of Cup Case Study

The single-use PET cup is clearly the least advantageous scenario, and a multiuse PC cup with no losses or transportation for washing is clearly the best scenario. But, there is no clear-cut conclusion about which is the better scenario between the paper cup and the more realistic multiuse PC cup that assumes losses and washing transportation. A comparison of the two PC cup scenarios clearly shows the negative environmental impacts of losses and transportation for washing; we find that accounting for these reverses the relative ranking of the PC cup and the paper cup. This is partly because a 5% loss per event reduces the actual number of reuses from 150 to 20, increasing the manufacturing impact per functional unit by more than a factor of seven. Moreover, transportation to the washing facilities leads to considerable impacts. This study assumes a 50 km journey by a fully loaded truck with 1000 cups, but the actual impacts will depend on the type of vehicle and the load; a cup carried by a smaller or only partially loaded car has a bigger impact than a cup carried in a large truck containing 20,000 cups. The large impact contributions of losses and transportation demonstrate the sensitivity of the results to the hypotheses made, and the need to best reflect the actual situation.

Finally, it should be noted that the results of the impacts of washing do not account for the washing agent and should, therefore, be treated with caution. The impact of soap can indeed be significant, particularly on ecosystem quality (eutrophication and ecotoxicity).

2.4.5 Conclusions of Cup Case Study

For a single-use cup, this example finds that a paper cup has less environmental impact than a PET cup. It also shows the value of using multiuse cups if there are negligible losses and transportation needs. In practice, the losses should be assessed and included, since a loss of 5% causes the impacts of multiuse cups to become equivalent to or even more harmful than those of single-use paper cups.

Zooming out to consider such large entertainment events as a whole, it is clearly beneficial to reduce the environmental impacts of cups, but it is even better to act where the impacts are highest. For a sporting event, for example, the impact of the cups is relatively small compared with that of the transportation of people to the location of the event; in fact, the impact of one paper cup is approximately equivalent to the impact of transporting one person by car over only 100 m (Figure 2.6). This means that a 10 km trip is 100 times more harmful to the environment than a paper cup. Based on these results, efforts to reduce the impacts of such an event should focus first on the transportation of people to the stadium; for example, by active promotion of the use of public transport. Materials and waste management should be addressed as a second priority.

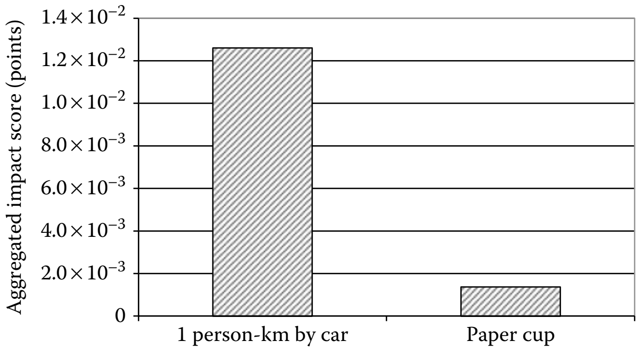


FIGURE 2.6 Comparison between the environmental impact of a single-use paper cup and the transportation of one person by car over 1 km (1 person-km).

EXERCISES

Exercise 2.1: Choose the Best Environmental Evaluation Method and Key Metrics

Decide which assessment method listed in Table 2.1 is most appropriate for the following situations. List key reasons for using this method, and find an appropriate metric/basis for comparison.

1. An electricity company is investing $50 million to integrate photovoltaics into the design of commercial and residential buildings. It wants to estimate the environmental benefits of this design, assuming 1000 buildings will be constructed around the country.
2. An airline company would like to optimize its company's greenhouse gas emissions.
3. You need decide whether to use paper or plastic bags to carry your groceries home.
4. An electricity company is deciding in which of two cities to build its new power plant.
5. You want to decide whether to take the car, bus, train, or airplane from Chicago to New York City based on environmental impacts.
6. A chemical leak occurs in a manufacturing plant, and it needs to decide whether or not to evacuate people from the area.
7. Afterward, the manufacturing plant in (6) must determine the best decontamination method for the site where this leak occurs.
8. Regional authorities are considering creating a recycling auction for old materials and want to decide which materials to include.
9. Congress wants to examine the impacts of using biofuels in the federal fleet of vehicles.

Exercise 2.2: Comparing Cups for a Stadium Event

Based on the information and example provided in Section 2.4, answer the following questions.

1. List two preliminary conclusions you can make based solely on the numbers in the inventory on Table 2.2. List two benefits of subsequently applying impact assessment to this inventory.
2. What are the key parameters affecting the environmental impact of the multiuse PC cup?
3. What is the approximate total aggregated impact score (in points) of a multiuse PC cup, still assuming 5% loss, but assuming that the washing facility is right next to the stadium (use Figure 2.5 for help)?
4. Which result surprised you most about the cup case study and why?
5. In performing an environmental assessment of a sports game, list two other factors to consider (and provide reasoning) (other than cup usage and transportation of spectators to the game).
6. Provide a functional unit that would enable you to compare the relative impacts of a spectator drinking from a cup at the game and the transportation of a spectator to the game.

3 Goal and System Definition

Olivier Jolliet, Gabrielle Soucy, Shanna Shaked, Myriam Saadé-Sbeih, and Pierre Crettaz

The goal and system definition is the first phase of a life cycle assessment (LCA). It may seem trivial, but the LCA results are often strongly dependent on the choices made in this key phase. In the International Organization for Standardization (ISO) norm 14040 (described in Chapter 1), this phase is referred to as the *goal and scope definition*. In this book, we refer to it as the *goal and system definition* to highlight to the reader the importance of clearly delineating and describing a system. This phase consists of, firstly, describing the objectives—what is the purpose of the LCA, what are the results going to be used for, who is the audience, and who are the stakeholders? Secondly, the function of the system considered is analyzed to define a unit that represents this function. Different scenarios are described to achieve this functional unit (FU). Finally, the system boundaries are specified.

3.1 OBJECTIVES

The goal and system definition begins with a description of the study objectives, which determines the problem and defines the intended application for the LCA results, including the intended audience, the stakeholders, and the scope of the study. In contrast to the subsequent more technical stages in an LCA, this step is more descriptive. Moreover, it requires discussion of all options and possible alternatives among the different stakeholders in order to increase credibility and ensure relevant results.

3.1.1 Goal: Type of Application, Intended Audience, and Stakeholders

According to ISO 14044 (section 4.2.2),

> in defining the goal of an LCA, the following items shall be unambiguously stated:
>
> - The intended application,
> - The reasons for carrying out the study,
> - The intended audience i.e., to whom the results of the study are intended to be communicated, and
> - Whether the results are intended to be used in comparative assertions intended to be disclosed to the public.

The intent of a given LCA should be clearly specified to avoid ambiguity among the potential applications and audiences, as demonstrated by the following examples.

- *Information on an existing product:* LCA has often been used to provide information on the environmental impacts of products, most commonly as a comparison of available alternatives.
- *Development of a new product:* In developing a new product, an LCA can first be conducted on the existing or initial product prototype. Improvement options are then selected and evaluated based on environmental, technical, or financial factors. The different production variants are compared against one another and the initial product.
- *Elaboration of political strategies:* Because of its potential for broad application, an LCA can also be used to compare different political strategies. For example, to provide input on future European agricultural policy, a comparison of the environmental impacts of intensive, integrated, and biological production systems would be relevant. ADEME (France's agency for environment and energy management) has published several state-of-the-art reviews on LCAs of waste treatment, biomass use, and agricultural systems (BIO Intelligence Service S.A. 2002; Houillon et al. 2004).
- *Regulation of an existing product:* LCA can also be used to evaluate a product to provide information for regulatory purposes. It is, however, primarily intended as a voluntary tool rather than a compliance-oriented tool.

Even if all audiences are interested in decreasing environmental impacts, consumers, producers, and governments each have different perspectives on how to do so. Consumers want to know product impacts to make wise purchasing choices. Manufacturers generally want to know how to reduce the pollution caused by their products or, less frequently, to highlight their environmental advantages. Governments need reliable information to refine environmental policies or to devise incentives to promote environmental behaviors.

The identity and addresses of the main stakeholders should also be provided, including the sponsors, authors, advisory board, analysts, and optional independent peer reviewers (as addressed in Section 6.4.2). If the intended audience is external, the credibility of the LCA is increased by having the LCA commissioners, analysts, and peer reviewers all be independent entities.

An external review is generally optional, but becomes necessary for an ISO-compatible study involving a comparative assertion. Section 6.4 further addresses the peer review process, which needs to be planned and budgeted from the start of the study.

3.1.2 Scope

Once the goal is determined, the scope of an LCA must take into account and clearly describe the following elements (ISO 14044, Section 4.2.3.1):

- The product system to be studied (in this work, we refer to this as the *system* rather than the *product system* term used by ISO, because the system can also be used to analyze services)
- The function of the system, or of the systems in the case of comparative studies

- The functional unit (FU)
- The system boundary
- Allocation procedures (Chapter 4)
- Life cycle impact assessment (LCIA) methodology and types of impacts (Chapter 5)
- Interpretation to be used (Chapter 6)
- Data requirements
- Assumptions
- Value choices and optional elements
- Limitations
- Data quality requirements
- Type of critical review, if any
- Type and format of the report required for the study

The scope of the study must be sufficiently well defined to ensure that the breadth, depth, and level of detail match the set objective. LCA is an iterative approach (Section 2.2); thus, the scope may be adjusted based on information collected during the analysis.

In the following sections, we describe and provide examples of the most fundamental elements of the goal and scope definition: a description of the system function, a definition of the FU (and associated reference flows), and, finally, how the system is defined and bounded. The remaining key elements, such as the allocation of impacts among coproducts and by-products and the selection of LCIA methods, are described in more detail in Chapters 4 and 5.

When considering the scope of an LCA, we can broadly distinguish between two types of modeling approaches, attributional and consequential LCA, as defined by Finnveden et al. (2009, p. 3): "Attributional LCA is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems. Consequential LCA is defined by its aim to describe how environmentally relevant flows will change to possible decisions (Curran et al. 2005)." A typical example of the difference between these two approaches is the choice of the electricity mix to model the electricity production in a process LCA. An attributional LCA would typically consider the average electricity mix in the considered region of interdependent electricity distribution, such as the average European electricity mix, with the risk that a given decision would affect only one mode of production (e.g., natural gas). A consequential LCA would aim at first identifying and using the marginal mode of electricity production in a growing market (e.g., gas power plants could be built in Europe if electricity demand increases) or in a shrinking market (e.g., coal power plants could be shut down if demand decreases), with the risk that the marginal technology could be misidentified (e.g., when using a general equilibrium model in a rapidly changing world).

As discussed by Zamagni et al. (2012) and Suh and Yang (2014), both approaches have various strengths and weaknesses and still tend to overlap in LCA applications. In this book, we will help the reader to pragmatically consider the question raised by Suh and Yang (2014, p. 1183): "How can a model, or a combination of models, best be used to answer a question recognizing both strengths and weaknesses of different modeling frameworks and available data?"

3.2 SYSTEM FUNCTION

Once the goal of a comparative study is defined, various systems or products must be compared based on a common function. *Scenarios*, which represent the different alternatives, are chosen to satisfy the same function. This *system function* needs a clear definition, because it is the basis for determining two essential LCA elements: the *functional unit* (Section 3.3) and the *system boundaries* (Sections 3.4 and 3.5). To most objectively consider each alternative, the function should be determined before defining the FU or the system boundaries.

It is not always easy to select one exact function of a system, because a single product can have multiple functions. In such a case, the main function and the secondary functions should be identified (see examples in Table 3.1). The primary function, by definition, is common to the different alternatives. The secondary functions are specific to each scenario, and if they differ greatly between alternatives, they can reveal bias in the comparison. For example, one might try to compare a pair of boots with a pair of sandals, because they both meet a primary function of protecting the feet. However, the boots additionally protect the feet from cold; thus, a direct comparison of the two types of shoes is not generally useful.

When a component of a larger system is studied, the function chosen is generally that of the whole system. For example, although the main function of a car door is to provide access to the vehicle and passenger protection, it only becomes effective when the vehicle is used to transport passengers. Thus, a more meaningful function is to provide access and protection for the transportation of passengers over the car's lifespan. In this way, the LCA would consider not only the impacts related to door manufacturing, but also the influence of the weight of the car door on subsequent fuel consumption during use.

> When reviewing an LCA, the system function must be checked for validity in all scenarios, making explicit any differences between scenarios. Particular attention should be given when secondary functions of alternative scenarios differ.

TABLE 3.1
Primary and Secondary Functions of Some Sample Products

Product	Primary Function	Secondary Functions
Pair of shoes	Protect feet	Protection from weather
		Protection from cold
		Social stature
Restaurant	Serve meals	Socializing
		Heated space
Car door	Help to ensure safe use of the car	Protection from theft
		Safety in case of accident
		Sealing the car shut
Potato	Food for humans, animal feed, or raw material (starch)	Maintenance of arable land
		Protection of the landscape and environment

3.3 FUNCTIONAL UNIT AND REFERENCE FLOW

3.3.1 Definitions

Once the system function is determined, the functional unit (FU) can be defined.

> According to ISO 14044 (2006), the FU is the "quantified performance of a product system for use as a reference unit." This measure quantifies the function of a system in terms of the service offered. The FU is the same for all scenarios, with inventory flows and impacts for each scenario calculated per FU (Chapters 4 and 5). For example, different transportation methods are often compared based on a FU of transporting one person over a distance of 1 km (i.e., 1 person-km).

The FU is *not* a ratio and must be *quantifiable* and *additive*, such that the impact of two FUs is double that of one FU. The FU for assessing one component of a larger system is based on the FU of the system as a whole, analogous to the choice of a system function described in the previous section. Thus, the FU for a car door can be one car door ensuring access and safety during the transportation of a person over 1 km.

For a given FU, the *reference flows* are the amounts of goods or services purchased to fulfill the function and generate this FU. In the case of transporting 1 person-km, key reference flows include (a) the gasoline used for 1 vehicle-km divided by the number of passengers and (b) the fraction of a vehicle needed to transport one person over 1 km (i.e., one divided by the number of person-kilometers transported over the vehicle's lifespan). These reference flows usually vary among scenarios, and are discussed in greater detail in Section 3.4.1.

Table 3.2 presents examples of FUs and reference flows. In the shoe example, the FU is chosen to be one pair of shoes in good condition for 1 year (the time period is arbitrary, but must remain consistent). Thus, in the case of high-quality shoes that last 2 years, only half a pair of shoes needs to be purchased per year to satisfy this FU (i.e., one pair every 2 years). For lower-quality shoes with a life span of six months, two pairs of shoes must be purchased to be able to wear a suitable pair of shoes for 1 year. In this example, the effective lifetime of the shoes has a direct effect on the reference flows.

In the case of hand-drying with a paper towel or an electric hand-dryer, the FU selected is one pair of dried hands. Since doubling this number simply doubles all corresponding reference flows in all scenarios, the chosen number of pairs of hands is arbitrary but must be consistent among scenarios and explicitly defined. In these two scenarios, the main reference flow needed to dry a pair of hands is either an average of 1.5 paper towels or a consumption of 30 s of electric hand-drying (at a power of 1800 W, this consumes 15 Wh of electricity). The reference flows must also include a fraction of the towel dispenser and of the hand-drying devices. Assuming the paper towel dispenser is operational for about 10 years, with 50 uses per day, it dries about 182,625 hands; thus, 1/182,625 of a holder is required for one pair of

TABLE 3.2
Sample Functional Units and Reference Flows for Various Scenarios

System	Functional Unit (Service Offered)	Reference Flows (What is Purchased)	Key Parameters (Linking Reference Flows to FU)
Pair of shoes	1 pair of functional shoes for 1 year	0.5 pair of high-quality shoes (2 year lifespan) 2 pairs of low-quality shoes (6 month lifespan)	Lifetime of shoe
Hand-dryer	1 pair of dried hands	1.5 paper towels 1/182,625th paper towel dispenser 1800 W for 30 s 1/365,250th electric dryer	Number of towels per usage Power of dryer and duration of use
Wall paint	100 m^2 of wall painted for 20 year	30 kg of long-lasting paint that lasts 20 year 2 × 25 kg of less durable paint that lasts 10 year	Amount of paint applied per square meter Lifetime of paint

dried hands. Assuming the electric hand-dryer is operational for 20 years, with the same frequency of use, 1/365,250 of a dryer is required per pair of hands.

The parameters relating reference flows to the FU can often be identified as *key parameters* directly affecting environmental impacts (Table 3.2), such as the number of towels per usage in the hand-drying example. In the electric-dryer scenario, the amount of electricity per usage is key, expressed in more tangible parameters as the power use and duration of use. For such products where the use dominates impacts, the product efficiency plays a dominant role. In applications where the manufacturing or disposal stages have the dominant impact, such as for a shoe, which requires no energy during use, the product *lifetime* and *amount of material used* often play an essential role, as does the *number of uses*. Doubling the lifetime of such a product cuts its emissions almost in half.

Key parameters often measure environmental performance as ratios of material needed per function, whereas the FU itself is additive and not a ratio (when the FU doubles, so do the impacts).

> To summarize, a meaningful comparison among systems or scenarios must all be based on the same function characterized by the same (FU). The *reference flows* for each scenario represent the amounts of goods or services purchased per FU and constitute the basis for establishing the environmental inventory.

3.3.2 Critical Choice of a Functional Unit: Popcorn as a Packaging Material

To show the importance of choosing the right FU, we describe a study comparing various packing materials (Jolliet et al. 1994). Plastic packaging materials (such as polystyrene peanuts) are traditionally used for economic and technological reasons, despite being nonbiodegradable and produced from nonrenewable raw materials (oil). Various renewable and biodegradable materials have been proposed as alternatives, including popcorn, an "all-natural" option that feels greener. Even if the ethical issue of using food as industrial raw material is still controversial, is popcorn preferable over polystyrene from an environmental perspective? Answering such questions is the role of LCA.

Figure 3.1a compares the environmental impacts of polystyrene and popcorn by mass, showing that polystyrene results in three to four times more impacts per

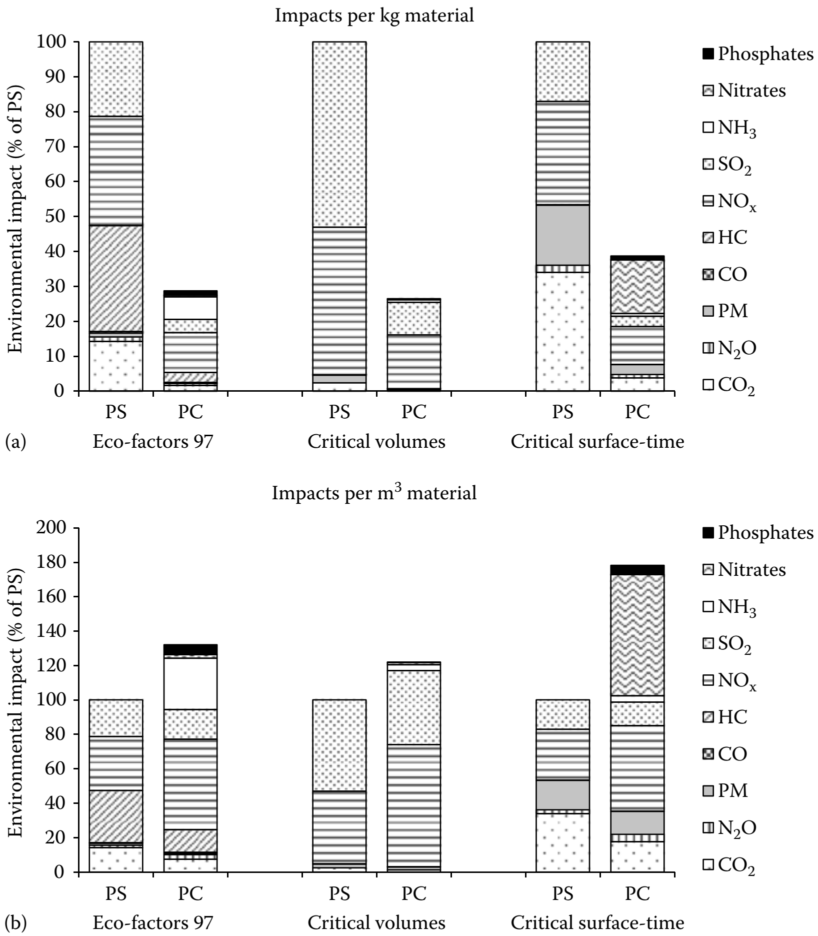


FIGURE 3.1 Comparison of total environmental impacts due to filling a package with polystyrene peanuts (PS) or popcorn (PC) (a) per mass of material and (b) per volume. Impacts of pollutant emissions were characterized and weighted based on three impact assessment methods (as discussed in Chapter 5), where the lower impacts correspond to the more environmentally friendly option. Representation (b) is relevant to the function of filling a package. Substance abbreviations are defined at the front of the book. (Adapted from Jolliet, O., et al., *Agriculture, Ecosystems and Environment*, 49, 253–266, 1994.)

kilogram than popcorn (relative impacts vary slightly depending on the impact assessment method). This analysis may suggest that popcorn is the more environmentally friendly alternative.

However, mass is not the relevant quantity for the function at hand, which is the filling of cardboard to ensure safe transit. Thus, a suitable FU is the filling of one unit of volume, with reference flows corresponding to the mass needed to achieve this. Since popcorn is 4.6 times denser than polystyrene peanuts, the resulting comparative impacts per FU of volume (Figure 3.1b) are reversed relative to the impacts per mass.

The key parameter determining the differences in environmental impacts between polystyrene and popcorn is not primarily the nitrate leaching or fertilizer use from growing the corn, but rather the density of the popcorn. Reducing nitrate leaching and fertilizer use can slightly reduce the impacts of popcorn, but a reduction in density is necessary to make any substantial improvement over polystyrene packaging. This leads to what some may consider a surprising result: the industrial non-natural product (polystyrene) is more environmentally friendly. The concept of "natural" is not necessarily consistent with that of "environmentally friendly"! This density reduction is now achieved industrially by extracting the corn starch and expanding it in a manner similar to that for polystyrene or by using plastic bags filled with air.

Another common key factor is the reuse of materials. For example, assuming that the popcorn is used once on average, compared with the polystyrene peanut, which is used twice, the polystyrene emissions and subsequent impacts should be halved for each FU (i.e., only half the impacts are allocated to each use). This would further enhance the advantage in this study of using polystyrene.

3.3.3 Electric Light Bulbs: Setting Up the Life Cycle Assessment

The comparison of two light bulbs illustrates how to identify the FU, the reference flows, and the key environmental parameters (Table 3.3), along with how this can inform product design from the beginning (Section 7.3.2). This example will be referred to throughout the book.

The main function of most light bulbs is, of course, to illuminate; thus, the FU must account for a reference amount of visible light, or luminous flux, chosen here to be 800 lm. The FU must also specify a reference duration of illumination, chosen here to be 5000 h. The secondary functions, such as heating, creating a pleasant atmosphere, and being fashionable, can vary according to the bulb type, but are not addressed by the FU and could be considered as complementary performances. The reference flows purchased to fulfill the FU of 800 lm over 5000 h include one fluorescent bulb or five incandescent bulbs (as the latter's service life is only 1000 h). This adds up to 5×35 g = 175 g of materials for the incandescent light bulbs and 160 g (one bulb) of materials for the fluorescent bulb. Electricity consumption must also be purchased, with 60 W $\times$ 5000 h = 300 kWh for the incandescent bulbs, and 13 W $\times$ 5000 h = 65 kWh for the fluorescent one. The key environmental parameters can now be identified as ratios that relate the FU to the reference flows. For the use stage, the key environmental parameter is the luminous efficacy, which is how efficiently the bulb produces visible light, expressed in lumens per watt (LPW). The

TABLE 3.3
Function and Functional Unit for Different Light Bulbs

Products	Main Function	Secondary Functions	
Scenario 1: Incandescent	Illumination	Heat	
		Security	
Scenario 2: Fluorescent		Ambience/style	
Product or system	**Functional unit (service offered)**	**Reference flow (what is bought)**	**Key environmental parameters (linking reference flows to functional unit)**
Scenario 1: Incandescent bulbs	800 lumens for a duration of 5000 h	60 W × 5000 h/FU = 300 kWh/FU 5 bulbs (=5 × 35 g = 175 g)	Lumens per watt (use stage) Bulb lifespan (manufacturing stage)
Scenario 2: Fluorescent bulbs		13 W × 5000 h/FU = 65 kWh/FU 1 bulb (160 g)	Amount and type of materials per bulb (material extraction and manufacturing stages)

key parameters of bulb service life and materials are more relevant to the manufacturing stage. Of course, designers generally aim to maximize luminous efficacy and lifespan while minimizing materials, but outlining the FU and reference flows helps to quantify and prioritize the various parameters and may also identify less obvious parameters. The subsequent LCA stages more clearly quantify the relative importance of these key parameters, as well as the impacts caused by toxic substance emissions. Section 4.2.2 presents the next step of detailing energy and CO_2 balances for this case study.

3.3.4 Functional Unit and Reference Flows: A Common Basis for Both Environmental and Cost Analyses

In addition to estimating the environmental impacts of existing products and providing information to guide the design of new products, the FU and reference flows can also be used to estimate costs over the whole life cycle of the product. This process of life cycle costing (LCC) is not technically part of the environmentally focused LCA, but since it uses life cycle thinking and the same framework and concepts, the two are easily combined. A short LCC example is given in the next subsection, with further details in Chapter 6.

3.3.4.1 Electric Light Bulbs: Life Cycle Costs

A fluorescent light bulb is generally more expensive than an incandescent one at purchase, but a fluorescent bulb lasts longer and saves electricity and thus money. Since the reference flows represent what must be purchased to achieve the FU, it

TABLE 3.4
Analysis of Life Cycle Costs for the Light Bulb Example

Product or System	Functional Unit (Service Offered)	Reference Flow (What Is Bought)	Life Cycle Cost
Scenario 1:		300 kWh/FU	300 kWh/FU × \$0.1/kWh = \$30/FU
Incandescent bulbs	800 lm for a duration of 5000 h	5 bulbs	5 × \$1.5 = \$7.5/FU
			Total = \$37.5/FU
Scenario 2:		65 kWh/FU	65 kWh/FU × \$0.1/kWh = \$6.5/FU
Fluorescent bulbs		1 bulb	1 × \$8 = \$8/FU
			Total = \$14.5/FU

is only logical to use them as a starting point in LCC (Table 3.4 and Section 6.8.1). To provide the service of 800 lm for 5000 h, the five incandescent bulbs cost \$7.50/FU, compared with \$8/FU for one low-energy fluorescent bulb. Even at a competitive rate of \$0.1/kWh, the cost of electricity during the use stage comes to \$30/FU for the incandescent bulbs and \$6.5/FU for the fluorescent bulbs. Thus, despite their higher retail price, fluorescent bulbs generate significant savings of approximately \$23 per bulb over the course of 5000 h or 5 years.

The results of these environmental and cost comparisons are thus ideal, where the fluorescent bulbs are better from both an energetic and economic point of view. Because of this, the sale of incandescent bulbs has recently been phased out in many countries. Why, then, do people continue to purchase incandescent bulbs that consume more energy and end up costing more than fluorescent bulbs? Incandescent bulbs are not fully socially accepted for a variety of reasons. First, a higher initial cost is a barrier, emphasizing the importance of information on future benefits when purchasing the product. Also, fluorescent bulbs can take up to a minute to reach full brightness after being turned on, which can be partially compensated by purchasing a bulb with higher luminous intensity (i.e., a compact fluorescent lamp [CFL] of 1100 lm at 18 W can replace an incandescent bulb of 800 lm at 60 W) or by purchasing the even more energy-efficient light-emitting diode (LED) bulbs. These behavioral factors illustrate the importance of the social dimension in the following statement by the Food and Agriculture Organization: “sustainable development … is environmentally non-degrading, technically appropriate, economically viable and socially acceptable” (FAO 1995).

> To summarize, the FU and reference flows are a starting point for two complementary tools: environmental LCA and LCC.

3.3.5 Multifunctional Products

When comparing products or services, if their functions are close but not identical, it is important to clearly indicate this and evaluate the potential consequences on the study results.

Functions may differ in the performance of the product or service. When comparing environmental impacts of different modes of transport (rail, road, and air), the time to travel a given distance can vary greatly, yet this travel time cannot be directly included in the FU. Still, as is done with economic performance, this technical performance can be measured and compared with the environmental performance in the final interpretation phase.

In cases where the multifunctional aspect changes the necessary reference flows, the FU can be adjusted to account for this. This adjustment approach is practically identical to that used for allocating emissions to coproducts, using the same hierarchy of methods applied in Section 4.5.

There are some systems which simultaneously achieve many different functions, in which case the function chosen for analysis is important to identify. For example, the system of wheat crop production (Charles et al. 2006) can have many functions: landscape upkeep, basic wheat production, or production of quality wheat for bread making. For landscape upkeep, the FU is 1 ha, with environmental impacts reported per unit of surface area. For wheat production as the function, the FU is 1 t of produced grains. Bread making implies not only a given quantity of flour but also a minimal protein content of 13%; thus, the FU is 1 t of grains with a 13% protein content.

Different choices of system function can yield different environmental impact rankings between scenarios. Figure 3.2 shows that choice of system function can actually reverse the ranking of nonrenewable primary energy consumption in wheat production systems for different fertilization levels. For example, the low fertilization intensity option is energetically favorable for landscape upkeep, but it is the most energy-intensive option when growing bread-quality wheat. This is because low fertilization reduces both yield and protein content, necessitating the use of a

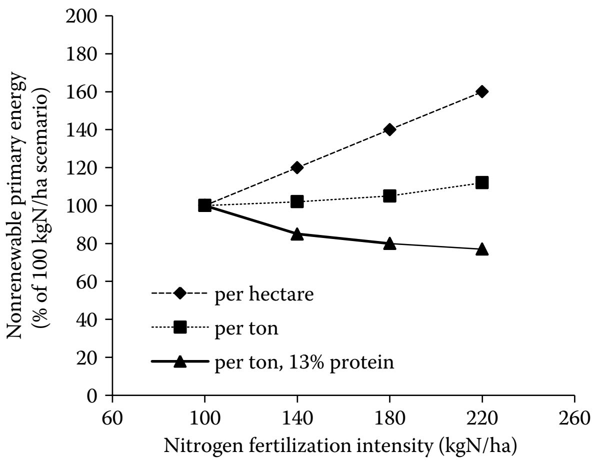


FIGURE 3.2 Consumption of nonrenewable primary energy necessary for wheat production as a function of nitrogen fertilization intensities for three different functional units: per surface area (hectare), per mass of wheat (ton), and per mass of bread-quality wheat (ton, 13% protein). (Adapted from Charles, R., et al. *Agriculture Ecosystems & Environment*, 113, 216–225, 2006.)

high-protein wheat variety that leads to a further reduction in yield and thus higher impacts per unit mass of bread with a 13% protein content.

3.4 SYSTEM DEFINITION

3.4.1 Principles of System Modeling

The reference flows and subsequent impacts for each FU are calculated based on a well-defined system. System modeling is based on a holistic approach that provides a global understanding of the system by considering it as a whole, in all of its dynamics and complexity (Le Moigne 1990). The system is more than the sum of its elements. The system modeling approach focuses on the relationships between elements that make up the system rather than the elements themselves. The system is then described in terms of these relationships and their significance to the function of the system. In LCA, the world can be schematically decomposed into the environment, the system providing a product or function, and the rest of the economic activity (Figure 3.3).

The system here is defined as a group of dynamically interacting elements, organized to achieve one or more functions. It is identified by the elements it contains, called *processes*, the links between these elements, and the boundaries that delineate it from the surroundings (environment plus economy). The inputs of the environment into the system are the *extracted resources*, which include the energy and the land used; the outputs of the system into the environment are *emissions* to air, water, and soil. The output of the system into the economic world is the service provided by the product.

The assessed and modeled system is built by linking different process modules. The processes and elements required to fulfill the function are identified, and these are expressed as a series of *unit processes* (Figure 3.4), the smallest elements in the analysis, for each of which inputs and outputs are quantified. Unit processes are linked to one another within the system by *intermediary flows*, expressing the quantity of each unit process needed for the subsequent unit process. The outputted *product flows* to the economy are any products that leave the system. Unit processes are linked to the environment by *elementary flows*, with input elementary flows corresponding to the use of natural resources, such as extracted raw material, energy, and land use. Elementary flows exiting a unit process are emissions to water, air, or soil.

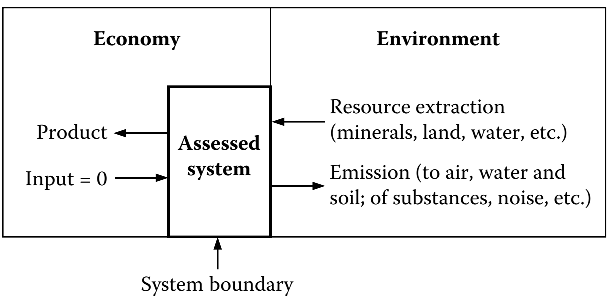


FIGURE 3.3 Relationships and exchanges between the studied system, the economic world, and the environment.

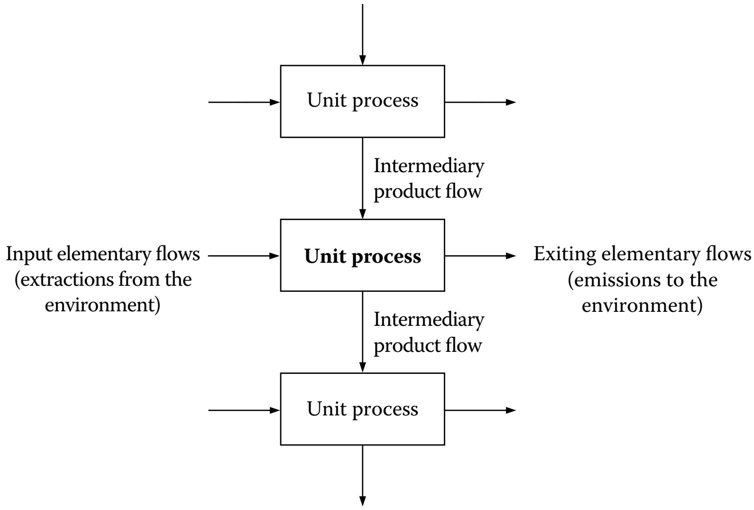


FIGURE 3.4 Example of a set of unit processes in a system. (After ISO, *ISO 14040 Environmental Management—Life Cycle Assessment—Principles and Framework,* 2006.)

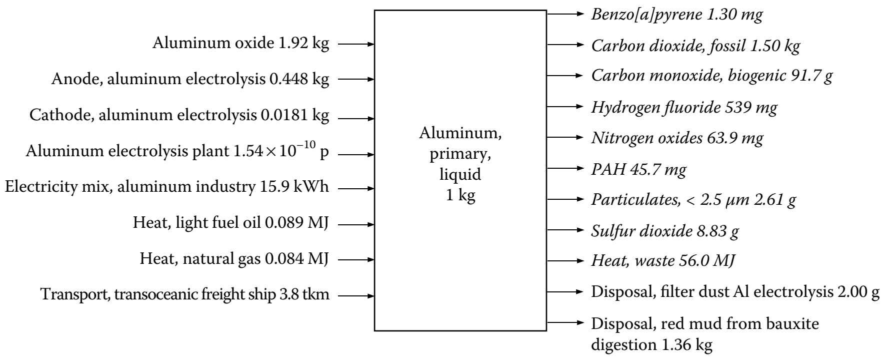


FIGURE 3.5 Unit process flows associated with liquid primary aluminum at plant, ecoinvent 2.2. Elementary flows from and to the environment are shown in italics.

Figure 3.5 presents the intermediary and elementary flows of primary liquid aluminum, as described in ecoinvent 2.2. The manufacturing of liquid aluminum makes use of the following intermediary flows: aluminum oxide, electricity and quantities of anode and cathode for electrolysis, and heat produced by burning light fuel oil and natural gas. It also involves transoceanic freight and treatment of wastes from aluminum production. Direct air emissions during liquid aluminum manufacturing results in some of the following elementary flows: benzo[*a*]pyrene and other polycyclic aromatic hydrocarbon emissions, carbon dioxide and chlorofluorocarbon-14 (CFC-14), sulfur dioxide, nitrogen oxides, and $PM_{2.5}$, as well as waste heat. Each of the intermediary flows requires other intermediary flows and generates

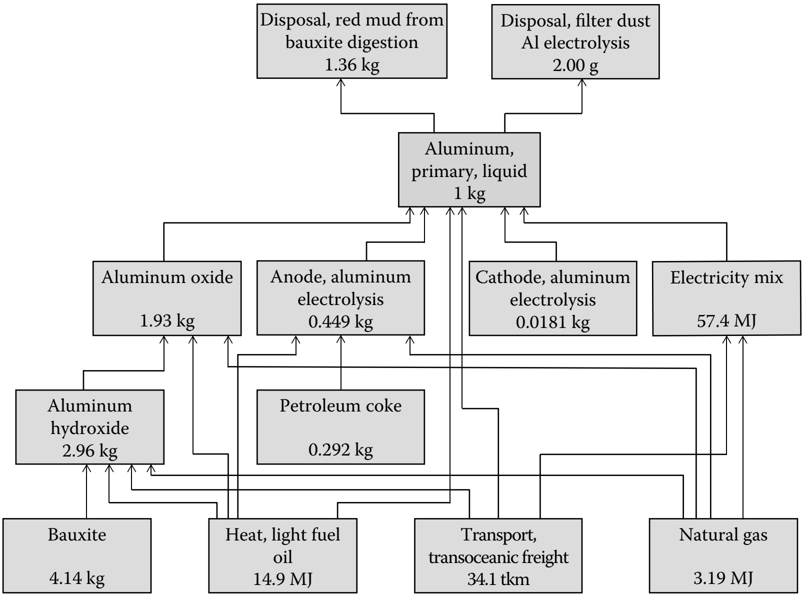


FIGURE 3.6 Flowchart for the manufacturing of liquid primary aluminum at plant, based on ecoinvent 2.2.

additional direct emissions. For example, the manufacturing of aluminum oxide will itself use aluminum hydroxide, as well as additional electricity and heat from light fuel oil and natural gas, and will generate some more waste heat.

The level of detail required in modeling a unit process depends on the objectives of the study. Each unit process can be further subdivided into other unit processes down to the level of necessary detail (Figure 3.6). Since this is a physical system, mass and energy balances can be carried out to check that unit processes and the global system respect conservation of mass and energy (see example in Figure 8.3).

The theoretically ideal system is one defined such that the economic world has no inputs to the system and only one output from the system, namely the product corresponding to the studied function (Figure 3.3). All processes required to fulfill the system function should be part of the system. In practice, this is often not possible, either because of a lack of data or time to carry out the LCA. Moreover, the system may have outputs other than the studied product, resulting in coproducts of value to which a portion of the emissions must be allocated (Section 4.5).

3.4.2 Flowchart

The *flowchart* or *flow diagram* or *process tree* (such as the one depicted in Figure 3.6) provides a clear overview of the processes and their relationships. It depicts each unit process considered within the system and quantifies the intermediary flows linking these unit processes. The flow diagram is built starting from the reference flows (what you need to buy for one FU), then identifying the first-tier intermediary flows (quantities of preceding unit processes) associated with each reference flow. The operation is then repeated, starting from the Tier 1 intermediary flow, yielding a second set of Tier 2 elementary flows. In practice, for a new study, the flow diagram will display all linkages from reference flows up to existing database unit processes, whose upstream and downstream links are described in further detail in the database itself and therefore do not necessarily need to be shown in the flowchart. Figure 3.6 presents the flowchart for liquid aluminum production. With the goal of displaying all key unit processes, the system boundaries also include the upstream Tier 2 processes, such as aluminum hydroxide.

3.4.3 Description of Scenarios

Each scenario being compared in the LCA must have its own flowchart to be properly visualized and broken down into unit modules. Each scenario must cover the same functional reality and yield the same FU, which means having the same primary function. Of course, scenarios may share certain unit processes, as described in a case study in Chapter 8 (Figure 8.1).

3.5 SYSTEM BOUNDARIES

3.5.1 Principles of System Boundaries

The system boundaries determine which specific modules are included and excluded when modeling the system. They are defined to ideally include all the required processes, from cradle to grave, to fulfill the function. This may sound simple, but quickly becomes complicated in actual applications.

A complete LCA would require coverage and modeling of all global production processes occurring at any point in the production, use, or disposal chain. In agricultural production, for example, an LCA should account for emissions from the use of a tractor, but is it necessary to consider the emissions associated with manufacturing the tractor, or associated with making the machinery that helped to manufacture the tractor? For the LCAs described thus far, based on sets of processes, it is important to determine the criteria for the inclusion or exclusion of certain processes and apply them according to ISO 14000 standards. This section first describes the main types of processes to consider, and then presents an example to describe how consistency rules are defined to best determine system boundaries.

Note that the input–output method of LCA avoids many of these issues associated with cutting off the supply chain, with advantages and disadvantages discussed further in Section 4.4.4.

> According to ISO 14044, the system boundary is the "set of criteria specifying which unit processes are part of a product system" (p. 5).

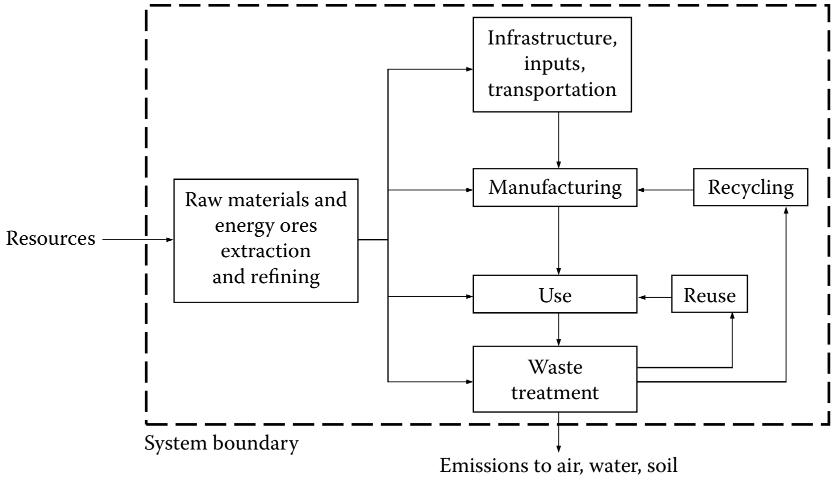


FIGURE 3.7 Flowchart of major processes and stages in the life cycle of a product.

3.5.2 Main Considered Processes

Starting from the reference flows, the system should include all flows required to fulfill the function and cover the main life cycle stages (ISO 14040, p. 12 and Figure 3.7):

- Extraction and refining of raw materials and energy
- Provision of infrastructure, machinery, inputs, and transport
- Main manufacturing stage
- Use stage, including maintenance
- Waste treatment, taking into account the recovery of used products (including reuse, recycling, and energy recovery)

Since eliminating waste generates emissions, waste treatment must be included within the system boundaries. Waste flows (e.g., kilograms of anode waste sent to a landfill) are intermediary flows rather than elementary ones and, therefore, should not be reported in the inventory as such. It is the emissions and other elementary flows associated with the waste treatment stages (e.g., kilograms of aluminum emitted to ground and surface water) that are reported in the inventory.

3.5.3 Importance of System Boundaries: Comparing a Fast-Food and a Traditional Restaurant

An LCA study compared a fast-food restaurant with a traditional one in the early 1990s. The results obtained (Figure 3.8) indicate that the fast-food restaurant consumed six times less energy and seven times less water, and produced five times less waste per customer.

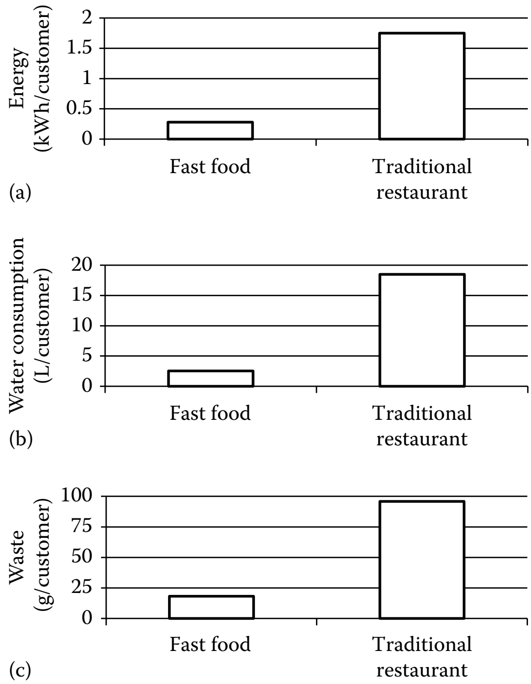


FIGURE 3.8 Initial results comparing the impacts of a fast-food restaurant with a traditional restaurant: (a) energy, (b) water, (c) waste.

Faced with these rather surprising results, the considered system boundaries and other assumptions underlying this study should be closely examined. Table 3.5 lists the main life cycle stages for each scenario, including those that were not considered in the original study. Can you see some of the flaws of this study?

The first flaw with this study is that the system boundaries are chosen as the walls of the restaurant, only accounting for the processes that occur within the restaurant. This is a physical rather than functional basis for a system boundary, and thus excludes initial food preparation and packaging from the fast-food system. Moreover, the dish production and waste management also occur outside the restaurant and are not included for either system, thus preventing a comparison of their differences. These excluded processes are generally unfavorable to fast-food restaurants and therefore falsely bias the study. A second criticism of this work is that the function of a fast-food restaurant is different from that of a traditional one, which offers more space to customers to linger (Table 3.1).

The study described was repeated by Lang et al. (1994), who addressed these key criticisms. To compare two restaurants with the same function, the first fast-food restaurant (FF1), using all disposable dishes, was compared with a restaurant of the same style in which cutlery is washed and reused (FF2). Moreover, the system

TABLE 3.5
Comparison of Food Preparation Steps for Fast-Food and Traditional Restaurants

Fast-Food Restaurant	Traditional Restaurant
Agricultural production chain	*Agricultural production chain (same as fast-food)*
Transport	*Transport (same)*
Production chain for plastic tableware (knives, forks, cups, etc.)	*Production chain for reusable dishes*
Initial preparation and packaging of food (preparation of burgers, salads, etc.)	Preparation of food and cooking
Final cooking	
Cleaning, heating, and lighting of restaurant	Cleaning, heating, and lighting of restaurant (same)
	Clean reusable dishes
Management of packaging and food waste	*Management of food and packaging waste*

Note: Items in italic were not included in the system boundary of the original study.

boundaries were chosen to account for all the processes needed to achieve equivalent functions (Figure 3.9), thus accounting for dishwashing and waste treatment. Finally, the number of meals served at FF1 was recounted and revised to a lower value than in the original study.

For these more consistent system boundaries, Figure 3.10 shows the environmental impact comparison obtained by Lang et al. (1994). Using the Ecoscarcity method to aggregate different impacts (see Section 5.5.5), the environmental load due to FF1 is about twice that of FF2. Therefore, the corrected system boundaries and comparison between similar functions (along with improved data for FF1) reverse the results of the flawed original study, finding that the fast-food restaurant with reusable dishes is more environmentally friendly.

3.5.4 Rules to Define System Boundaries

The example in the previous subsection illustrates the need to identify rules for consistency when defining system boundaries, to provide a framework that helps to reveal and avoid errors.

> *Rule 1* The system boundaries must cover the same functional reality in all scenarios.

The first study, comparing the fast-food with the traditional restaurant, had two system boundaries that clearly did not cover the same reality, including all food preparation and dishwashing for the traditional scenario without including analogous processes for the fast-food one. The system boundaries are not physical boundaries, but functional ones.

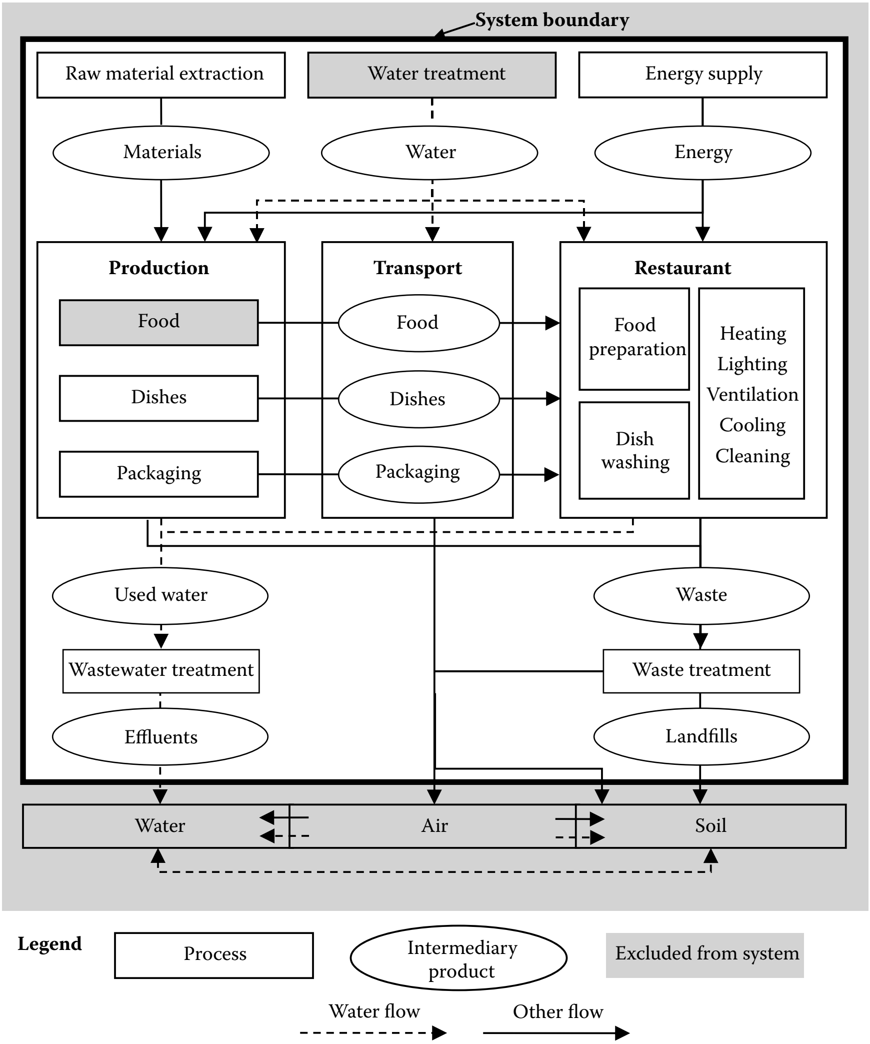


FIGURE 3.9 System boundary to compare environmental impacts of two fast-food restaurants. (After Lang, B., et al., *GAIA*, 3, 108–115, 1994.)

Rule 2 Cut-off criteria for the inclusion of processes in the system boundary must be clearly described, including assumptions, reasoning, and effects on study results. Processes should only be excluded if they contribute less than the cut-off percentages (e.g., 1%) to the mass of the product, the energy consumption of the system, or the environmental impacts (e.g., emissions of a given pollutant) (ISO 14044, 2006). These cut-off criteria are fixed in advance.

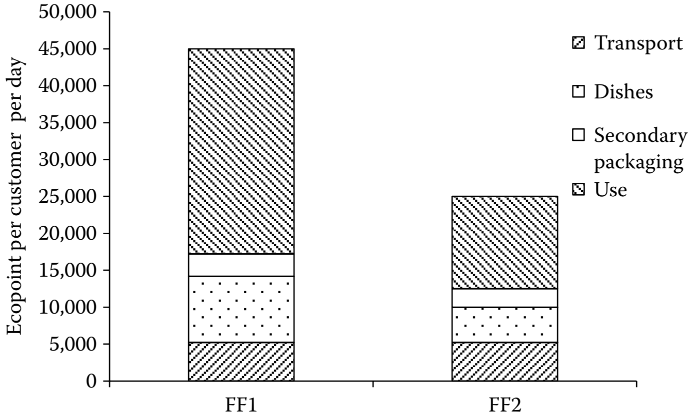


FIGURE 3.10 Comparison of environmental impacts (in ecopoint) per customer served in fast-food restaurants FF1 and FF2. (After Lang, B., et al., *GAIA*, 3, 108–115, 1994.)

This rule is consistent with the iterative process of LCA, in which a preliminary assessment (screening) is conducted over all phases (inventory, impact assessment, and interpretation) to determine the order of magnitude of different contributions. The subsequent detailed analysis further examines the processes that exceed the cut-off criteria. Because of the array of LCA modules and applications, it is not possible to determine an overarching cut-off threshold that applies to all LCAs. In practice, the cut-off criteria are applied as a minimum inclusion requirement, ensuring that all processes contributing to more than the cut-off are included. All existing processes available in databases or collected data on new processes are usually kept, even if they contribute to less than the cut-off.

Rule 3 Processes that are identical in the different scenarios can only be excluded if the reference and intermediary flows affected by these processes are strictly equal (i.e., the intermediary flows of each excluded process per FU are exactly the same for the different scenarios).

In other words, the LCA must still account for identical processes occurring in all scenarios if the scenarios require different quantities of processes.

As an example, imagine comparing the use of two herbicides, A and B, for the harvest of a given quantity of wheat. Clearly, the system boundaries must include the processes of herbicide manufacturing and treatment. But, can we exclude the gasoline required for plowing the field and other wheat cultivation steps that must occur in both scenarios? These steps are identical regardless of the herbicide and thus appear unnecessary in the comparison (Figure 3.11).

To answer this question, it is essential to consider the necessary intermediary flows *per FU* rather than per other default quantities. Per hectare, it is true that the

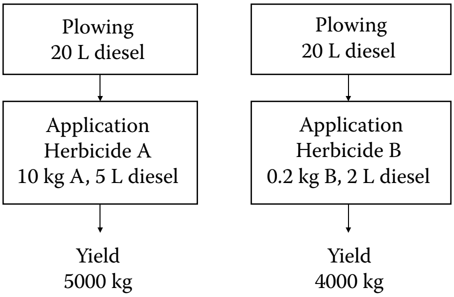


FIGURE 3.11 Treatment of a hectare of wheat with Herbicide A and Herbicide B.

same amount of diesel is used to plow a field of wheat. In these scenarios, however, the yield per hectare is not identical between the scenarios, due to the differences in the effectiveness of the herbicides. Thus, the tractor needs to plow a larger area for Herbicide B to yield a given amount of wheat; the Herbicide B scenario requires 0.005 L of diesel per kilogram of wheat compared with 0.004 L for Herbicide A. In other words, when the results are calculated per FU (per kilogram of wheat), emissions per hectare plowed remain the same, but must be divided by different yields. Since the herbicide affects the ultimate yield, it is not only the herbicide stage but all wheat production stages that must be accounted for.

A second example shows how seemingly identical stages of a product must still be included in the system boundaries because of other product differences. A study compared three kinds of toothpaste packaging (Haydock 1995): a standard tube in cardboard packaging, a self-standing tube not needing the outer box, and a dosing toothpaste pump. Although the type and amount of toothpaste considered in each packaging was identical, the packaging influenced how much toothpaste was ultimately discarded (Figure 3.12). More wasted toothpaste increases the effective

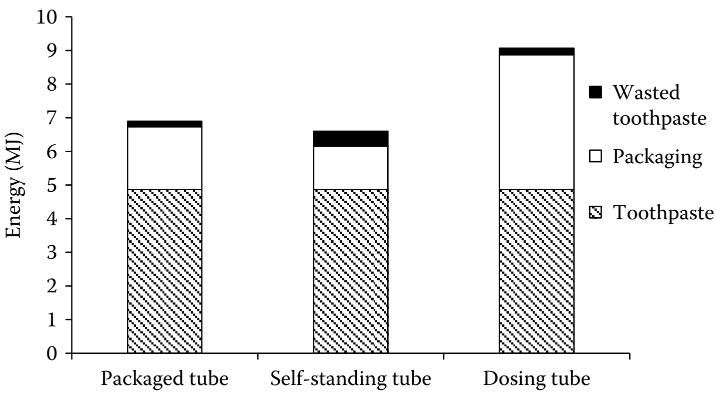


FIGURE 3.12 Total energy requirements associated with the use of toothpaste and its packaging. (From Haydock, R., *SETAC Case Studies Symposium*, 1995.)

energy used and the emissions associated with that packaging. Another effect of packaging (not shown in Figure 3.12) is the influence of the tube opening on the amount of toothpaste used per brushing. Therefore, the production energy and emissions associated with toothpaste manufacturing should still be included in the system boundaries to consider differences in waste, even if it is the packaging that is being compared.

EXERCISES

Exercise 3.1: Paper or Plastic?

You want to decide whether to use a plastic or paper bag at the supermarket. The plastic bag weighs 40 g with a capacity of 9 L, and you estimate you will use it twice on average (reuse one additional time after the first use). The paper bag weighs 30 g with a capacity of 6 L and cannot be reused. Use the forms AIV.2 and AIV.3 in Appendix IV to

a. Determine the FU, assuming the bags carry their maximum load.
b. Determine the reference flows and key parameters linking the reference flows to the FU for the two scenarios, assuming manufacturing and disposal are negligible.

Exercise 3.2: Paper Towels or Hot-Air Dryer?

Use LCA to compare the use of paper towels in a public restroom with that of a hot-air dryer. Many key assumptions and parameter values are provided as bullet points and in Table 3.6. Use the forms IV.2 through IV.4 in Appendix IV to

a. Describe the function and any secondary functions of the system.
b. Choose an FU that represents the function of the system.
c. For each case, list the reference flows and key parameters corresponding to the selected FU.
d. System boundary.

TABLE 3.6
Table of Parameter Values for Exercise Comparing Hand-Drying Options

Hot-Air Hand-Dryer		Paper Towel	
Device carcass	8 kg cast iron	Dispenser weight	3 kg plastic
Devise base	4 kg steel		
Device lifetime	20 year	Dispenser lifetime	10 year
Electricity rate	1800 W	Paper towel weight	3.58 g
Drying time	30 s		
Transport	100 km truck	Transport	100 km truck

i. Starting from the FU and the reference flows, draw the flowchart and system boundary for each scenario; you may want to stop when you link to the process of an existing database.
ii. For each scenario, find a secondary function that may save energy and draw it on the diagram.
iii. Based on the masses of the various reference flows involved, decide whether the transportation of the paper towel dispenser should be included in the system boundary, and explain.

Assume the following:

- Fifty uses per day for both scenarios.
- The hot-air hand-dryer device is made out of cast iron. It is activated by a button that blows hot air for 30 s.
- On average, 1.5 paper towels are used for each pair of hands.
- Manufacturing energy for towels, towel dispenser, and air-dryer device are negligible.

Exercise 3.3: Elementary and Intermediary Flows of 1 kg Aluminum Hydroxide

Based on Figure 3.13, determine the elementary and the intermediary flows related to the production of 1 kg of aluminum hydroxide.

Product or System	**Functional Unit (Service Offered)**	**Reference Flow (What Is Bought)**	**Key Environmental Parameters (Linking Reference Flows to FU)**
Scenario 1: Incandescent bulbs	800 lm for a duration of 5000 h	60 W × 5000 h/FU = 300 kWh/FU 5 bulbs (= 5 × 35 g = 175 g)	Lumens per watt (use stage) Bulb lifespan (manufacturing stage)
Scenario 2: Fluorescent bulbs		13 W × 5000 h/FU = 65 kWh/FU 1 bulb (160 g)	Amount and type of materials per bulb (material extraction and manufacturing stages)

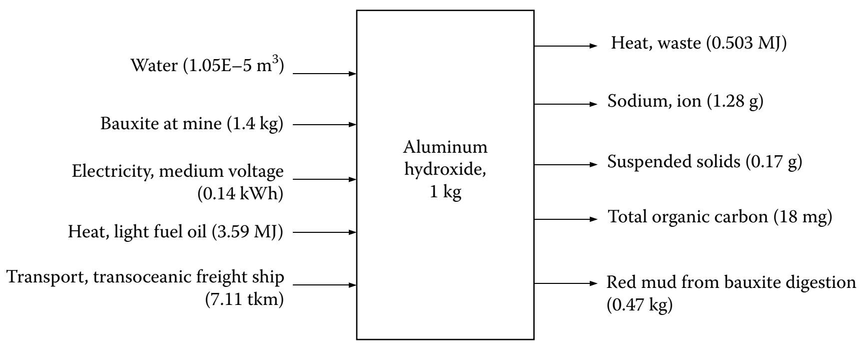


FIGURE 3.13 Unit process flows of aluminum hydroxide at plant, ecoinvent 2.2.

4 Inventory Analysis of Emissions and Extractions

Once the first phase of a life cycle assessment (LCA) has defined the different scenarios, the functions they fill, and the systems to be studied, everything is ready for the second phase of an LCA, which is the focus of this chapter. It is now time to quantify the inventory of the various flows of material extractions and substance emissions crossing the system boundary.

Section 4.1 provides an overview of the essential principles in inventory analysis. Two methods to calculate the inventory currently prevail: the process-based approach and the input–output (I/O) approach. The specificities and complementarities of these two approaches will be explained in the remainder of this chapter.

Section 4.2 thoroughly explains the more commonly used process-based approach, which is based on physical flows. The inventory simply combines the previously calculated reference flows of unit processes in the system with emissions and extractions for each unit process. (Remember from Chapter 3 that the *reference flows* are the amounts of goods or services purchased to fulfill the function and generate a functional unit FU.) A complete inventory generally accounts for hundreds of substances out of the 100,000 possible anthropogenic emissions. This section first focuses on the assessment of energy consumption and CO_2 emissions—two key inventory items—identifying the key points of the process-based approach. It then applies the resulting basic calculations to the simple case study of a car front-end panel to illustrate the inventory procedure.

Although the principles of inventory calculation are relatively simple, the collection of data may require a substantial effort. Fortunately, databases now exist that integrate data for a wide range of processes, leaving only the processes specific to the considered application and industries to be modeled in detail. Section 4.3 introduces the databases commonly used for the process-based approach, describes in further detail the widespread ecoinvent database, and addresses quality issues in databases.

The second way to calculate the inventory is to calculate the emissions and extractions not on the basis of physical flows, but on that of economic flows generated by the product or service concerned. Called *input–output* (I/O), this method is explained in Section 4.4, along with the main databases available. This section ends with a comparison of the process and I/O methods and their possible combinations in a so-called hybrid LCA approach.

Many processes result in more than one product, so Section 4.5 deals with the complex issue of how to allocate emissions and extractions to coproducts and by-products. The final section (Section 4.6) provides exercises to practice the skills taught in this chapter.

4.1 PRINCIPLES OF INVENTORY ANALYSIS

4.1.1 Comparison of Process-Based Inventory with Input/Output Inventory

The first approach to calculate the inventory is called the *process-based approach* and uses physical reference flows and intermediary flows to identify and link the unit processes of a system (Sections 4.2 and 4.3). The second approach, called *input–output*, bases its analysis on the economic flows generated by the product or service considered (Section 4.4).

4.1.2 Definitions

The *inventory of elementary flows or emissions and extractions* is, by definition, the quantitative description of flows of matter, energy, and pollutants that cross the system boundary. This includes the emissions of polluting substances to the environment as well as the amounts of extracted resources from the environment (minerals, energy carriers, soil surface area, etc.) throughout the life cycle of the analyzed product or service.

For the process approach, this inventory is calculated by multiplying the reference flows and corresponding intermediary flows per FU by the direct *emission or extraction factors* of each unit process. The emission and extraction factors, available in various databases (see Section 4.3), provide the quantities of ore extracted and pollutant emissions emitted for each unit process. When possible, we perform the mass balance of each substance in the studied process to verify that the utilized data and calculated elementary flows still conserve mass.

For the I/O approach, the inventory is calculated by using economic data to first relate the direct *demand* for a good or service to the total demand in the entire economy. The inventory of emissions and extractions is then calculated by multiplying the total demand per FU by the emissions per dollar spent in each sector. Section 4.4 explains the principles and potential applications of this approach.

4.1.3 Problem of Aggregation over Time and Space

The processes in the life cycle of a product or service generally occur at various points in time and space. If a product leads to a total emission of 5 kg of SO_2 into air, this may consist of 1 kg emitted in India in 2000, 0.1 kg emitted in Switzerland in 1995, 3 kg emitted in Brazil in 2010, and 0.9 kg emitted on the "world market" (without geographical specification) in 2014. Accurately accounting for the specific time and place of every emission can lead to an overwhelming amount of necessary data and calculations.

To reduce this amount of work, an aggregation is generally performed. The first step is an aggregation over time, assuming that the effect is independent of the time at which it takes place. Another common aggregation for a preliminary analysis is to sum all emissions regardless of physical location. These aggregations are based on the assumption that the impact of a substance is mainly based on its intrinsic properties and its total emission rather than the surrounding temporospatial landscape, which is clearly a simplification. In fact, the same substance emission in different places can result in differences in substance transformation (due to different climates or soils), population exposure (based on proximity of a populated zone), and effective toxicity (based on ecosystem sensitivity). If these factors are considered influential, the subsequent iteration of the analysis should keep track of the time and location of each emission. These issues will be discussed through this chapter.

4.2 PROCESS-BASED CALCULATION OF THE INVENTORY

This section begins with a step-by-step procedure to calculate a process-based inventory. It then describes in detail how to calculate energy and CO_2 inventory results, for which a hand calculation is often useful as a check of software computations. This is extended to the total emissions and extractions associated with a concrete example, where software is then used to limit the chances of error.

4.2.1 Step-by-Step Procedure for Process-Based Inventory Analysis

1. Start with the reference flows (what you actually buy) corresponding to the FU (Table 3.3). Using the intermediary flows of materials and processes associated with each reference flow, design the flowchart of the core unit processes involved in the system, both upstream and downstream from these reference flows. In practice, you may want to stop when you link to the process of an existing database (see Figure 3.6 and Section 3.4.2 for further details).
2. For each unit process, find its inputs (quantified intermediary flows) and direct emissions (elementary flows). These emissions and extractions factors and intermediary flows can be found (a) in databases, (b) by measurements, or (c) by direct contact with companies.
3. Document the data on a flowchart or in a table, describing the source of information used.
4. Calculate emissions of each unit process by multiplying the amount of each unit process per FU by its emission and extraction factors.
5. Calculate the total aggregated emissions and extractions by summing all elementary flows of all unit processes (Figure 4.1).

The International Organization for Standardization (ISO) and the greenhouse gas protocol from the World Resources Institute (WRI) and World Business Council for Social Development (WBCSD) provide more detailed recommendations for some steps. As expressed in an earlier version of the greenhouse gas protocol (ISO 2009), whenever possible, "primary data shall be collected for all foreground processes and significant background processes under the financial control or operational control of the company undertaking the product inventory. For all other processes, data of the highest quality shall be collected." This can be by extrapolation, adapting a similar (but not representative) process to match the considered process, or by simply using a similar process as a proxy.

As suggested by the greenhouse gas protocol (WRI and WBCSD 2011, appendix C), it may be useful to establish a management plan, identify all unit processes, collect screening data for all processes, and refine the data collection for the most

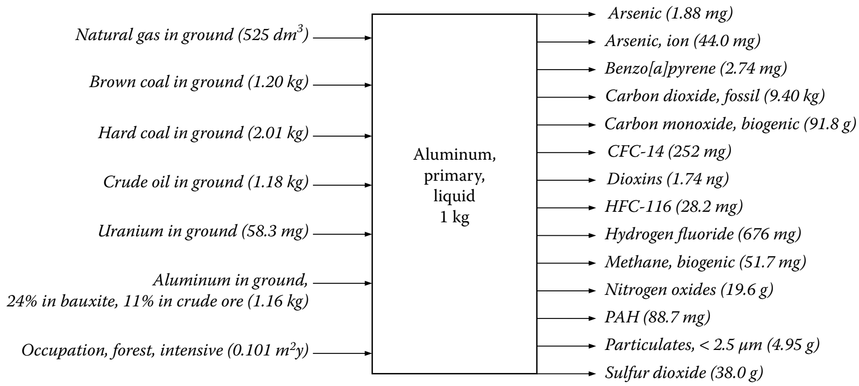


FIGURE 4.1 Aggregated inventory of extractions and emissions for liquid primary aluminum at plant, taken from ecoinvent 2.2. Elementary flows from and to the environment are shown in italics.

contributing processes. Sonnemann and Vigon (2011) provide additional guidance on how to collect data for a given unit process.

To reuse the example of liquid aluminum production (Section 3.4.2), the summary of the aggregated extracted materials and emissions in different environmental media per kilogram of aluminum are detailed in Figure 4.1. Compared with Figure 3.5, which focused only on the direct elementary flows, the aggregated inventory of extractions and emissions include additional flows from upstream and downstream processes. For example, aluminum manufacturing requires bauxite digestion, which leads to arsenic emitted to water during the red mud disposal. For this reason, the 1.5 kg CO_2 of direct emission in Figure 3.5 is supplemented by upstream emissions to yield an aggregated total of 9.4 kg fossil CO_2 emissions in Figure 4.1.

4.2.2 Calculation and Assessment of Energy Consumption and CO_2 Emissions

4.2.2.1 Assessment of Energy Consumption

As a preliminary approach, calculating the demand in *nonrenewable primary energy* per FU constitutes a useful and effective way for identifying processes that are likely responsible for most emissions and extractions. The primary nonrenewable energy flows are technically part of the impact assessment and not part of the inventory. Strictly speaking, the inventory results related to energy consist only of the mineral ore extractions needed for the energy carriers (petroleum, coal, gas, uranium, wood, etc.). These ore extractions are then multiplied by calorific values to obtain nonrenewable primary energy flows. Most analyses, however, consider the nonrenewable primary energy consumption in this early LCA phase, because it is often correlated to many inventory items and is thus an excellent way to test the magnitudes and validity of the inventory results.

Primary energy is defined as the energy contained in the energy carriers at the point of extraction from the environment. It is the sum of the *final energy* purchased by the consumer and the upstream energy usage for extraction, preparation, and distribution. The ratio between final energy and primary energy defines the *energy efficiency* of the supply chain.

A part of the primary energy is *nonrenewable*, which means that the basic resource of this energy is nonreplaceable or is replaced very slowly through natural processes. Nonrenewable primary energy generally stems from fossil fuels (petroleum, coal, natural gas) and uranium. This energy is eventually dissipated to the environment in the form of unusable heat. Energy from sources such as hydroelectric dams, thermal solar collectors and photovoltaic cells, wind power, or wood combustion is all technically renewable but requires the use of nonrenewable primary energy for the infrastructure manufacturing and use.

The nonrenewable primary energy demand and CO_2 emissions for different materials and processes are calculated and listed in the ecoinvent database (Section 4.3.2). Table 4.1 presents some typical values for a commonly used set of processes and materials, showing that the nonrenewable primary energy consumption and emissions from electricity vary substantially by country of origin; the consumer use of 1 kWh of Swiss electricity requires 7.9 MJ of nonrenewable primary energy, compared with 10.5 MJ needed for a European electricity mix and 12.1 MJ for a U.S. electricity mix. The difference is even more noticeable for CO_2, with a variation of more than a factor of four (from 0.11 kg_{CO2}/kWh for Switzerland to 0.49 kg_{CO2}/kWh for Europe and 0.71 kg_{CO2}/kWh for the United States). This difference is due to the composition of the Swiss electricity mix, 40% of which is nuclear and 57% hydroelectric, a renewable energy source that needs minimal nonrenewable energy for infrastructure. If the level of energy consumption varies by scenario, the choice of energy source can strongly influence the results of a study. However, the variation in nonrenewable primary energy requirements among different databases may be larger than the variation among regions within the same database. So, for the comparative purposes of an LCA, where consistent data sets are a priority, it is sometimes better to adapt high-quality data to another geographical context rather than compare electricity mixes from the appropriate regions but different databases.

The values provided in Table 4.1 must be interpreted with care and generally cannot simply be compared only on a per unit mass basis. For example, aluminum requires seven times more energy and emits seven times more CO_2 per unit mass than steel. However, these materials should be compared by function rather than mass and different amounts may be needed depending on the function. It is therefore necessary to relate all energy consumptions and CO_2 emissions back to the FU for the considered application (example in Table 4.2).

Appendix III provides the nonrenewable primary energy consumption and CO_2 emissions for a large number of materials and processes. System reference flows can be identified based on Section 3.3 concepts, and then Appendix III data may be used for a preliminary evaluation of the principal contributions to the total energy demand.

TABLE 4.1
Nonrenewable Primary Energy and CO_2 for Different Types of Energy Carriers, Materials

	Nonrenewable Primary Energy (MJ per unit)	CO_2 (kg per unit)	g_{CO2}/MJ ratio
Energy Carriers			
1 kWh electricity (Europe)	10.5	0.49	47
1 kWh electricity (United States)	12.1	0.71	59
1 kWh electricity (Japan)	11.5	0.53	46
1 kWh electricity (Switzerland)	7.9	0.11	13
1 kWh electricity (China)	10.4	0.98	94
1 L gasoline (no combustion)[a]	42.9	0.49	11
1 L gasoline (with combustion)	42.9	2.88	65
1 kg light oil (42.7 MJ final)	56.8	3.71	65
Transportation			
1000 km-kg transportation by 16–32-ton lorry	2.6	0.15	58
1 person-km by train (Intercity)	0.98	0.06	58
1 person-km by airplane (European flight)	3.28	0.19	60
1 person-km by car	3.0	0.17	57
Material			
1 kg steel, low alloy	27.4	1.63	59
1 kg primary aluminum	160.4	9.55	60
1 kg recycled aluminum	22.4	1.32	59
1 m^3 concrete	1,381	257	186
1 kg copper	31.2	1.86	60
1 m^3 water	5.55	0.30	54
1 kg newsprint paper	24.3	1.22	50
1 kg polyethylene HDPE[a]	76.4	1.56	20
1 kg glass	11.5	0.63	55
End of Life			
1 kg landfilled steel	0.197	0.00657	33
1 kg landfilled aluminum	0.521	0.02010	39
1 kg incinerated municipal solid waste (MSW)	0.43	0.50	1,161
1 kg incinerated polypropylene	0.209	2.53	12,060

Note: Figures are extracted from ecoinvent 2.2 and aggregated over the entire life cycle.

[a] Includes the gasoline extraction and refinement processes, but does not include combustion.

TABLE 4.2
Nonrenewable Primary Energy for an Output of 800 lm during 5000 h

LCA Stage	Reference Flows and Main Intermediary Flows	Specific Energy Demand	Nonrenewable Primary Energy
(a) Incandescent Light Bulbs			
Extraction and preparation of raw materials	0.02 kg glass/bulb × 5 bulbs/FU = 0.10 kg/FU	Glass: 11.5 MJ/kg	0.10 kg/FU × 11.5 MJ/kg = 1.15 MJ/FU
	0.015 kg copper/bulb × 5 bulbs/FU = 0.075 kg/FU	Copper: 31.2 MJ/kg	0.075 kg/FU × 31.2 KJ/kg = 2.34 MJ/FU
Manufacturing	5 bulbs/FU	0.38 MJ/bulb	1.90 MJ/FU
Packaging	5 bulbs/FU × 0.01 kg/bulb = 0.05 kg/FU	Paper: 24.3 MJ/kg	1.2 MJ/FU
Transportation	1000 km × 5 bulbs/FU × (0.035 kg/bulb + 0.01 kg/bulb) = 225 km-kg/FU	Transport by 16–32-ton truck: 2.6 MJ/1000 km-kg	0.57 MJ/FU
Usage	60 W/bulb × 1000 h/bulb × 5 bulbs/FU = 300 kWh/FU	Electricity (U.S.): 12.1 MJ/kWh	3633 MJ/FU
Waste disposal	5 bulbs/FU × (0.035 kg/bulb + 0.01 kg/bulb) = 0.225 kg/FU	Municipal waste incineration: 0.43 MJ/kg	0.1 MJ/FU
Avoided burden (burned paper in MSW incineration)	−0.05 kg/FU × 18 MJ/kg × 11%/3.6 MJ/kWh = −0.028 kWh/FU	Electricity (U.S.): 12.1 MJ/kWh	−0.33 MJ/FU
Total			3640 MJ/FU
(b) Fluorescent Light Bulbs			
Extraction and preparation of raw materials	0.06 kg electronics/FU	Electronics: 896 MJ/kg	53.7 MJ/FU
	0.1 kg glass/FU	Glass: 11.5 MJ/kg	1.15 MJ/FU
Manufacturing	1 bulb/FU	10.6 MJ/bulb	10.6 MJ/FU
Packaging	0.04 kg/FU	Paper: 24.3 MJ/kg	1.0 MJ/FU
Transport	1000 km × (0.16 kg/bulb + 0.04 kg/package) = 200 km-kg/FU	Transportation by 16–32 t truck: 2.6 MJ/1,000 km-kg	0.5 MJ/FU
Usage	13 W/bulb × 5000 h/bulb = 65 kWh/FU	Electricity (U.S.): 12.1 MJ/kWh	787 MJ/FU
Waste disposal	5 bulbs/FU × (0.035 kg/bulb + 0.01 kg/bulb) = 0.225 kg/FU	Municipal waste incineration: 0.43 MJ/kg	0.1 MJ/FU
Avoided burden (burned paper in MSW incineration)	0.04 kg/FU × 18 MJ/kg × 11%/3.6 MJ/kWh = −0.022 kWh/FU	Electricity (U.S.): 12.1 MJ/kWh	0.27 MJ/FU
Total			854 MJ/FU

4.2.2.2 Energy Consumption of Electric Light Bulbs

As an example of energy analysis, we quantify the energy consumption of each life cycle stage of a light bulb based on the FU introduced in Section 3.3.3. Incandescent light bulbs are composed of copper (15 g) and glass (20 g), which require extraction and preparation energies of 31.2 (copper) and 11.5 (glass) MJ/kg (Table 4.1). The FU is a service of 800 lm of illumination during 5000 h, which requires five bulbs, so the nonrenewable primary energy consumed for the extraction and preparation of these raw materials is calculated as follows: [(0.015 kg copper/bulb) × (31.2 MJ/kg copper)] + [(0.020 kg glass/bulb) × (11.5 MJ/kg glass)] × 5 bulbs/FU = 3.5 MJ/FU. Assuming the U.S. electricity mix, the energy consumption for the use of incandescent light bulbs is 300 kWh/FU × 12.1 MJ/kWh = 3633 MJ/FU.

We assume that the 0.05 kg/FU of packaging paper (lower heat value of 18 MJ/kg) is burned in a waste-to-energy municipal incinerator, with an 11% average energy efficiency for conversion to electricity. This substitutes 0.28 kWh of energy, corresponding to avoiding 0.33 MJ/FU, which is very small compared with the 3633 MJ/FU consumed.

Table 4.2 provides the remaining calculations comparing energy usage of incandescent and fluorescent light bulbs, showing that for both bulbs, the energy demand of the production stage is clearly less than the use stage. For this reason, future improvements should be focused on light bulb efficiency to maximize the number of lumens per watt.

Recent studies compare the environmental impacts of incandescent, fluorescent, and light-emitting diode (LED) bulbs (U.S. Department of Energy 2013; Principi and Fioretti 2014). Similar to the previous scenarios, the bulb use is still the major energy-consuming stage for LED bulbs. While nonrenewable energy consumption is similar for manufacturing fluorescent bulbs and LED bulbs (the latter having a slightly lower energy consumption), there is a greater potential to improve LED performance, which is a less established technology. Regarding toxicity and resource depletion, fluorescent bulbs and LED bulbs may have a higher impact than incandescent light bulbs because of their metallic content (Lim et al. 2013).

4.2.2.3 Assessment of CO_2 Emissions

CO_2 emissions are assessed in a similar fashion to that of energy. CO_2 emissions are generally correlated with nonrenewable primary energy usage, since a large part of anthropogenic CO_2 emissions arise from the combustion of fossil energy carriers. The rule of thumb is that high energy consumption implies high CO_2 emissions unless nuclear energy is used. Furthermore, the amount of CO_2 formed per unit of energy depends on the type of combustible used; the larger the hydrogen to carbon ratio (H/C ratio) of combustible molecules, the less CO_2 will be produced. Natural gas, essentially made up of methane (CH_4: 50 $g_{CO2}/MJ_{eq\text{-}gas}$), therefore results in fewer CO_2 emissions per energy used than coal (typically $C_{24}H_{12}$: 80 $g_{CO2}/MJ_{eq\text{-}gas}$). Due to degradation or incineration, a substantial portion of CO_2 emissions may also occur during the end-of-life stage of a product, making it essential to account for. Since many materials have well-established ratios of CO_2 emissions to energy usage, you can compare the ratio found in your LCA with established ratios for main LCA processes to verify results.

4.2.2.4 Checking the Ratio of CO_2 Emitted per Megajoule of Nonrenewable Primary Energy

Figure 4.2 provides the ratios of CO_2 to nonrenewable primary energy for an array of materials, fuels, and transportation methods. The ratios for energy sources and transportation are tabulated for 50 individual processes in Appendix III.

Plastic materials, produced from petroleum, have a ratio close to 30 g_{CO2}/MJ for their production only. This ratio grows to 60 g_{CO2}/MJ if waste treatment is accounted for, because incineration or long-term degradation result in additional CO_2 emissions. Fuels themselves have a very low ratio of less than 10 g_{CO2}/MJ if combustion is excluded, demonstrating the importance of considering the fuel's end of life. For other processes using petroleum-related products (oil for heating or road, air, and marine transportation), this ratio is 55–70 g_{CO2}/MJ. The highest ratios occur for materials such as mortar, cement, and concrete (130–230 g_{CO2}/MJ), which release CO_2 through a chemical reaction while drying.

The ratio of CO_2 to nonrenewable primary energy is around 50 g_{CO2}/MJ for natural gas, which is lower than the ratio of oil and coal (80–90 g_{CO2}/MJ) due to its smaller H/C ratio, as discussed in the previous subsection.

Electricity can have highly variable g_{CO2}/MJ ratios, partly due to the combustible involved, as previously mentioned. The value for nuclear electricity is very low (< 1 g_{CO2}/MJ), while the ratio for hydroelectric power adds up to around 80–90 g_{CO2}/MJ (construction of concrete dams and low consumption of nonrenewable primary energy). Swiss electricity, on average, has a relatively low ratio

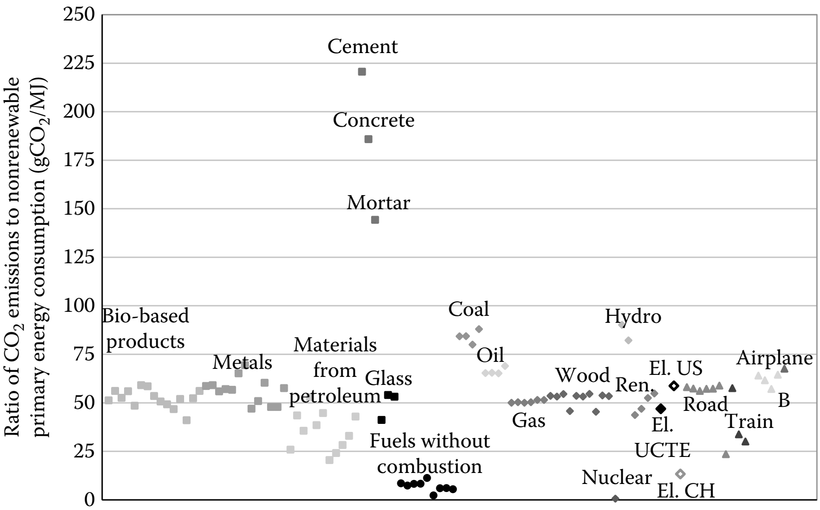


FIGURE 4.2 Ratios of fossil CO_2 emissions to consumption of nonrenewable primary energy for different materials, energy systems, and means of transportation. The coal, oil, gas, and wood categories represent the energy processes yielding electricity or heat. "Ren." refers to renewable wind and solar energy, as well as heat pump technology. "El. CH," "El. UCTE," and "El. US" refer to the Swiss, European, and US electricity mixes, respectively, calculated from the ecoinvent database.

(16 g_{CO2}/MJ) due to the high usage of nuclear power. The average ratio of the European electricity mix is 43 g_{CO2}/MJ, which is lower than that of the more fossil fuel-dominated electricity mix of the United States of 50 g_{CO2}/MJ.

For an FU that involves reference flows of electricity (such as a light bulb), these different ratios of CO_2 emissions per primary energy can lead to substantially different emissions per electricity use; the CO_2 emissions of the U.S. mix sum to 0.71 kg_{CO2}/kWh, compared with 0.5 kg_{CO2}/kWh in Europe and Japan, and only 0.11 kg_{CO2}/kWh in the Swiss mix.

4.2.2.5 CO_2 Assessment of Electric Light Bulbs

The CO_2 analysis of incandescent light bulbs confirms the results obtained for energy consumption, showing that the use stage results in the majority of CO_2 emissions (Table 4.3).

The ratios of CO_2 emissions to nonrenewable primary energy consumption are, as expected, close to 58 g_{CO2}/MJ for transport and 59 g_{CO2}/MJ for U.S. electricity usage (Figure 4.2). These ratios allow a quick check of the consistency of the results. The high ratio for manufacturing is due to the use of Chinese electricity.

4.2.2.6 Classifying Products

Based on the quantification of nonrenewable primary energy consumption and CO_2 emissions by life cycle stage, products can be classified in a generic manner according to the following criteria (where a stage is considered "dominant" when it has the most energy consumption or CO_2 emissions):

TABLE 4.3
CO_2 Emissions Due to Incandescent Electric Light Bulbs per Functional Unit[a]

LCA Stage	Reference Flows and Main Intermediary Flows (unit/FU)	Emission per Unit (kg_{CO2}/unit)	Emission per FU (kg_{CO2}/FU)	Ratio Check (g_{CO2}/MJ)
Extraction and preparation of raw materials	Glass: 0.10 kg/FU	Glass: 0.63 kg_{CO2}/kg	Glass: 0.063	55
	Copper: 0.075 kg/FU	Copper: 1.86 kg_{CO2}/kg	Copper: 0.14	60
Manufacturing	5 bulbs/FU	0.035 kg_{CO2}/bulb	0.175	94
Packaging	0.05 kg/FU	1.59 kg_{CO2}/kg	0.08	53
Transport	225 km·kg/FU	0.15 kg_{CO2}/1.000 km-kg	0.0331	58
Usage	300 kWh/FU	0.711 kg_{CO2}/kWh	213	59
Waste disposal	0.225 kg/FU	0.50 kg_{CO2}/kg	0.11	1160
Avoided burden (burned paper in MSW incineration)	−0.028 kWh/FU	0.711 kg_{CO2}/kWh	−0.02	59
Total			214	

[a] Output of 800 Im for 5000 h.

- Active versus passive: A product is active if the use stage is dominant. In the case of an active product, the ecodesign efforts should be focused on increasing energy efficiency during use. For passive products, efforts should focus on the choice of material used, the recycling of raw materials, and the product effective lifetime.
- Mobile versus fixed: A product is mobile if the transport stage is responsible for the dominant energy consumption and CO_2 emissions. The components of a car are generally mobile, because most energy consumption is associated with transportation. For such a product, weight reduction is essential.

According to this classification, an electric bulb is an active fixed product, thus making it important to select bulbs with the highest efficiency. An electric drill is an active product when developed for and used intensively within a profession, but may be de facto a passive product in the case of private individual usage, if the drill is only used for a few minutes per year. In this latter case, energy consumption associated with raw materials and drill manufacturing may dominate over lifetime usage.

For passive products, the lifetime of a material or component is only important if it directly influences the usage duration of the product or service that is offered. For example, the actual lifetime of a single-use plastic bottle is generally not important, because it will last much longer than the time needed for the function considered, which is to contain the drink until it is consumed.

4.2.3 Example of Process-Based Life Cycle Inventory: Front-End Panel of an Automobile

Although energy consumption and CO_2 emissions are often good indicators of overall emissions and impacts of energy-related processes (Beck 1999; Huijbregts et al. 2010), they are not sufficient to estimate other impacts, such as human toxicity and ecotoxicity. For these impact categories, all emissions must be carefully inventoried to analyze their different impacts. The generalization of the inventory analysis phase to a large number of substances emitted or extracted is illustrated using the concrete example of the front-end panel of a car. For this study, the goal definition (the first phase of the LCA) can be summarized as follows:

- Function and FU: The function of the front-end panel is to hold various parts (lights, ventilator, etc.) throughout the lifetime of the automobile. The FU is a front-end panel with an adequate rigidity, transported over 200,000 km, which is a typical distance traveled by an automobile over its lifetime.
- System boundary: The analysis accounts for the entire chain of energy extraction and preparation needed for the component, the manufacturing and disposal of the component, and, most importantly, the component use during car operation.
- Scenarios and reference flows: Four scenarios are considered and compared, each with a front-end panel made from one of the following

materials—steel, a composite plastic material, virgin aluminum, and recycled aluminum. The necessary reference flows for each scenario are listed in Table 4.4.

Table 4.4 presents the reference and first-tier intermediary flows for each front-end panel scenario, including the material mass and electricity use for each process in the extraction, manufacturing, use, and end-of-life stages. The quantity of gasoline used to transport a front-end panel over 200,000 km is calculated assuming consumption of 0.00004 L more gasoline for each additional kilogram that is carried 1 km. At the end of life, the steel and virgin aluminum front-end panels are placed in landfills. The composite front-end panel is assumed to be incinerated in an incineration plant for household waste. Finally, the recycled aluminum front-end panel is recycled once more after usage, so we do not include any ultimate end of life in this scenario.

The emission and extraction factors for each reference flow are given in Table 4.5, and Table 4.6 presents the final inventory results of extraction and emission.

The emission and extraction matrix **E** lists the emissions or extraction factors of each substance over the whole production chain for each process; it specifies the extraction from or emission to different environmental media (air, water, and soil) per main intermediary unit process (material, electricity, etc.). Table 4.5 presents an excerpt of this matrix for a few substances and for the nonrenewable primary energy consumption. The complete inventory considers more than 500 substances.

TABLE 4.4
Reference Flows and Main Intermediary Flows for a Front-End Panel Transported over 200,000 km

	Unit	Steel	Composite	Virgin Aluminum	Recycled Aluminum
Materials					
Final weight	kg	10.0	7.0	3.8	3.8
Manufacturing					
Electricity	kWh	19.7	4.7	15.2	15.2
Oil	kg	2.3	0.56	1.8	1.8
Use					
Gasoline	L	80.0	56.0	30.4	30.4
End of Life					
Incineration	kg	–	7.0	–	–
Controlled landfilling	kg	–	–	3.8	–
Landfilling for inert materials	kg	10.0	–	–	–

TABLE 4.5

Matrix E of Aggregated Emission and Extraction Factors for the Inputs Involved in the Production of a Front-End Panel Transported over 200,000 km (excerpt from ecoinvent 1.0[a])

		Steel	Composite Material	Nonrecycled Aluminum	Recycled Aluminum	Electricity (Europe)	Oil	Gasoline	Landfilled Steel	Landfilled Aluminum	Propylene Incineration
		kg	kg	kg	kg	kWh	kg	L	kg	kg	kg
Resources											
Energy	MJ	24.7	79.9	162	21.8	10.5	56.9	43.2	0.21	0.53	0.21
Emissions to Air											
CO_2	kg	1.28	1.85	9.50	1.20	0.45	3.67	2.80	0.01	0.02	2.54
CO	kg	0.023	0.00076	0.0057	0.0011	0.00016	0.0013	0.00067	0.000042	0.000097	0.00026
CH_4	kg	0.0027	0.0060	0.015	0.0015	0.00064	0.0032	0.0013	1.8E–05	3.8E–05	2.2E–05
$N2_O$	kg	3.8E–05	1.3E–07	0.00027	2.5E–05	1.1E–05	4.1E–05	8.1E–06	2.1E–07	5.4E–07	4.8E–06
NO_x	kg	0.0054	0.0096	0.022	0.0025	0.00082	0.0037	0.0018	0.00015	0.00029	0.00039
SO_2	kg	0.0040	0.013	0.038	0.0035	0.0018	0.0052	0.0044	1.1E–05	3.0E–05	1.9E–05
Particles	kg	0.0021	0.00038	0.0055	0.00043	0.00012	0.00024	0.00018	1.4E–05	2.8E–05	1.3E–05
Pb	kg	5.6E–06	5.1E–09	1.9E–06	4.2E–05	6.5E–08	3.5E–07	1.7E–07	2.3E–09	1.0E–08	7.7E–09
Emissions to Water											
Nitrates	kg	1.6E–05	1.9E–05	1.9E–04	1.6E–05	7.9E–06	1.1E–05	7.4E–06	5.3E–08	1.9E–07	5.4E–05
Pb	kg	1.5E–05	1.0E–06	1.1E–05	4.8E–06	4.5E–07	1.3E–06	7.1E–07	2.9E–08	2.6E–05	1.5E–06

[a] For future studies, we recommend using the latest ecoinvent data for these factors.

TABLE 4.6
Inventory of Emissions and Extractions for a Front-End Panel Transported over 200,000 km

Substance	Unit	Steel	Composite	Aluminum	Recycled Aluminum
Resources					
Energy	MJ	4043	3061	2193	1658
Emissions in air					
CO_2	kg	253.9	176.4	134.6	103.2
CO	kg	0.294	0.045	0.047	0.029
CH_4	kg	0.154	0.122	0.112	0.062
N_2O	kg	0.0013	0.0005	0.0015	0.0006
NO_x	kg	0.221	0.172	0.156	0.083
SO_2	kg	0.439	0.348	0.315	0.184
Particles	kg	0.0383	0.0136	0.0287	0.0095
Pb	kg	7.16×10^{-5}	1.03×10^{-5}	1.41×10^{-5}	1.65×10^{-4}
Emissions in water					
Nitrates	kg	9.30×10^{-4}	6.44×10^{-4}	1.07×10^{-3}	4.25×10^{-4}
Pb	kg	2.13×10^{-4}	5.13×10^{-5}	9.70×10^{-5}	4.91×10^{-5}

For each scenario (Equation 4.1), the inventory vector *u* of emissions and extractions per FU (Table 4.6) is the product of the matrix of aggregated emission and extraction factors **E** (Table 4.5) with the demand vector *y* of first-tier intermediary flows (Table 4.4).

$$E \times y = u \tag{4.1}$$

$$\begin{pmatrix} 24.7 & 10.5 & 56.9 & 43.2 & 0.21 \\ 1.28 & 0.45 & 3.67 & 2.80 & 0.01 \\ 0.023 & 0.00016 & 0.0013 & 0.00067 & 4.2\times10^{-5} \\ 0.027 & 0.00064 & 0.0032 & 0.0013 & 1.8\times10^{-6} \\ 3.8\times10^{-5} & 1.1\times10^{-5} & 4.1\times10^{-5} & 8.1\times10^{-6} & 2.1\times10^{-7} \\ 0.0054 & 0.00082 & 0.0037 & 0.0018 & 0.00015 \\ 0.0040 & 0.0018 & 0.0052 & 0.0044 & 1.1\times10^{-5} \\ 0.0021 & 0.00012 & 0.00024 & 0.00018 & 1.4\times10^{-5} \\ 5.6\times10^{-6} & 6.5\times10^{-8} & 3.5\times10^{-7} & 1.7\times10^{-7} & 2.3\times10^{-9} \\ 1.6\times10^{-5} & 7.9\times10^{-6} & 1.1\times10^{-5} & 7.4\times10^{-6} & 5.3\times10^{-8} \\ 1.5\times10^{-5} & 4.5\times10^{-7} & 1.3\times10^{-6} & 7.1\times10^{-7} & 2.9\times10^{-8} \end{pmatrix} \begin{pmatrix} 10.0 & \text{kg steel} \\ 19.7 & \text{kWh electricity} \\ 2.3 & \text{kg fuel} \\ 80.0 & \text{L gasoline} \\ 10.0 & \text{kg landfilled steel} \end{pmatrix}$$

$$= \begin{pmatrix} 4{,}043 & \text{MJ} \\ 253.9 & \text{kg}\,CO_2 \\ 0.29 & \text{kg}\,CO \\ 0.15 & \text{kg}\,CH_4 \\ 0.0013 & \text{kg}\,N_2O \\ 0.22 & \text{kg}\,NO_x \\ 0.44 & \text{kg}\,SO_2 \\ 0.038 & \text{kg}\,\text{particles} \\ 7.2\times 10^{-5} & \text{kg}\,\text{Pb(air)} \\ 9.3\times 10^{-4} & \text{kg}\,\text{nitrates} \\ 2.1\times 10^{-4} & \text{kg}\,\text{Pb(water)} \end{pmatrix}$$

The units and calculations can be checked by writing out the matrix multiplication for a whole row, such as CO_2 emissions in the second row:

1.28 kg_{CO2}/kg steel × 10.0 kg steel/FU + 0.45 kg_{CO2}/kWh electricity × 19.7 kWh electricity + 3.67 kg_{CO2}/kg fuel × 2.3 kg fuel/FU + 2.80 kg_{CO2}/L gas × 80.0 L gas/FU + 0.01 kg_{CO2}/kg landfilled steel × 10.0 kg landfilled steel/FU = 253.9 kg_{CO2}/FU.

In comparing the inventories for the different scenarios (Table 4.6), we see that no single scenario has the minimum values for all polluting substances and energy consumption. Indeed, the steel scenario results in the highest CO_2, CO, and SO_2 air emissions, but the virgin aluminum scenario emits more nitrates, and the recycled aluminum scenario emits more lead (Pb) to air. For many substances (e.g., CO_2, CH_4, NO_x), the composite scenario emits less than the steel scenario, but more than both aluminum scenarios. At this stage, it is therefore impossible to rank the scenarios by environmental impact. The subsequent impact assessment phase (Chapter 5) is essential to compare the impacts generated by these different emissions and thus compare scenarios.

It is nevertheless interesting to interpret the raw inventory results, which have less uncertainty than the impact assessment results and can already provide guidance on the effects of different assumptions or choices. The most primary energy is consumed in the steel scenario, which is mainly due to gasoline usage under the assumption of 200,000 km travel distance. By recalculating primary energy usage based on varying distances (Figure 4.3), we find that although the recycled aluminum front-end panel consumes the least energy regardless of distance traveled, the virgin aluminum front-end panel actually consumes more energy than the steel and composite front-end panels below around 27,000 and 50,000 km, respectively. The steel scenario consumes more primary energy over its life cycle than the composite scenario as soon as the distance traveled surpasses 10,000 km.

In the example given previously, the calculation is based on the emission and extraction factors provided by large inventory databases, and are the result of aggregated data from hundreds of unit processes. The use of these factors is easy, but determination of their values requires lengthy, rigorous work; thus, it is advantageous

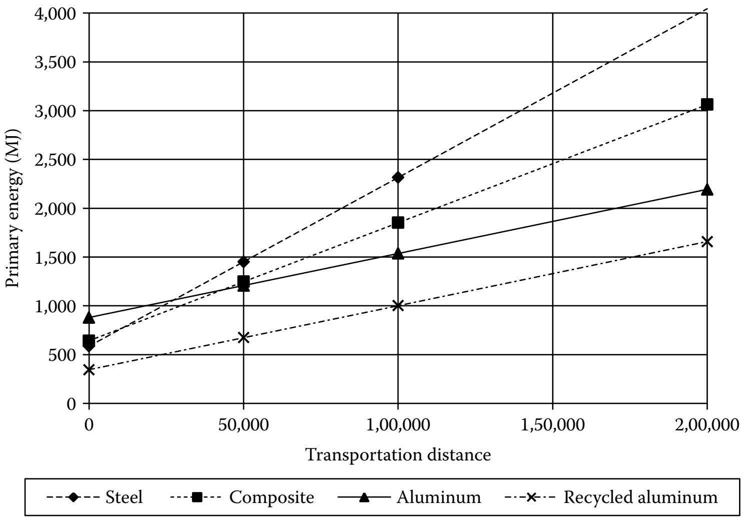


FIGURE 4.3 Variation of the nonrenewable primary energy consumption of a front-end panel as a function of the distance traveled by the vehicle.

to use existing databases whenever possible. Use of such databases also helps scale up the simple and practical approach illustrated here to calculate the environmental inventory for a large number of unit processes using matrix calculations (see Section 4.2.4).

4.2.4 Generalization and Process Matrix Approach

As stated in Section 3.5, the system boundary must be carefully delimited and include all relevant background processes necessary for the direct processes considered. For example, the total emissions and extractions associated with the production of 1 kg of aluminum must include those associated with its extraction, fabrication, disposal, and any other important stages. But should we also account for the creation of the machines that built the infrastructure necessary to extract the raw materials? And should we account for the energy needed to create these machines? And the aluminum needed to extract the energy to create the machines? If so, the chain of processes to consider for the provision of 1 kg of aluminum becomes infinitely long with closed loops (Figure 4.4).

Accounting for this infinitely long process chain can be addressed by matrix inversion, using the approach taken by the ecoinvent inventory database, which uses a model constituted of the technosphere (economic system) and the ecosphere (environmental system). Each unit process can exchange intermediary flows with any number of the m total unit processes of the technosphere and can be associated with any of the n elementary flows extracted from or emitted to the environment. The *technology matrix* **A** ($m \times m$) is a square matrix consisting of a row and column entry for every unit process in the economy (Equation 4.2) and the *environmental matrix*

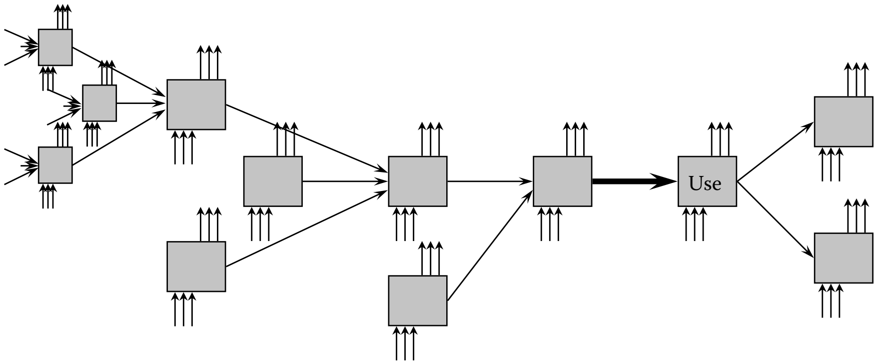


FIGURE 4.4 Flowchart of background processes. Boxes represent processes which are connected by elementary flows represented by horizontal arrows, with elementary flows of extractions indicated by lower arrows and of emissions indicated by upper arrows.

B ($n \times m$) has a column for every unit process and a row for every elementary flow from or to the environment (Equation 4.3).

$$\mathbf{A} = \begin{pmatrix} a_{11} & \cdots & a_{1m} \\ \vdots & \ddots & \vdots \\ a_{m1} & \cdots & a_{mm} \end{pmatrix} \tag{4.2}$$

$$\mathbf{B} = \begin{pmatrix} b_{11} & \cdots & b_{1m} \\ \vdots & \ddots & \vdots \\ b_{n1} & \cdots & b_{nm} \end{pmatrix} \tag{4.3}$$

The element a_{ij} (row i, column j) of the **A** matrix represents the amount of technological process i used by process j, and the element b_{kj} of the matrix **B** is the elementary flow of substance k extracted from the environment or emitted in the environment through process j. In other words, column j of the **A** matrix contains the amount of all processes used by process j. Similarly, column j of the **B** matrix contains all extractions and emissions directly associated with process j.

The matrix **E** of aggregated emission and extraction factors for each unit process (equivalent to Table 4.1 or Table 4.5) is a sum of the following infinite chain:

- The direct elementary flows for each first-tier unit process (**B**)
- The elementary flows associated with the processes needed for each unit process (**B**·A) (second tier, e.g., the flows associated with the machine needed)
- The elementary flows associated with the processes that are needed for the processes needed for each unit process (**B**·A^2) (e.g., the flows associated with the machine that made the machine needed)
- And so on

This can be expressed as follows (Equation 4.4):

$$\mathbf{E} = \mathbf{B}\left(\mathbf{I} + \mathbf{A} + \mathbf{A}^2 + \mathbf{A}^3 + \mathbf{A}^4 \ldots\right) = \mathbf{B}\left(\mathbf{I} - \mathbf{A}\right)^{-1} \tag{4.4}$$

where **I** is the identity matrix with entries of 1 along the diagonal (and zeros everywhere else); just as $1 + x^2 + x^3 = (1-x)^{-1}$ for $x < 1$, $(\mathbf{I} + \mathbf{A} + \mathbf{A}^2 + \mathbf{A}^3 + \mathbf{A}^4 \ldots) = (\mathbf{I} - \mathrm{A})^{-1}$ for $a_{ij} < 1$ for all i, j. This matrix inversion allows the inclusion of an infinite process chain in theory, but, practically, the system is still truncated because certain unit processes are just not taken into account in the process LCA (emissions from law offices, hotel use, etc., as described in Section 3.5).

The emissions and extractions inventory vector u (each column in Table 4.6) is calculated by multiplying matrix **E** (Table 4.5) by the demand vector y (Table 4.4) that quantifies first-tier intermediary flows or inputs per FU (Equation 4.5).

$$u = \mathbf{E} \times y = \mathbf{B}\left(\mathbf{I} - \mathbf{A}\right)^{-1} y = \mathbf{B} \times x \tag{4.5}$$

where $x = (\mathbf{I} - \mathrm{A})^{-1} y$ is the total output vector, that is, the total amount of goods and services in each sector needed to meet the demand.

4.3 INVENTORY DATABASES FOR PROCESS-BASED APPROACH

During goal setting, the system boundaries are defined and all processes included within these boundaries are listed and quantified. To calculate the associated elementary flows, both incoming (raw materials and energy carriers) and outgoing (substances released in air, water, etc.), process inventory data are sought from industrial partners or from the literature. Obtaining reliable inventory data, clearly described and regularly updated, is not easy and could severely hinder the application of life cycle assessment, but existing published inventory databases now facilitate this work. After a brief overview of the existing databases, we will focus on ecoinvent, one of the leading databases. We will also emphasize the importance of data quality analysis, which is an integral part of the LCA.

4.3.1 Existing Databases

The worldwide inventory databases available to the public are listed by Norris and Notten (2002) and Sonneman and Vigon (2011), building on previous work by the Society for the Promotion of Life Cycle Assessment (SPOLD; Fussler 1993; Weidema 1999). This collection is regularly updated under the Life Cycle Initiative of the United Nations Environment Programme (UNEP), and a set of global guidance principles for life cycle assessment databases was published following a consensus-building workshop in 2011 (Sonneman and Vigon 2011). This document provides guidelines on how to build unit processes as well as aggregate data sets within a consistent database. The European Commission also provides a regularly updated collection of LCA resources, including access to 20 database websites, 35 tools, and 80 service providers. (See Appendix I for the URLs of these collections.)

The majority of early databases was developed in Europe in the mid-1980s. These databases were principally developed for studies done by universities or consultants to characterize specific industrial sectors or product groups. The resulting databases were very diversified, fragmented, and not well harmonized.

The development of more consistent databases began mainly in Switzerland with an early study by the OFEFP (Office Fédéral de l'Environnement des Forêts et du Paysage) focusing on packaging materials (Bus 1984; Habersatter and Widmer 1991). The study, which consisted of studying the environmental consequences of different types of packaging (aluminum, glass, plastic, paper, cardboard, and tin), required higher-quality data on energy systems. In response to this need, the ESU (Energie-Stoff-Umwelt) group of the Eidgenössische Technische Hochschule Zürich (ETH Zürich) created more complete databases on the following energy systems Frischknecht et al. (1994):

- Petroleum products
- Natural gas
- Nuclear energy
- Hydroelectric energy
- Geothermal energy
- Coal (coal, lignite)
- Wood
- Solar-thermal energy
- Photovoltaic energy
- Mixed energy

Using this energy database, the data on packaging materials were updated by Habersatter and Fecker (1998) (Swiss Federal Office for the Environment, Forests, and Landscape [OFEFP] 250). This led to a collaboration among the Swiss Federal Institutes of Technology, with a goal to create a centralized, coherent, and more complete database, called *ecoinvent* (Section 4.3.2).

The European Reference Life Cycle Database (ELCD) with European scope inventory data sets was created by the Joint Research Centre (JRC) at Ispra, Italy. The ELCD core database is comprised of Life Cycle Inventory (LCI) data from EU-level business associations, as well as other sources for raw materials, energy carriers, transport, and waste management. Efforts have been focused on data quality, consistency, and applicability, but data are only provided at an aggregated level from cradle to gate, without detailed information on unit processes. This limits the use of this database for uncertainty analyses (common processes cannot be accounted for—see Section 6.5) or for more advanced studies on system boundaries. This database is accessible free of charge with unrestricted use for all LCA practitioners. The European Platform on Life Cycle Assessment also provides the guidance document, the *International Reference Life Cycle Data System (ILCD) Handbook*, which describes available and recommended practices for LCA data in general, LCI data sets, and LCIA.

For Europe, in addition to ecoinvent and the JRC ELCD databases, the following country-specific databases are available: (a) LCA Food Database—Denmark (from the Danish Institute of Agricultural Sciences; data also available in the LCA tool SimaPro); (b) Swedish National LCA database SPINE@CPM (contains more than 500 well-documented LCI data sets in the SPINE format); (c) IVAM Environmental Research database on Dutch building materials (based in Amsterdam, the Netherlands); and (d) other sector-specific databases on industry association websites (e.g., Association of Plastic Manufacturers in Europe and European Aluminium Association).

In Asia, the LCA National Project in Japan, funded by the Ministry of Economy, has developed LCI data for approximately 200 products, with data gathered from about 50 industrial associations (Narita et al. 2004). The Korea National Cleaner Production Center (KNCPC) is constructing the national Korean LCI database for Korean industries (Korea National LCI Database, KNCPC 2010), established with the support of the Ministry of Commerce, Industry, and Energy. The database is based on a series of industry-requested surveys and is accessible through the KNCPC website (listed in Appendix I).

In Australia, the Life Cycle Inventory Data Research Program is led by the RMIT in Melbourne and aims to develop detailed inventory resources. The Centre for Design's LCA resources are published in spreadsheets, and are also available in SimaPro LCA (see website in Appendix I) software.

For North America, the inventory database of Franklin LCI 98 is freely available in software such as SimaPro. However, these data must be used with caution due to certain inconsistencies with other LCI data sets. For example, many types of paper in the Franklin database require nonrenewable primary energies comparable to if not greater than that of plastics (80 MJ/kg paper), while this value is 15–30 MJ/kg paper in the ecoinvent database. The National Renewable Energy Laboratory (NREL) is creating a new North American database (US LCI database). This database will need to be verified and compared with other databases due to errors already identified, as well as the calling of intermediary flows that do not yet correspond to any existing process. The LCA Digital Commons from the US Department of Agriculture National Agricultural Library aims to provide open-access life cycle assessment data sets and tools. The project makes North American LCA data more accessible to the community of researchers, policy-makers, industry process engineers, and LCA practitioners. The North American project likely to provide very reliable regional data is currently underway at the Canadian institution, the International Reference Centre for the Life Cycle of Products, Processes and Services (CIRAIG), with the goal to adapt ecoinvent to North American conditions and produce a database for Quebec (already available), Canada, and, eventually, North America as a whole.

Databases are also developed for classes of products, such as the World Food LCA Database (http://www.quantis-intl.com/microsites/wfldb/) that provides data for more accurate food and beverages LCAs.

In summary, the compatibility and coherence of a database must be verified prior to usage or combination with other databases. For example, using established data from another continent can be better than mixing data from noncompatible databases.

4.3.2 Ecoinvent

4.3.2.1 The Project and Its Products

Ecoinvent is a project aiming to combine and enhance different existing inventory databases to obtain a unified and generic inventory data set of extremely high quality. Initially developed for Switzerland and western European countries, it is increasingly adapted to global data sets.

4.3.2.2 Description of the Ecoinvent 2.2 Database

Inventory data (v.2.2) are compiled for a large number of products and services, representing production and supply mostly from the year 2000. In addition to the quantitative information about inflows and outflows, supplementary descriptive information (metainformation) is provided on the technological, temporal, and geographic validity.

The database is organized according to the following main categories:

- Energy sources
- Construction materials and processes
- Chemicals
- Detergents
- Paper
- Waste treatment services
- Agricultural products and processes
- Transportation

The ecoinvent database consists of more than 4000 processes linked by material and energy flows covering more than 400 substances and resources. Ecoinvent CO_2 emissions and nonrenewable primary energy use are provided for approximately 50 processes in Appendix III.

4.3.2.3 Principal Characteristics of the Database

The ecoinvent database was initially developed for western Europe, with country- or region-specific values for certain processes. Whenever possible, data are provided on a unit process level, and only aggregated when unit process data are not available or confidential. The most common types of processes and emissions are described in the following paragraphs.

The inflows and outflows for the production processes are generally provided separately from those of the production infrastructure, allowing the user to choose whether to include certain infrastructures.

Electricity is modeled based on average electricity mixes, which are available for multiple European countries and other countries such as the United States, Japan, and China. Electricity mixes specific to other countries may be available and used to calculate new mixes. The database differentiates the production mix from the supply mix, where the latter is used for most processes requiring an energy demand. The production mix is only used for processes within the electric sector.

Transportation often occurs between the processes of a system. Due to the difficulty in determining the means and distances of transport for all individual intermediary products, standard distances are used by default.

Waste treatment is modeled like all other technical processes as another part of the system. If the specific waste treatment processes are not known, generic treatment processes are applied.

Certain elementary flows are always neglected; specifically, the sound emissions (noise) and H_2, N_2, and O_2 air emissions.

Regarding the material and energy flows, the nonrenewable primary energy is calculated based on the lower heating value (LHV) given in Table 4.7. The higher heating value (HHV) of a fuel is the amount of energy contained in the fuel; the LHV is the effective heat released during combustion, determined by subtracting the heat of vaporization of the water vapor from the HHV. For uranium, it is the reduction of its potential heating value due to its use in a power plant that is taken into account.

For air emissions, certain common pollutants are treated as follows:

- Benzene emissions are reported under the "benzene" label rather than as "aromatic hydrocarbons" or nonmethane volatile organic compounds (NMVOCs). When NMVOCs and benzene emissions are both measured and reported, the NMVOC emissions are input after subtracting benzene emissions.
- Particulate emissions to the air are classified based on three categories of particulate diameters. $PM_{2.5}$ refers to particles less than 2.5 μm in aerodynamic diameter, PM_{10}–$PM_{2.5}$ are particles between 2.5 and 10 μm, and TPM–PM_{10} are particles greater than 10 μm (total particulate matter [TPM] minus particles smaller than 10 μm).
- For CO_2, CO, and CH_4, a distinction is made between fossilized sources and biogenic sources. In the biogenic case, the carbon fixed during biomass growth is considered a CO_2 extraction from the atmosphere, which is rereleased during combustion or degradation. Maintaining this distinction allows the correct accounting of waste treatment processes responsible for

TABLE 4.7
Lower Heating Value (LHV), Higher Heating Value (HHV), and Densities of Fuel Sources in the Ecoinvent Database

		LHV (MJ)	HHV (MJ)	Density (kg/L[a])
Gasoline	kg	42.8	45.8	0.75
Diesel	kg	42.8	45.5	0.84
Kerosene	kg	43.3	46.0	0.80
Light oil	kg	42.7	45.4	0.84
Heavy oil used in boilers (Switzerland)	kg	40.6	43.0	0.95
Heavy fuel used in boilers/electric plants (Europe)	kg	40.0	42.3	1.00
Natural gas (Europe)	Nm^3	36.8	40.4	0.80
Nuclear	kg	560,000	–	–
Hardwood, dry	kg	18.3	–	239
Softwood, dry	kg	19.1	–	169
Mixed wood	kg	18.9	–	189

Source: Dones, R., et al., 2004. *Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries*, Data v1.1, ecoinvent report No. 5, Dübendorf, Switzerland.

[a] Except for the density of wood, which is expressed in kilograms per cubic meter.

a significant fraction of CO_2 emissions. If biogenic CO_2 fixed during biomass growth is accounted for, special care must be taken to ensure that the corresponding release of CO_2 is also taken into account during usage (e.g., food, combustion processes) or product end of life (e.g., incineration, landfill). A pragmatic solution to ensure reliability is to only account for fossil (nonbiogenic) CO_2 releases, and to assume that biogenically fixed CO_2 is eventually released over the whole life cycle. Note that in the case where methane is released instead of CO_2, this should then be taken into account due to the differences in the global warming potential of each substance.

- Sulfur and nitrogen oxides, SO_x and NO_x, are each grouped under the SO_2 and NO_2 labels, respectively.
- Air emissions of trace elements are provided by metal type and speciation, when available.
- For polycyclic aromatic hydrocarbons (PAHs), benzo[*a*]pyrene emissions are provided separately.
- The dioxin and furan emissions are expressed as equivalent 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) emissions.

For water-based transmissions, four aggregated parameters characterizing organic carbon content are reported: the biological oxygen demand (BOD5), the chemical oxygen demand (COD), the dissolved organic carbon (DOC), and the total organic carbon (TOC). If necessary, these parameters are calculated from individual pollutant quantities, which are also available in the inventory.

Particular attention has been given to land use and transformation. Land use contributes to increased competition between users, biodiversity loss, changes on climatic equilibrium, and the degradation of cultural assets. A distinction is made between land use (surface and duration required for the production of a certain quantity of goods and services) and land transformation (which relates the state of the land throughout an economic activity to its earlier and later state). No regional differentiation can be made, since data is collected at the level of national averages. As far as possible, unit processes are characterized by a geographic code, which specifies the country or continent where the use and transformation of soil take place.

In the ecoinvent database, careful attention has been given to the quality of data and their analysis. Section 6.5 describes the approach used to characterize uncertainty, and this can be applied to the assessment of new data.

4.3.2.4 New Features of Ecoinvent 3.1

The updated ecoinvent 3.1 improves on v.2.2 by consistently modeling water flows throughout the whole database, enabling the practitioner to determine water use and consumption to calculate the water footprint of products (Weidema et al. 2011). It also provides updated data sets for electricity production (more than 20 additional countries worldwide), the wood sector, recycling activities, chemical production, and fruit and vegetables. In addition to the more traditional attributional modeling, ecoinvent 3.1 can potentially support cut-off and consequential system modeling.

In the cut-off model, "the primary producer does not receive any credit for the provision of any recyclable materials. As a consequence, recyclable materials are available burden-free to recycling processes, and secondary recycled materials bear only the impacts of the recycling processes" (www.ecoinvent.org). Alternatively, allocation at the point of substitution allocates "the valuable by-products of treatment systems together with the activity that produced the material for treatment" (www.ecoinvent.org). While this is beneficial in avoiding difficult allocations, its present implementation may be problematic. For example, in the case of a plastic that was primarily used for agriculture and then recycled, the user of the recycled plastic would be allocated some nitrate burden from agriculture production, even though that had little to do with the primary plastic production itself. Due to these potential problems with allocation at the point of substitution, we currently recommend using the cut-off model. Before being a recommendable approach, we believe the allocation at the point of substitution should be preceded by a partial process separation. For example, in the recycled plastic example, only plastic-related processes (upstream plastic manufacturing) should be partially allocated to the recycled material, leaving emissions directly related to the agriculture practice entirely allocated to agriculture.

4.3.2.5 Tips for Using Ecoinvent Database

Some tips follow for selecting the appropriate energy and transportation data in the ecoinvent database.

For the electricity mix, it is important to choose the geographical region appropriate to the case being studied, which is either the consumption mix of the country of production or the electricity type that effectively responds to a marginal increase in electricity demand. For example, an increase in Swiss electricity consumption may have little effect on Swiss production, but may instead be satisfied by an increase in electricity production elsewhere in Europe, such as a thermal gas power plant. We also determine the right voltage level among the three existing types (low, medium, and high voltage). The medium voltage corresponds to industrial use, and low voltage to domestic use, commerce, and agriculture.

Data in the energy sector can be provided in three forms: (a) by mass or volume of combustible, (b) by megajoules of final energy, or (c) by megajoules of useful energy.

(a) The emissions and extractions listed by quantity of combustible (liters of oil, kilograms of petroleum, cubic meters of natural gas, or kilograms of wood) account for associated transport and distribution to users (industrial, commercial, agricultural, and domestic), but do not account for combustion. Emissions associated with the combustion of energy carriers need to be added separately, accounting for a separate process corresponding to the combustion type. The natural gas inventory data are provided for both low- and high-pressure networks, where "low pressure" generally corresponds to domestic, commercial, or agricultural consumption, and "high pressure" refers to industrial consumption.

(b) The processes expressed in megajoules of final energy—the energy bought by the client—account for the combustion and are named in a similar manner to combustibles, with the energy carrier's name followed by the term *burned*.

(c) The inventory data reported by megajoules of useful energy describe the supply of useful heat, such as the heat delivered inside a building. These processes also account for the combustion stage, and have names starting with *heat* followed by the energy carrier's name.

Regarding truck transportation, ecoinvent provides average data assuming an empty truck on the return trip (process names beginning with *transport*), but data are also available for fully loaded and empty trucks (process names beginning with *operation*). In this case, data must be added on vehicle production and disposal, as well as on traffic infrastructure.

Given the array of processes and potential for error in using the ecoinvent database, we advise using specialized LCA software (Section 6.7) to analyze and aggregate ecoinvent data. It is still recommended, however, to examine the raw ecoinvent data and estimate a simple CO_2 or energy balance by hand to check the results of software and identify potential problems.

4.3.3 Data Quality and Uncertainties

Once a relevant data set is found in ecoinvent, it cannot necessarily simply be applied. As discussed in the previous subsection in terms of energy and transportation data, the data set characteristics must be compared with the conditions and objectives of the study under consideration. For the ecoinvent database, such information is available in the form of metadata associated with the different emission and extraction factors. This information addresses:

- Geographical and temporal scope, which is the region and period of time for which the data is valid
- System boundaries covered by the data (e.g., "cradle to grave" or "cradle to factory doors")
- Data format (aggregated, averaged, or as a range of values)
- Data quality and gaps, along with documentation on quality control (comparison with other data sources, mass balance check, etc.)
- Data sources (literature or on-site measurements)

In addition to ensuring proper application of a data set to a given study, this metadata helps indicate uncertainty in data set factors. Few life cycle assessments have quantitatively estimated the uncertainty in inventory data, but there is increasing effort to generally quantify the uncertainty of LCA results. This will help analysts determine if the difference between two scenarios is significant or within the range of uncertainties. It is thus highly recommended to quantify the uncertainties. Section 6.5 describes how probability distributions are determined and parameterized for most ecoinvent inventory data. Section 6.6 also describes how to use these data to evaluate the uncertainty of each scenario studied.

In the case of ecoinvent data, the metadata gives an indication of the estimated probability distribution for each individual datum, and the uncertainties are quantified for all the inputs and outputs of the unit processes. These individual uncertainties are then combined for each FU with the help of statistical methods (such as Monte Carlo, as discussed in Section 6.6.4). We thus obtain the global uncertainty of a given product's inventory (Frischknecht and Jungbluth (2003) provide further details).

4.4 INPUT–OUTPUT APPROACH FOR EXTRACTIONS AND EMISSIONS INVENTORY

In the previous sections, the inventory is based on the processes thought to have a significant environmental impact. Thus, some processes are not included in the system boundary because they are believed to have a negligible contribution, which is often difficult to assess beforehand, particularly in the service industry. For example, what are the impacts associated with the use of hotels, or legal and banking services? The I/O approach (Miller and Blair 1985) described in the next subsection provides a way to estimate all associated extractions and emissions inventories for a given industry, rather than only those from the processes specifically accounted for. Instead of the physical flows in process LCA, the I/O method is based on monetary flows induced in the different economic sectors involved in the supply chain of a product, process, or activity. The expenses of each economic sector are then linked to energy consumption, extraction of resources, and pollutant emissions per monetary unit.

This top-down approach accounts for the entire economy of a country or region by deconstructing it into sectors and products. Taking advantage of detailed national economic statistics describing expenditures by sector, it is possible to exhaustively describe the chain of suppliers needed for a given service. The emission data per monetary unit spent in each sector is then estimated by dividing the total emissions of a sector by its financial output. Table 4.8 provides such data for various U.S. services.

4.4.1 Input–Output Calculations

The principle of the I/O method calculation is analogous to that of the process-based matrix calculation (Section 4.2.4), using an economic *I/O matrix* $\tilde{\mathbf{A}}$ (Equation 4.6) and an I/O so-called "satellite environmental matrix" $\tilde{\mathrm{B}}$ (Equation 4.7):

TABLE 4.8
Energy Use and Emissions Per Dollar Spent in Various U.S. Service Sectors, Accounting for the Preceding Supply Chain

Type of Service Sector	Primary Energy (MJ/$)	CO_2 Emissions (kg/$)	NO_x Emissions (g/$)
Banks	2.27	0.15	4.0×10^{-4}
Insurance	1.95	0.13	3.3×10^{-4}
Hotels	7.71	0.52	1.9×10^{-3}
Management and public relations	2.71	0.14	4.1×10^{-4}
Research and development	4.04	0.19	6.4×10^{-4}
Advertising	4.04	0.29	9.2×10^{-4}
Legal services	1.74	0.16	2.8×10^{-4}
Catering	7.30	0.47	1.5×10^{-3}

Source: Norris, G., *Guide to Using LCNetBaseTM*, Sylvatica, 1999. Based on the LCA netbase.

$$\tilde{\mathbf{A}} = \begin{pmatrix} \tilde{a}_{11} & \cdots & \tilde{a}_{1m} \\ \vdots & \ddots & \vdots \\ \tilde{a}_{m1} & \cdots & \tilde{a}_{mm} \end{pmatrix} \tag{4.6}$$

$$\tilde{\mathbf{B}} = \begin{pmatrix} \tilde{b}_{11} & \cdots & \tilde{b}_{1m} \\ \vdots & \ddots & \vdots \\ \tilde{b}_{n1} & \cdots & \tilde{b}_{nm} \end{pmatrix} \tag{4.7}$$

The element $\tilde{a}_{ij}$, also called the *input/output coefficient*, represents the monetary output from sector i required to produce \$1 of output from sector j (e.g., the dollar amount the aluminum sector spends in the electricity sector per dollar output of aluminum). The element $\tilde{b}_{ki}$ is the elementary flow k directly extracted from or emitted to the environment per monetary unit of sector i, referred to here as the *emission factor* (e.g., kg_{CO2} per dollar output in the electricity sector).

Similarly to the technology demand vector **y** of the process-based matrix calculations, I/O LCA uses the economic *demand vector* $\tilde{\mathbf{y}}$, where $\tilde{\mathbf{y}}_j$ represents the amount spent in sector j for providing one FU (in \$/FU). A direct purchase from sector j requires indirect purchases from the sectors that serve j, calculated by multiplying the I/O matrix $\tilde{\mathbf{A}}$ by $\tilde{\mathbf{y}}$. To account for the entire supply chain, we need to add this to the output due to second-tier suppliers, $\tilde{\mathbf{A}}^2\tilde{\mathbf{y}}$, and all preceding suppliers, yielding an infinite sum analogous to that of Section 4.2.4 (Equation 4.8):

$$\tilde{x} = (I + \tilde{A} + \tilde{A}^2 + \tilde{A}^3 + \ldots)\tilde{y} = (I - \tilde{A})^{-1}\tilde{y} \tag{4.8}$$

where **I** is the identity matrix, and the total output vector $\tilde{\mathbf{x}}$ is the total amount of goods and services in each sector needed to meet the demand $\tilde{\mathbf{y}}$.

The environmental matrix $\tilde{\mathbf{B}}$ multiplies the total output $\tilde{\mathbf{x}}$ to yield the quantities of emitted substances and extracted resources ($\tilde{\mathbf{u}}$) corresponding to the demand (e.g., in kg_{CO2}/FU, Equation 4.9).

$$\tilde{\mathbf{u}} = \tilde{\mathbf{B}}\,\tilde{\mathbf{x}} = \tilde{\mathbf{B}}\,(\mathbf{I} - \tilde{\mathbf{A}})^{-1}\,\tilde{\mathbf{y}} = \tilde{\mathbf{E}}\,\tilde{\mathbf{y}} \tag{4.9}$$

where $\tilde{\mathbf{E}} = \tilde{\mathbf{B}}\,(\mathbf{I} - \tilde{\mathbf{A}})^{-1}$ is the matrix of environmental emissions and resource extractions from each economic sector over the entire production chain. The elements of $\tilde{\mathbf{E}}$ are expressed as $\tilde{e}_{kj}$: the total elementary flow k extracted from or emitted into the environment per monetary demand of sector j (e.g., in kg_{CO2}/\$).

4.4.2 I/O Database

Calculating emissions with the I/O method requires two types of data: the expenses of each sector in every other economic sector ($\tilde{A}$), and the emission factors per dollar for each sector and pollutant ($\tilde{\mathbf{B}}$). These data are generally calculated using national statistics, as detailed further below.

4.4.2.1 Determining Economic I/O Matrix

The key advantage of the I/O approach is its use of national economic statistics to systematically determine the use of goods and services among different industries. Figure 4.5 shows a simplified national transactions matrix **Z**, which represents the total expenses of each sector in every other sector, and is available to varying degrees for most countries. The entry in column *j* and row *i* represents the expenditure of industry *j* in sector *i* to produce the total output of sector *j*. If Industry 2 corresponds to the aluminum sector and Industry 4 corresponds to the electricity sector, Figure 4.5 shows that the aluminum sector spends \$3.33 billion in the electricity sector to produce \$11.5 billion of output. The difference between its sum of intermediary expenses (\$8.17 billion) and its total industrial output (\$11.5 billion) is \$3.3 billion worth of added value, used for such payments as salaries and benefits.

For the national transactions matrix to be applied generically to a given amount of spending in a sector, it needs to be normalized to express the amount a given sector spends in each sector for a dollar of output. Each element $\tilde{a}_{ij}$ in the normalized economic matrix $\tilde{\mathbf{A}}$ is obtained by dividing each element z_{ij} in the national transactions matrix **Z** by the total output $\tilde{x}_j$ of that column's sector. In the aluminum and electricity example, we find that the aluminum sector spends \$0.29 on electricity per dollar of aluminum produced.

The sum of elements in each sector's row of the transaction matrix is the total sector production that is used by other industrial sectors (\$11.0 billion for Industry 2). By definition, the product of $\tilde{\mathbf{A}}$ with the total industrial output vector $\tilde{\mathbf{x}}$ gives this total industrial use. Since the economic system is closed, the total industrial output (\$11.5 billion) is the sum of the industrial use with the final demand $\tilde{\mathbf{y}}$ of consumer and government spending in each industry (\$0.5 billion for Industry 2). In terms of matrices, this is expressed as $\tilde{\mathbf{A}}\,\tilde{\mathbf{x}} + \tilde{\mathbf{y}} = \tilde{\mathbf{x}}$. By solving for **x**, we redefine the fundamental equation of the I/O approach: $\tilde{\mathbf{x}} = (\mathbf{I} - \tilde{\mathbf{A}})^{-1}\tilde{\mathbf{y}}$.

National economic transaction matrices are available for most countries, with varying levels of sector and time resolution, in ways that do not necessarily correlate with LCA data availability. Switzerland has relatively detailed and comprehensive data on LCA processes, but possesses an economic matrix of only 42 sectors (Nathani et al. 2006). The United States, on the other hand, is relatively poor in LCA process data, but differentiates 500 sectors in its economic matrix. Several *multiregional input–output* (MRIO) approaches have also compiled economic I/O matrices in a consistent manner for a majority of countries of the world (Section 4.4.2.4).

4.4.2.2 Determining I/O Environmental Matrix

Once the final demand is combined with the economic matrix to yield the industrial output (Equation 4.8), this must be multiplied by the environmental impacts per dollar output. Databases provide pollutant emissions and energy and resource use for each sector of the economy. So, each emission factor is calculated by dividing the total emissions for each sector by that sector's total output. Figure 4.6 compares the direct emissions of CO_2 per dollar spent in each U.S. economic sector for the Open-LC (Norris 1999) and Comprehensive Environmental Data Archive (CEDA) (Suh et al. 2004; Suh 2005) databases. The correspondence between the

Supplying Industries	Using Industries						Industrial Use		Consumers & Govt.		
Transaction Matrix **Z**	Ind. 1	Ind. 2	Ind. 3	Ind. 4	Ind. 5	Ind. 6	Total intermediate input	+	Final demand	=	Total industrial output
	(M$)	(M$)	(M$)	(M$)	(M$)	(M$)	(M$)		(M$)		(M$)
Industry 1	1400	200	0	575	0	1420	3595		6679		10274
Industry 2	2200	1700	0	128	270	6678	10976		494		11470
Industry 3	3425	0	333	0	120	0	3878		11005		14883
Industry 4	0	3330	4450	0	0	0	7780		6493		14273
Industry 5	159	2850	4920	5870	600	0	14399		391		14790
Industry 6	890	90	780	2200	7200	770	11930		4638		16568
Total intermediate input	8074	8170	10483	8773	8190	8868	$\tilde{\mathbf{A}}\boldsymbol{x}$	+	$\boldsymbol{y}$	=	$\boldsymbol{x}$
+											
Value added	2200	3300	4400	5500	6600	7700					
=											
Total industrial output	10274	11470	14883	14273	14790	16568					

	Ind. 1	Ind. 2	Ind. 3	Ind. 4	Ind. 5	Ind. 6
Economic matrix $\tilde{\mathbf{A}}$	($/$)	($/$)	($/$)	($/$)	($/$)	($/$)
Industry 1	0.136	0.017	0	0.040	0	0.086
Industry 2	0.214	0.148	0	0.009	0.018	0.403
Industry 3	0.333	0	0.022	0	0.008	0
Industry 4	0	0.290	0.299	0	0	0
Industry 5	0.015	0.248	0.331	0.411	0.041	0
Industry 6	0.087	0.008	0.052	0.154	0.487	0.046

FIGURE 4.5 Sample calculation of economic I/O matrix from the national transaction matrix. (Adapted from Norris, G., personal communication. With permission.)

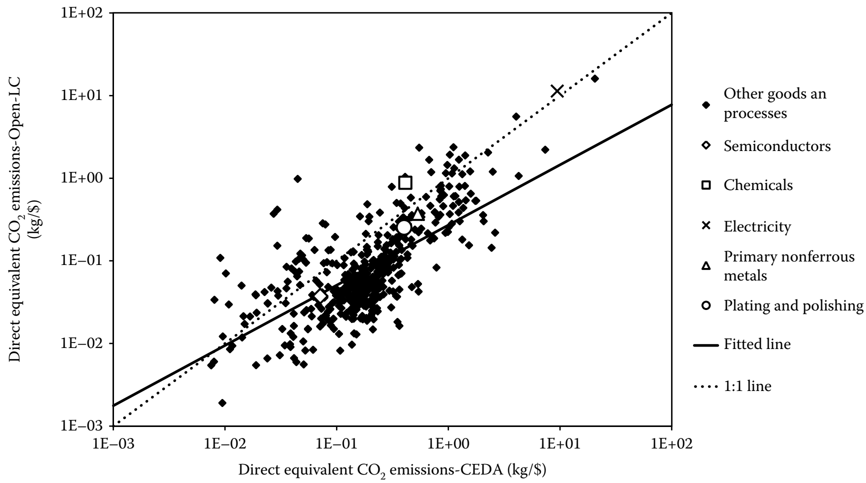


FIGURE 4.6 Comparison of direct CO_2 emissions for the American economic sectors, as modeled by Open-LC and CEDA. (Adapted from Loerincik, Y., *Environmental Impacts and Benefits of Information and Communication Technology Infrastructure and Services, Using Process and Input–Output Life Cycle Assessment*, PhD thesis, École polytechnique fédérale de Lausanne (EPFL), n° 3540, 2006, doi:10.5075/epfl-thesis-3540. With permission.)

two databases is limited, with the Open-LC emission factors smaller than the CEDA factors for the majority of sectors. However, the sectors with the highest emissions per dollar have more consistent factors between the two databases, particularly the electricity sector.

Figure 4.7 plots the same comparison, but accounting for emissions over the entire supply chain, resulting in much better correspondence between the two. This is because the cumulative coefficients are dominated by sectors with high emission levels, which have the best correspondence of direct emissions.

4.4.2.3 I/O Country-Specific Databases

Some software programs and databases directly provide the emissions or impacts associated with expenses in different sectors (see Appendix I for websites from which the information in this subsection was extracted and to access more detailed information).

For the United States, several databases extend the Bureau of Economic Analysis (BEA) economic I/O matrix for environmental assessment and LCA applications. Carnegie Mellon University provides the "Economic Input–Output Life Cycle Assessment (EIO-LCA) method to estimate the materials and energy resources required for, and the environmental emissions resulting from, activities in the US economy, with summary results for various environmental impacts" (Appendix I). The MIET freeware software, developed at Leiden University, provides an I/O database for the United States in 2002, with 1170 environmental inventory flows. CEDA is a "suite of environmentally extended input–output

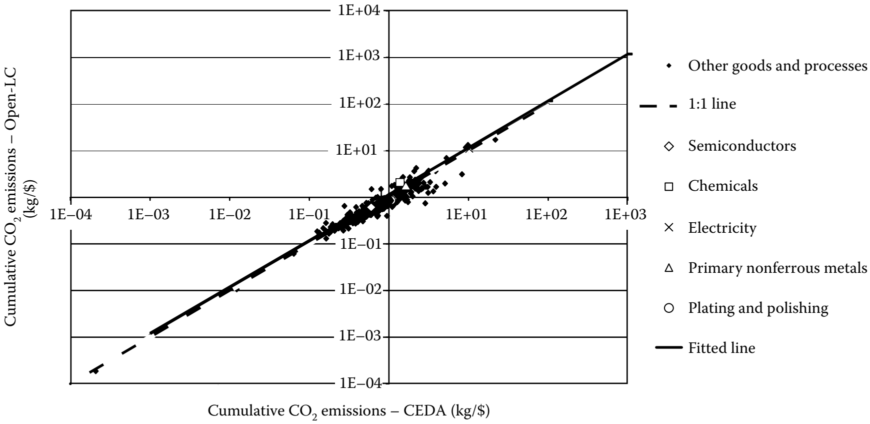


FIGURE 4.7 Comparison of cumulative CO_2 emissions for the American economic sectors, as modeled by Open-LC and CEDA. (Adapted from Loerincik, Y., *Environmental Impacts and Benefits of Information and Communication Technology Infrastructure and Services, Using Process and Input–Output Life Cycle Assessment*, PhD thesis, École polytechnique fédérale de Lausanne (EPFL), n° 3540, 2006, doi:10.5075/epfl-thesis-3540. With permission.)

databases that covers a comprehensive list of over 1500 environmental interventions including fossil fuels, water, metals ores and minerals, and various emissions to air, water and soil" (Appendix I; North American I/O CEDA database described in Suh et al. 2004; Suh 2005). Integrated into the SimaPro software, it consists of a commodity matrix from 2002, supplemented with data for capital goods. The I/O matrix is linked to a large environmental intervention matrix compiled from several data sources. By using the databases mentioned, the impacts of small and medium enterprises (SME) have been added to the environmental intervention matrix, along with those from diffuse sources such as transport.

In Asia, the Japanese I/O database (Nansai et al. 2012) was developed by the Environmental Technology Laboratory of the Corporate Research & Development Center of Toshiba Corporation. It utilizes the Japanese I/O table from the year 2000, as published in 2004, and it contains approximately 400 domestic industrial sectors in Japan exclusively.

On the European level, Danish and Dutch institutions have also created databases that have been integrated into the SimaPro LCA software. The I/O database for EU27 and Denmark (2003) is based on statistical data from 1999, which have been modified and improved by 2.0 LCA Consultants to make the I/O data more relevant to LCA applications. It is based on the hybrid I/O model FORWAST that provides complete balanced monetary and physical supply-use tables. The Dutch Input Output 95 library is based on a survey of average consumer spending in 350 categories grouped by economic sector (Goedkoop 2004). The I/O table was extended to also account for imports from both Organisation for Economic Co-operation and Development (OECD) and non-OECD regions. CML-LCA also allows a combination with the

E3IOT database (see Appendix I), which distinguishes approximately 500 production sectors.

4.4.2.4 I/O Multiregional Databases

Several methods have combined national I/O matrices into MRIO databases covering the whole world. Developed at the University of Sydney, the EORA MRIO (Lenzen et al. 2013) provides the highest level of detail, with data for 187 individual countries comprising more than 15,000 industry sectors. It has been applied to carbon, water, ecological footprinting (Lenzen et al. 2013), and employment, with a detailed study of uncertainties.

The EXIOPIOL project has developed a new series of European matrices directly coupled to environmental data, and then extended the system to a global scale to create the global commercial EXIOBASE (see Appendix I). Based on the year 2000, it covers 43 countries (95% of the global economy) and distinguishes 129 industry sectors and products by country, covering 30 emitted substances and 80 resources by industry. Peters and Hertwich (2008) used this data to determine the influence of global trade on CO_2 emissions.

The WIOD database includes 40 countries and a model for the rest of the world (see Appendix I). It includes data on employment capital stocks, gross output, and value, as well as on energy use, CO_2 emissions, and emissions to air at the industry level (Timmer 2012).

Finally, the tracking environmental impacts of consumption (TREIC) project (Friot 2009) combined matrices from the Global Trade Analysis Project with the emissions database for global atmospheric research (EDGAR) and an impact assessment model to evaluate the public health impacts associated with global consumption. Using the matrices to link a consuming region to the supply chain regions of production and emission, the TREIC project estimates health impacts due to each consuming region of the world, both locally and in the other regions.

4.4.3 Example of Input–Output LCA: Aluminum Front-End Panel of Automobile

4.4.3.1 Functional Unit, Reference Flow, and Final Demand

Let us revisit the example from Section 4.2.3 of an aluminum front-end panel of a car to illustrate a simplified I/O application, considering the same FU of a front-end panel with a given rigidity, transported over a distance of 200,000 km. Table 4.9 presents the major reference flows for the aluminum scenario, which by definition represent what is bought to achieve the FU (Section 3.3). It is thus straightforward to associate a price to each flow and to determine the final monetary demand ($\tilde{\mathbf{y}}$) by FU. This demand by sector will be the basis for calculating the inventory of emissions and resource extractions.

Our goal is to calculate the inventory of emissions and extractions using I/O factors rather than process-based factors. Equation 4.9 is used to calculate this inventory, using the relevant economic and environmental matrices that characterize the interactions among economic sectors and the associated emissions and extractions.

TABLE 4.9
Quantity and Price Data for Aluminum Front-End Panel of a Car

Purchased Good	Amount per FU	Price	Final Demand per FU for Each Good	Sector	Final Demand per FU for Each Sector
Aluminum	3.8 kg/panel	\$2.5/kg	**\$9.50**	Aluminum	**\$9.50**
Electricity for manufacturing	15.2 kWh/panel	\$0.07/kWh	**\$1.06**	Electricity	**\$1.06**
Oil for manufacturing	1.8 L/panel	\$0.32/L	**\$0.58**	Coal and petroleum	**\$11.58**
Gasoline consumed during use	30.4 L/panel	\$0.36/L	**\$11**		

4.4.3.2 Economic Data and Determination of the I/O Economic Matrix

As described in the previous section, each element of the I/O economic matrix $\tilde{\mathbf{A}}$ is calculated by dividing each entry in the relevant national transaction matrix $\mathbf{Z}$ by the total industrial output $\tilde{\mathbf{x}}$ of each column's industry (Table 4.10). $\mathbf{Z}$ represents the expenses of each sector in every other sector, so \$13,240 million is the amount spent by the electricity sector in the coal and petroleum sector. The total economic output of each sector is represented by $\tilde{\mathbf{x}}$, so \$132,400 million is the total amount spent by the electricity sector in every other sector, plus the value added by the sector itself.

The complete matrix $\mathbf{Z}$ reveals the principal supplying and consuming sectors for every other sector. In this simplified example, the dominant supplier to the aluminum sector is the electricity sector, providing \$1518 million of electricity. The largest consumer of the aluminum sector, out of the three industries considered here, is the aluminum sector itself.

The matrix $\tilde{\mathbf{A}}$ expresses intersector spending as a fraction of the total monetary output of a given sector, thus each $\tilde{a}_{ij}=z_{ij}/\tilde{x}_j$ (the expenses of sector j in sector i divided by the total output of sector j).

TABLE 4.10
Simplified Transaction Matrix Z and Total Industrial Output $\tilde{\mathbf{x}}$ for Three Economic Sectors

Transaction Matrix Z	Aluminum	Coal and Petroleum	Electricity	Total Output $\tilde{x}$
Aluminum	976	0	0	5,688
Coal and petroleum	0.50	5,877	13,240	109,680
Electricity	1,518	1,243	27	132,400

Source: U.S. Bureau of Economic Analysis (see website listed in Appendix I).
Note: Values are in millions of dollars.

$$\tilde{\mathbf{A}} = \begin{pmatrix} \dfrac{976}{5688} & \dfrac{0}{109680} & \dfrac{0}{132400} \\ \dfrac{0.50}{5688} & \dfrac{5877}{109680} & \dfrac{13240}{132400} \\ \dfrac{1518}{5688} & \dfrac{1243}{109680} & \dfrac{27}{132400} \end{pmatrix} = \begin{pmatrix} 0.17 & 0 & 0 \\ 0.000088 & 0.054 & 0.1 \\ 0.27 & 0.011 & 0.0002 \end{pmatrix}$$

The term 0.27, for example, means that for \$1 of aluminum produced, the aluminum sector has spent \$0.27 in the electricity sector.

4.4.3.3 Environmental Data and Determination of the Environmental Matrix

Each element of the environmental matrix $\tilde{\mathbf{B}}$ is calculated by dividing the direct environmental emissions or extractions of a sector (Table 4.11) by the total output $\tilde{\mathbf{x}}$ of that sector (Table 4.10).

$$\tilde{\mathbf{B}} = \begin{pmatrix} \dfrac{0}{5688} & \dfrac{6.26\times10^{13}}{109680} & \dfrac{0}{132400} \\ \dfrac{1.1\times10^{9}}{5688} & \dfrac{76\times10^{9}}{109680} & \dfrac{1.5\times10^{12}}{132400} \end{pmatrix} = \begin{pmatrix} 0 & 571 & 0 \\ 0.19 & 0.69 & 11.3 \end{pmatrix}$$

The first row of $\tilde{\mathbf{B}}$ lists the direct extraction of nonrenewable energy per amount spent in each sector (MJ/\$M, where \$M is millions of dollars), and the second row contains the direct CO_2 emissions per amount spent in each sector (kg_{CO2}/\$M). It may seem surprising that the aluminum and electricity sectors have zero primary energy consumption per dollar. This is because rather than directly extract nonrenewable energy sources from the environment, these two sectors buy energy from another sector. In practice, most of the nonrenewable energy extraction is done by the "coal and petroleum" sector, with the extraction of uranium often not considered. CO_2 emissions, on the other hand, occur in all sectors considered above, since almost any sector that uses energy directly emits CO_2.

TABLE 4.11
Direct Extraction of Nonrenewable Primary Energy from the Environment and Direct CO_2 Emissions, by Sector and by Year

Sector	Direct Extraction of Nonrenewable Primary Energy (MJ/year)	Direct CO_2 Emissions (kg/year)
Aluminum	0	1.1×10^9
Coal and petroleum	6.26×10^{13}	7.6×10^{10}
Electricity	0	1.5×10^{12}

4.4.3.4 Calculation of Total Monetary Output per Functional Unit

As described in Subsection 4.4.3.1, the total monetary output $\tilde{\mathbf{x}}$ per FU is the amount of money spent by each industry in all sectors to meet the demand $\tilde{\mathbf{y}}$ of one FU. It is calculated (Equation 4.8) by combining $\tilde{\mathbf{y}}$ with $(\mathbf{I}-\tilde{\mathrm{A}})^{-1}$, the associated indirect demand from all other sectors. In this example,

$$\mathbf{I}-\tilde{\mathbf{A}} = \begin{pmatrix} 0.83 & 0 & 0 \\ -0.000088 & 0.95 & -0.1 \\ -0.27 & -0.011 & 1.0 \end{pmatrix}$$

$$\left(\mathbf{I}-\tilde{\mathbf{A}}\right)^{-1} = \begin{pmatrix} 1.2 & 0 & 0 \\ 0.034 & 1.1 & 0.11 \\ 0.32 & 0.012 & 1.00 \end{pmatrix}$$

The lower left value of 0.32 in the $(\mathbf{I}-\tilde{\mathrm{A}})^{-1}$ matrix indicates that every dollar of demand in the aluminum sector induces \$0.32 spent in the electricity sector when accounting for the entire supply chain. This also induces \$0.034 spent in the coal and petroleum sector and \$1.2 spent within its own sector over the whole supply chain.

The total output per FU for a front-end panel is thus calculated as follows:

$$\tilde{\mathrm{x}} = (\mathrm{I}-\tilde{\mathrm{A}})^{-1}\tilde{\mathrm{y}} = \begin{pmatrix} 1.2 & 0.0 & 0.0 \\ 0.034 & 1.1 & 0.11 \\ 0.32 & 0.012 & 1.0 \end{pmatrix} \times \begin{pmatrix} 9.5 \\ 11.58 \\ 1.06 \end{pmatrix} = \begin{pmatrix} 11.4 \\ 12.7 \\ 4.3 \end{pmatrix}$$

The front-end panel thus induces \$11.4 worth of goods or services produced by the aluminum sector, \$12.7 by the coal and petroleum sector, and \$4.3 by the electricity sector.

4.4.3.5 Primary Energy and CO_2 Emissions per Functional Unit over the Supply Chain of Front-End Panel and Gasoline

To reach an inventory of emissions and extractions over the FU, we multiply the output of each sector over the FU by the emissions and extractions per dollar in each sector. The environmental matrix $\tilde{\mathbf{B}}$ multiplies by the total output $\tilde{\mathbf{x}}$ to yield $\tilde{\mathbf{u}}$, the life cycle environmental emissions and resource extractions per FU (Equation 4.9):

$$\tilde{\mathrm{u}} = \tilde{\mathrm{B}}\,(\mathrm{I}-\tilde{\mathrm{A}})^{-1}\tilde{\mathrm{y}} = \tilde{\mathrm{B}}\,\tilde{\mathrm{x}} = \begin{pmatrix} 0.0 & 571 & 0.0 \\ 0.19 & 0.69 & 11.3 \end{pmatrix} \times \begin{pmatrix} 11.4 \\ 12.7 \\ 4.3 \end{pmatrix} = \begin{pmatrix} 7250 \\ 59.5 \end{pmatrix}$$

According to these calculations, the manufacturing and use of this front-end panel (including gasoline) requires 7520 MJ of primary energy and results in 59.5 kg_{CO2} emitted over the whole supply chain.

To have a general matrix that can be applied to any final demand, we calculate the $\tilde{\mathbf{E}}$ of the cumulative pollutant emissions and resource extractions over the whole supply chain per amount of demand (Equation 4.9):

$$\tilde{E} = \tilde{B}(I - \tilde{A})^{-1} = \begin{pmatrix} 0.0 & 571 & 0.0 \\ 0.19 & 0.69 & 11.3 \end{pmatrix} \times \begin{pmatrix} 1.2 & 0.0 & 0.0 \\ 0.034 & 1.1 & 0.11 \\ 0.32 & 0.012 & 1.00 \end{pmatrix}$$

$$= \begin{pmatrix} 19.7 & 604 & 60.4 \\ 3.9 & 0.87 & 11.4 \end{pmatrix}$$

As described previously, the vector $\tilde{\mathbf{u}}$ can also be calculated by directly multiplying $\tilde{\mathbf{E}}$ by the final demand per FU y.

4.4.3.6 CO_2 Emissions during Usage Stage

The I/O calculation in the previous subsection accounts for the supply chain of the aluminum front-end panel and consumed gasoline, but does not include the direct emissions during the use of the car. The combustion of 30.4 L of gasoline, with 2.32 kg_{CO2} emission per liter (Table 4.1), results in 70.5 kg of CO_2 emitted during the use stage. Adding this to the 59.5 kg calculated in the previous subsection yields 130 kg of CO_2 emitted over the manufacturing and use of a front-end panel on a car that travels 200,000 km.

4.4.3.7 Comparison with Process LCA

According to results from a process-based life cycle assessment (Section 4.2.3), an aluminum front-end panel uses 2193 MJ of nonrenewable primary energy and emits 135 kg_{CO2} over its life cycle (Table 4.6). Compared with the I/O result of 130 kg_{CO2}, the two approaches appear to give similar results.

On the other hand, in this very simplified example, the nonrenewable primary energy consumption calculated by the I/O approach is over three times that of the process-based approach. Such differences are not uncommon, and it is therefore important to be aware of differences in the approaches and to carefully verify the compatibility of I/O and process data if they are ever combined in a single study.

4.4.3.8 Analysis of Impacts by Supply Chain Tier

An advantage of the I/O approach is the ability to separately consider the matrix for each tier of the supply chain, and thus analyze which tier results in the dominant contributions to emissions and resource use—do greater impacts result from the final production step or throughout the supply chain?

Figure 4.8 presents the cumulative contributions of energy use in each supply chain tier for the aluminum front-end panel. The three tiers closest to final production are responsible for 90% of the energy used in manufacturing. The third tier is responsible for the largest energy consumption, since the aluminum sector (first tier) spends a lot in the electricity sector (second tier), which itself spends a lot in the coal and petroleum sector (third tier), which is directly responsible for substantial energy consumption.

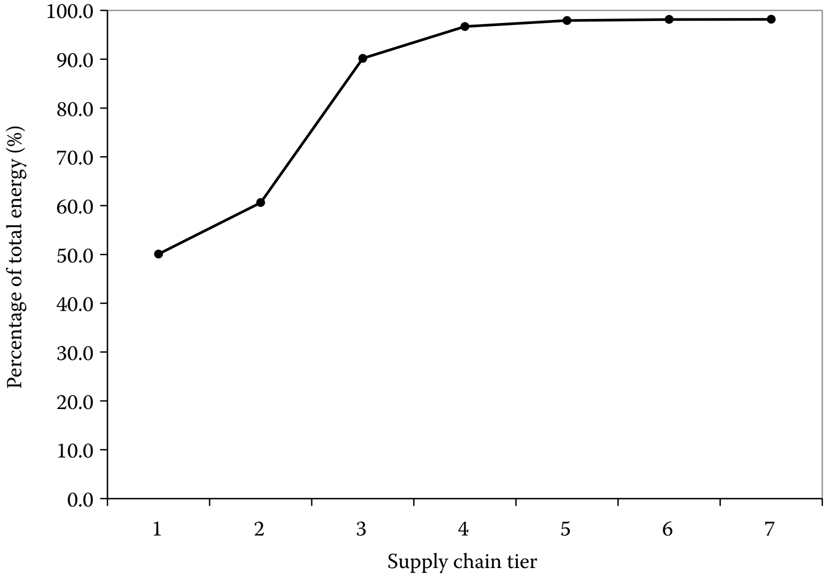


FIGURE 4.8 Cumulative contribution of each tier to the total nonrenewable energy consumption in manufacturing an aluminum front-end panel.

4.4.4 Advantages and Limitations of I/O Approach

The I/O approach describes the supply chain with less detail but in a more comprehensive manner than the approach based on specific processes and subprocesses. With I/O, it becomes possible to use the expenses of each sector to infinitely climb up the suppliers' chain. Moreover, process-based LCA can include some unnecessary subprocesses, while omitting important ones that do not seem relevant to the analyst. In I/O there are no such arbitrary input exclusions, since the inverted matrix accounts for all sectors that interacted in any way to yield the final product, thus including sectors that may have appeared negligible. I/O models are thus very useful in defining environmental management priorities, estimating missing data, and verifying that no important processes have been missed. I/O also allows the extension of the life cycle approach to certain socioeconomic aspects, providing a platform for consistently comparing environmental, social, and economic performances of a product or service (Section 6.8).

The main limitations of I/O stem from the necessary aggregation of subsectors into economic sectors for which data are available. Numerous and varied products and technologies are thus all combined into the same sector and treated as equivalent. Since two products that serve the same function are often produced by the same sectors, it is difficult to use I/O to differentiate such products, and a process-based approach would better identify the differences in emissions and energy use.

Another limitation of the I/O approach is the availability of environmental data to link to the relatively accessible economic data. Emissions data can be difficult to find, and are not necessarily grouped in ways that are compatible with economic sectors. Moreover, I/O data are generally relatively old due to the lag time between data gathering and publication of I/O coefficients, which limits application to emerging technologies. Finally, there can be substantial inhomogeneity in emissions per dollar among products within a given sector, which can over- or underestimate emissions for a product that is far from the sector average.

Finally, it is important to note that the I/O approach generally focuses only on the manufacturing supply chain of a product or service, without covering the use or waste treatment stages. Transportation data is also difficult to use in the I/O approach, because some sectors arrange their own transportation without using professionals categorized in the transportation sectors. For a full assessment, it is crucial that the use, transportation, and waste treatment stages are separately taken into account and added to the I/O results for manufacturing.

4.4.5 Combined Hybrid Use of Process and I/O Approaches

Table 4.12 sums up the pros and cons of the I/O and process-based approaches as discussed by Loerincik (2006).

The I/O and process approaches are, in fact, complementary and can be used in parallel or be combined with various levels of complexity.

4.4.5.1 Level 1: Verification of System Boundaries

A quick I/O calculation can help verify that the choice of system boundaries in a process-based LCA includes all major contributions. For example, an I/O approach can show that the aerial transport sector has a significant contribution to the total

TABLE 4.12
Advantages and Disadvantages of I/O and Process-Based Approaches

	I/O Approach	Process-Based Approach
Advantages	Considers the whole economic system without truncation	Allows comparison of products in the same sector, accounting for various detailed techniques
	Uses readily available country-specific I/O matrix data	Is a powerful tool for analysis of ecodesign alternative solutions
	Has the potential to account for imports and exports by coupling of the national transaction matrices in multiregional I/O matrices	Has databases available for a large number of processes, particularly in Europe
Disadvantages	Cannot distinguish between products from the same sector and therefore cannot differentiate their impacts	Still needs a large amount of inventory data for new and not-yet-available processes, particularly in emerging countries
	Has difficulty in gathering direct emission data by sector	Does not generally include services
	Can have compatibility problems between different geographic regions	Has difficulty including all system processes, leading to truncations

emissions and impacts, but has not yet been included in the process approach. The system boundaries can then be expanded to include the aerial processes in the process-based approach.

4.4.5.2 Level 2: Impacts of Services

I/O can also supplement the process-based approach by adding the impacts linked to relevant service sectors. Loerincik (2006, figure 9) shows that services typically represent between 1% and 70% of the impacts due to each U.S. economic sector. However, services induce less than 5% of the impacts from sectors with the largest cumulative emissions.

4.4.5.3 Level 3: Hybrid Approach

Suh (2002) proposes a stronger integration between I/O and process-based approaches, rather than simply supplementing data from one with the other. Having established a theoretical basis for such an approach, the implementation is still in the research and development stage. This will improve as the linkages between product categories and economic sectors become better defined. The creation of a fully integrated hybrid approach is, however, a challenging task, since tiers in process LCA and I/O do not correspond and a given process can include contributions of multiple economic sectors and vice versa.

4.5 COPRODUCTS AND ALLOCATION

4.5.1 Issues When Multiple Products Are Made by One System

Many agricultural, industrial, and waste treatment processes are multiproduct systems, yet a given life cycle assessment will generally focus on only one of these products. We thus need to allocate environmental emissions and resource use among the products studied and the other coproducts. The way to treat this problem depends on the nature of the products and systems studied.

4.5.2 Product Categories and Allocation

We first distinguish different types of products based on economic value.

4.5.2.1 Coproducts

In addition to the principal product, a given system can generate one or more secondary products that have economic value, but do not correspond to the studied function. For example, the production of meat and milk can occur in parallel on a farm, with resources shared among the two production processes. The simultaneous production of straw and wheat is another example, and one that we will follow throughout this section (Figure 4.9). Which emissions and resource usage should be allocated to wheat and which to straw, in which proportions, and following which criteria? This requires an answer to the following question: Which economic activity is responsible for which environmental problem? Inversely formulated, which environmental problem is caused by which economic activity?

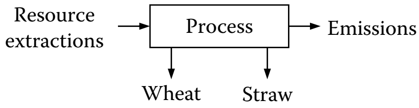


FIGURE 4.9 Example of single process that simultaneously yields two coproducts: straw and wheat.

Economic activity and emissions can be linked by different types of causalities. In some cases, a physical material in one coproduct results in a specific emission that is independent of the other product. Or, the economic value of one product induces a pollution-emitting process that would not otherwise occur. Section 4.5.3 describes different methods for properly allocating emissions and resource use in a system that yields multiple products.

4.5.2.2 Waste (to Be Disposed Of)

Waste refers to what remains that has no positive economic value and requires treatment. Packaging is a good example of waste. Since disposal of waste generates emissions, its treatment and associated emissions must be considered in the system boundary. The waste itself is not considered as an elementary flow to the environment that crosses the system boundary, so waste treatment is considered part of the system.

Because treatment plants simultaneously treat waste flows from many different systems, the emissions associated with the plant must somehow be allocated among the systems. Analogous to the issue of coproduct allocation in the previous subsection, a systematic approach is needed to attribute emissions as those due to waste from System A and those due to waste from System B (Figure 4.10).

4.5.2.3 Recycled Waste and By-Products with Low Economic Value

The distinction between waste and by-product is often not clear, since the economic context helps determine if a product has any value. Recycled paper is a typical example. Depending on the amounts of paper recycled and used, as well as the global price fluctuations, used paper is either bought or someone must pay to get rid of it. Manure is another product with low economic value that can be negative in the case of local overproduction (Figure 4.11).

If the waste by-product can be reused within the studied system, called *closed-loop recycling*, it is easy to account for recycling by reducing the need for raw materials and corresponding emissions.

In the more complicated but frequent case of *open-loop recycling*, the recycled by-product is used outside the system being studied. The emissions during the recycling,

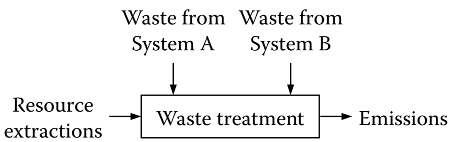


FIGURE 4.10 Example of allocation for waste treatment.

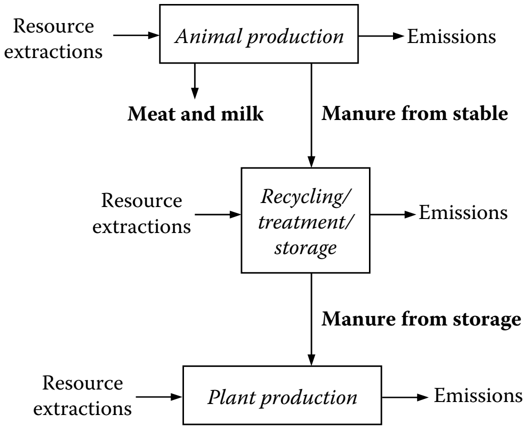


FIGURE 4.11 Allocation of by-products with low economic value. Manure can be either a by-product or waste, and can be considered a part of multiple systems.

treatment, and storage of the by-product can be allocated either to the system that created the by-product or the system that will use the recycled product. Ways to resolve this problem are described in Section 4.5.5.

In addition to explicitly performed allocation, allocation may also be implicitly performed, such as in the case of a combined transport where two products are simultaneously transported (e.g., a ton-kilometer is an implicit allocation by weight) or for tractors used for multiple crops (e.g., a tractor-hour is an implicit allocation based on time, in which the tractor resource use is allocated by dividing the number of tractor-hours for the considered crop by the number of tractor-hours over its lifetime).

4.5.3 Allocation Methods for Coproducts

It is first important to point out that the problems described in the previous section can be totally avoided by modifying the objectives of the study to calculate the impacts of simultaneous production of all resulting coproducts. This involves enlarging the system boundary to include all coproducts. In the example depicted in Figure 4.11, the study can be expanded to evaluate the global impact of agricultural production of milk, meat, and grains. In such a case, it is no longer necessary to attribute the emissions of animal production to either meat, milk, or manure used in wheat production. But, since most LCAs focus on a single FU and, therefore, only on one product, we must somehow allocate impacts from a multiproduct system to a unique product.

The LCA norms from the International Organization for Standardization (ISO 14044, p. 14) define a hierarchy of allocation methods depending on the ISO denomination.

4.5.3.1 Allocation Procedure from ISO 14044

1. Step 1: Wherever possible, allocation should be avoided by

a. Dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or
b. Expanding the product system to include the additional functions related to the co-products.

2. Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.
3. Step 3: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products. (ISO 2009)

4. Step 4: As made explicit in the greenhouse gas protocol (ISO 2009), factors such as mass, energy, and volume that are selected using value choices or arbitrary assumptions are the least preferred basis for allocation decisions.

Each of these options is described in more detail in the following subsections, and illustrated using the example of the simultaneous production of straw and wheat (Figure 4.12; case study developed from Audsley et al. 1997). In calculating the resources used for wheat, we also need to account for the straw produced along the way. How do we allocate the used resources between the two processes? After answering this question, we then briefly examine how other coproduct problems can be solved.

4.5.3.2 (a) Avoiding Allocation

Wherever possible, allocation should be avoided by either process subdivision or system expansion.

4.5.3.3 Process Subdivision

Even for a system that results in multiple products, certain subprocesses of the system may only be relevant to one of the coproducts. In such cases, allocation may be

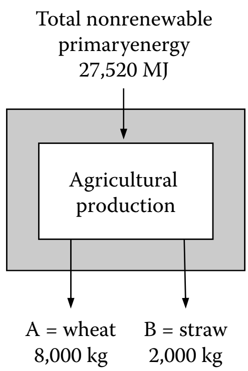


FIGURE 4.12 Coproduction of wheat and straw.

avoided by increasing the level of detail. Any subprocess that does not directly apply to the main product should be excluded from the system boundary.

Applying this principle to the case of straw and wheat production does not entirely solve the problem of allocation, but helps separate some components. The processes of energy production, fertilizer and pesticide production, and mechanical soil work and treatment are all common to the two coproducts and cannot be exclusively allocated to either the straw or the wheat. On the other hand, the creation and transport of hay bales are processes only relevant to straw production. These subprocesses should therefore be excluded from the system boundary when we only focus on the production of wheat in order to make bread. The emissions and raw materials used for these subprocesses should not be considered in the wheat balance. Thoma et al. (2013) present a variant of the "process subdivision" approach applied to allocation between meat and milk in dairy production; they separate the calculation of the feed fraction necessary to animal growth and meat production from the feed fraction required for milk production.

4.5.3.4 System Expansion

If the coproduct B of a system has any value, it can be considered to replace a product B′ that would otherwise get used. When B′ is replaced, its associated emissions are avoided. To include this effect in LCA, the system boundaries are extended to include the resource use and emissions for a product B′ that is considered equivalent to the coproduct B of the main system. Instead of adding these emissions to the system total, they are subtracted to represent that they are avoided due to the by-product (Figure 4.13).

For this method to be applicable, a substituted product must exist, with data available on its emissions and resource use, from cradle to grave. The calculation is only valid when we can demonstrate that the substitution has actually happened or is the most likely use of the coproduct.

The method of avoiding allocation by system expansion can be demonstrated in the example of coproducing wheat and straw. The straw has no direct substitute, but

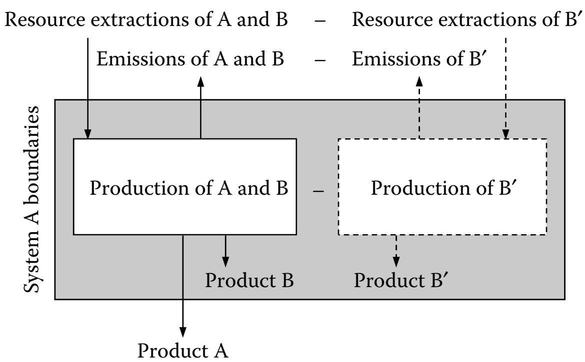


FIGURE 4.13 Avoiding allocation by system expansion. Since the coproduct B replaces a product B′, the raw materials and emissions associated with B′ are avoided and therefore subtracted from those of the main system.

it can be burned and used to replace gas heating, a common practice in Denmark due to its subsidies (Figure 4.14). It is assumed here that straw substitutes gas for heating. The agricultural process that yields 8,000 kg/ha of wheat also yields 2,000 kg/ha of straw, which can be burned to yield 20,000 MJ of heat energy. This amount of heat energy could otherwise have been produced by petroleum that would take 25,690 MJ of primary energy to refine and extract. The primary energy use attributed to wheat is the difference between the primary energy for agricultural production and the petroleum primary energy avoided due to the straw coproduct. Thus, 2,000 kg of straw substitutes enough petroleum to avoid using 25,690 MJ of nonrenewable primary energy, 1,844 g of CO_2 emissions, and 2.9 g of NO_x (Table 4.13). These values for energy and emissions are subtracted from those associated with the coproduction of wheat and straw (27,250 MJ, 2,200 g_{CO2}, and 13.6 g_{NOx}). This attributes only 1830 MJ, 376 g_{CO2}, and 10.7 g_{NOx} to each hectare of wheat produced.

A second possible use of the straw is for electricity–heat cogeneration (Figure 4.15). In this case, the straw substitutes both gas used for heating and electricity production, which ends up avoiding 34,670 MJ of primary energy, 2,129 g_{CO2}, and 3.8 g_{NOx} per hectare of wheat grown (Table 4.13). Since the energy avoided is actually greater than the amount necessary to produce the wheat and straw, the wheat production is

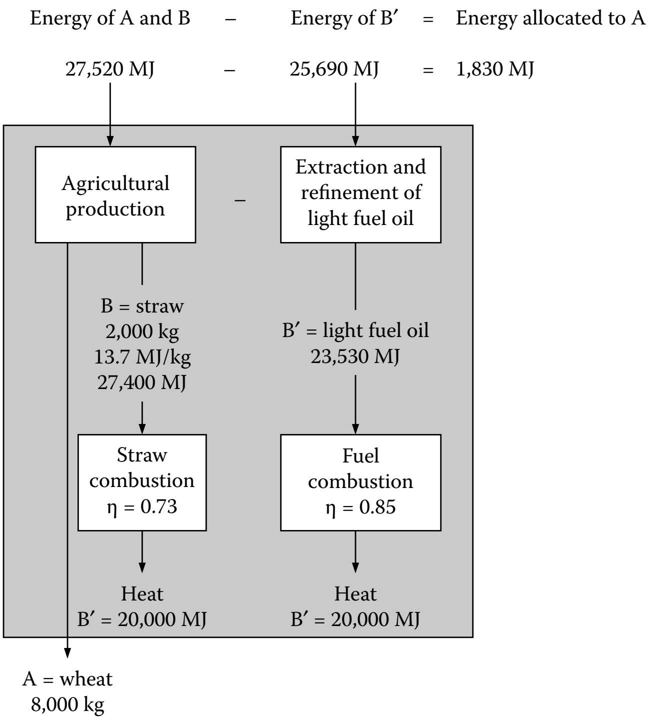


FIGURE 4.14 Application of system expansion for the allocation of nonrenewable primary energy to the coproduction of straw and wheat, where straw is combusted and used as a heat source, which substitutes the alternate heat source of light fuel oil.

TABLE 4.13
Allocation of Nonrenewable Primary Energy and CO_2 and NO_x Emissions between Straw and Wheat for Different Methods

	System Expansion		System Expansion		Marginal Variation			
Allocation Method	**Straw for Heating**		**Straw for Heating and Electricity**		**Straw Reincorporation to Replace Fertilizer**		**Financial Allocation**	
Energy wheat (MJ/ha)	1,830	(7%)	−7,150	(−26%)	27,100	(98.5%)	27,250	(99%)
Energy straw (MJ/ha)	25,690	(93%)	34,670	(126%)	420	(1.5%)	270	(1%)
Energy total (MJ/ha)	27,520	(100%)	27,520	(100%)	27,520	(100%)	27,520	(100%)
CO_2 wheat (g/ha)	376	(17%)	91	(4%)	2,186	(98.5%)	2,198	(99%)
CO_2 straw (g/ha)	1,844	(83%)	2,129	(96%)	34	(1.5%)	22	(1%)
CO_2 total (g/ha)	2,220	(100%)	2,220	(100%)	2,220	(100%)	2,220	(100%)
NO_x wheat (g/ha)	10.7	(79%)	9.8	(72%)	12.9	(95%)	13.5	(99%)
NO_x straw (g/ha)	2.9	(21%)	3.8	(28%)	0.7	(5%)	0.01	(1%)
NOx total (g/ha)	13.6	(100%)	13.6	(100%)	13.6	(100%)	13.6	(100%)

Source: Audsley, A., et al., *Harmonisation of Environmental Life Cycle Assessment for Agriculture*, Final Report for Concerted Action, AIR3-CT94-2028, 1997.

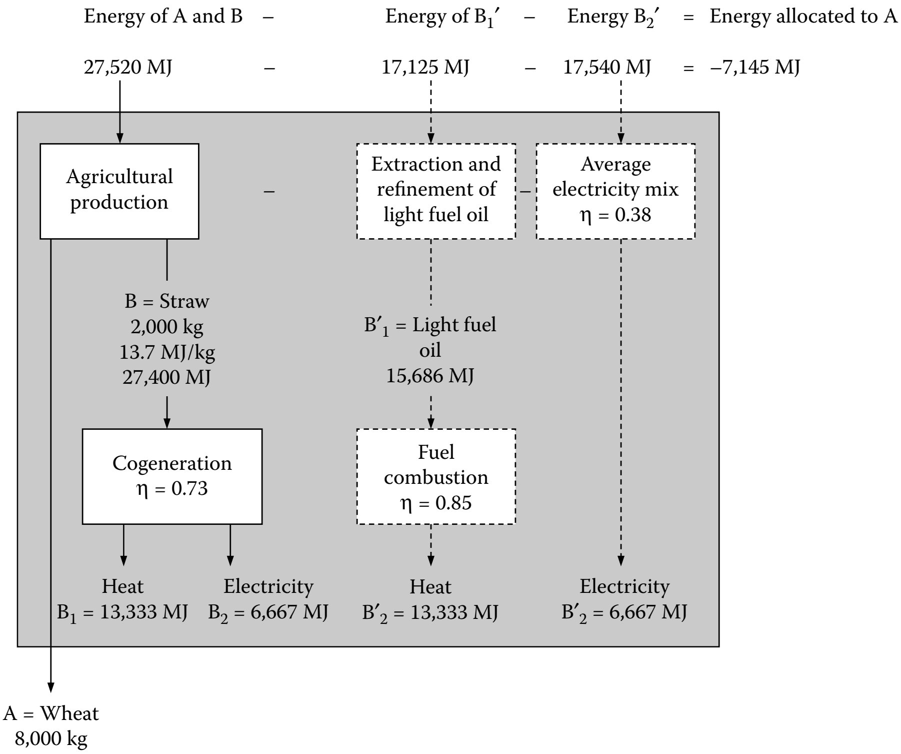


FIGURE 4.15 Application of system expansion for the allocation of nonrenewable primary energy to the coproduction of straw and wheat, where straw is used for cogeneration of heat and electricity.

attributed an energy bonus of −7150 MJ/ha, as well as emissions of 91 g_{CO2}/ha and 9.8 g_{NOx}/ha.

These two possibilities for straw substitution indicate that the choice of how the coproduct is used plays a crucial role in the amount of attributed energy and emissions. If the decision-maker finds a heavily polluting product to replace with the coproduct, this would be an indirect way to reduce the environmental impacts of the main product.

4.5.3.5 (b) Physical Allocation

When not possible to avoid allocation, emissions and resource use should be attributed to different coproducts based on physical causal relationships. There are three main ways for doing so, described as follows:

4.5.3.6 (b1) Marginal Variation

This method is applicable when we can vary at will the ratio of coproducts in a way that corresponds to actual practice. We determine the emissions and resource use in two cases: (i) associated with the baseline situation and (ii) in the case of a

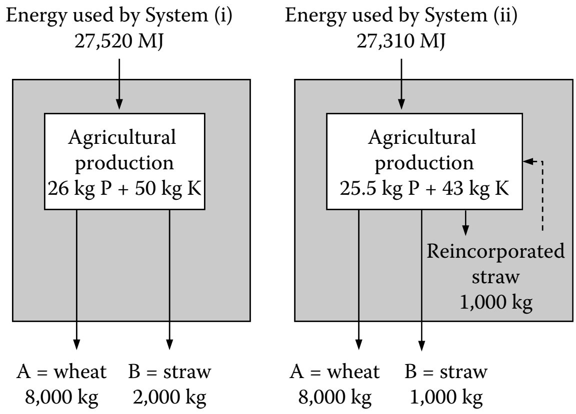


FIGURE 4.16 Application of marginal variation for the allocation of nonrenewable primary energy to the coproduction of straw and wheat. It marginally changes the amount of straw produced to estimate the marginal change in energy use.

variation of the quantity of coproducts. The change in impacts divided by the change of coproduct quantity estimates the amount of impact per unit coproduct.

In the case of wheat and straw production, straw can be reincorporated into the soil (Figure 4.16) rather than collecting it into a hay bale. This reduces the need of fertilizer for the next seeding. The two cases are as follows:

(i) In the baseline case, the system produces 8000 kg of wheat and 2000 kg of straw, and requires fertilizer containing 26 kg phosphorus and 50 kg potassium. The system uses a total of 27,520 MJ of energy.

(ii) If the system is marginally adjusted to reincorporate 1000 kg of straw to replace some fertilizer, it only requires 25.5 kg phosphorus and 43 kg potassium. Due to decreased fertilizer use, the total energy required for the system is 27,310 MJ.

The difference in impacts between the two scenarios is a 210 MJ decrease in energy use. The difference in coproduct use is 1000 kg straw, so the marginal variation is 0.21 MJ/kg straw. This yields 420 MJ allocated to 2,000 kg of straw, which leaves 27,100 MJ allocated to wheat. Other emissions, impacts, and uses of raw materials can be allocated in the same manner. As is true for all alternatives, one must ensure that the incorporation of straw into the soil is a realistic scenario and that less fertilizer is actually used because of this. This scenario can also be examined from the system expansion approach, in which the reincorporated straw is considered a fertilizer substitute.

For marginal variation, we can vary the quantity of straw produced, the amount reincorporated, or the height at which it is cut, all independently from the quantity of wheat produced.

4.5.3.7 (b2) Representative Parameter in the Case of a Common Function

In the case that the coproducts provide an identical function, it is possible to make an allocation based on a quantity or parameter representative of this function.

To apply this allocation, the parameter must represent the common function of the coproducts, this function must correspond to that defined in the study objectives, and the two coproducts must effectively be used for this function. For example, a process that yields both heat and electricity could potentially use energy or rather exergy content as an allocation factor, to also reflect the potential of each energy vector to produce mechanical work. A process that involved chemical reactions could use chemical composition as an allocation factor, and a process that created some type of nutrient could use protein content as an allocation factor.

In the case of allocation between wheat and straw, the allocation could potentially be made based on the energy content of the two products if they were both intended as animal fodder. This function, however, does not correspond to the definition of the goal in this example, which is the production of wheat to make bread for human consumption. This approach thus cannot be applied here. An allocation on the basis of respective masses does not make sense either, because mass is not representative of a common function nor of the emissions in this example.

4.5.3.8 (b3) Property Reflecting a Causal Physical Relation

This method is applicable when we can determine a physical indicator that captures a *cause-and-effect relationship* between the coproducts and associated emissions or resources used. All coproducts must have emissions and resource uses that are strongly linked to the same physical indicator. Thus, the physical indicator must capture the cause of the emission. In any case, the allocation factor requires a scientific and verifiable direct causality as a basis for selection (WRI and WBCSD 2011).

This method is not directly applicable to the straw example, but does apply to the case of treating multiple types of waste (Figure 4.10). For example, in simultaneously treating waste plastics, batteries, and sludge, the heavy metal emissions may be significant. Since heavy metal emissions during waste treatment are directly proportional to their content in the product being treated, they can be allocated accordingly. More concretely, let us consider an example in which some batteries are simultaneously treated with other wastes, where the batteries contain 0.8 kg of cadmium and the other waste contains 0.2 kg of cadmium. If 10 g of cadmium are emitted to air as a result of this process, we allocate 8 g to the batteries and 2 g to the remaining waste.

Industry commonly allocates impacts by mass, even in nonwaste treatment scenarios, which is often questionable, because there is rarely such a direct cause-and-effect relationship, and mass allocation should be avoided unless this causality has been established. A recent interim report on greenhouse gas emissions provides further details on how to best implement this approach (WRI and WBCSD 2011).

If none of these conditions for physical causality (b1, b2, or b3) is fully met, we must not arbitrarily choose a physical parameter. We also must not choose a parameter simply because it is somewhat correlated to the value of the products. Even if the parameter is more stable than the fluctuating price, such an allocation is simply a disguised and imprecise economic allocation. In such a case, it is better simply to apply an economic allocation using a long-term time-averaged price.

4.5.3.9 (c) Economic or Functional Causality

4.5.3.9.1 Financial Allocation

When there is no clear physical relationship to allocate resource use or emissions by-product, we consider economic causality, thereby capturing financial incentives. That is, a product is considered as primarily made for its mercantile value, so we can allocate emissions among coproducts according to their respective values. If relevant, any subprocesses not pertinent to the final product are separated out, as described in Subsection 4.5.3.3. It is at this separation point that the economic value of each coproduct is calculated and used for allocation.

It does not matter whether or not the prices are actually linked to the environmental effects of the product. This allocation method accounts for the incentive of financial income, which is a main driver of production and associated emissions and resource use.

Returning to the example of allocation between straw and wheat, the separation point occurs after the harvesting and before the drying of the wheat and the making of straw bales. The final straw is sold for €0.026/kg, but that is after spending €0.002/kg on baling the straw, so the value of straw at the separation point is €0.024/kg; this effective value is called the *shadow price*. At this point, the price of the grain is around €0.6 €/kg. Thus, the 8000 kg of wheat and the 2000 kg of straw generate respective incomes of €4800 and €48 per hectare, which amounts to allocating 99% of resource use and emissions to the wheat and 1% to the straw.

We reiterate that financial allocation should only be applied if the other allocation techniques cannot be.

4.5.4 Sensitivity Analysis and Comparison of Different Methods

The preceding sections have suggested different methods for allocating energy use and emissions to wheat and straw, and these are summarized and quantified in Table 4.13.

The choice of allocation method greatly influences the amounts allocated to each coproduct. In the example of wheat, the methods of financial allocation and marginal variation have similar results for energy use and CO_2 emissions, whereas the system expansion methods yield completely different results. It is thus essential to indicate clearly the conditions for application of each method. The system expansion method can only be applied if the straw is used to provide energy. When straw is reincorporated, the marginal variation method with straw reincorporation is well adapted and gives results very close to the financial allocation.

Only the financial allocation leads to the same allocation percentages across energy use and emissions. In the other methods, these percentages vary across elementary flows based on the characteristics of the substituted product.

Allocation based on product mass is not recommended in this case, because there is no causality between product mass and energy use or emissions. If applied, this method would yield the arbitrary allocation of 20%–50% of energy and emissions attributed to straw, depending on the quantity of straw produced.

4.5.5 Open-Loop Recycling of Waste-Like Coproducts: Financial Allocation

4.5.5.1 Principle

For open-loop recycling (where the recycled by-product is used outside the considered system) of waste-like coproducts, the distinction between waste and product is often unclear. In such a case, we apply the same principles as for coproducts, with certain adaptations:

- First, reusing or recycling a product may require that the emissions and resources associated with the extraction and preparation of raw materials be allocated to any of the multitude of systems in which the product is reused. This may also apply to composting, waste-to-energy production, and other processes that can be assimilated to recycling or reuse.
- Second, the recycling and treatment processes may also need to be allocated to different production systems.

The allocation of processes shared among different production systems should be done based on the following characteristics:

- The eventual properties reflecting the physical causality
- The economic value of the products
- The number of successive reuses of the product

4.5.5.2 Example: The Case of Manure

The allocation procedure for waste-like coproducts is first illustrated using the example of manure (Figure 4.11). Manure, a coproduct of meat and milk, has waste-like characteristics, but can also be used as a fertilizer for a process such as wheat production. Figure 4.17 shows how such a coproduct can be classified based on its price before and after recycling or treatment. Based on this classification, the emissions associated with the treatment and storage of manure can be financially allocated between the meat and milk production and that of wheat.

For manure, it can be classified in three ways based on its price before and after treatment (Figure 4.17):

- *Waste case:* If the price of manure is still negative after treatment (the dairy producer will have to pay for its disposal), it is truly a waste product. All emissions associated with manure during animal production, storage, and treatment are allocated to the production of meat and milk. Moreover, the wheat production that uses the manure as fertilizer is effectively acting as waste treatment to the manure; thus, a fraction of its emissions should be allocated to the animal production that created this waste. This fraction can be defined based on the ratio between the amount paid to the wheat producer to dispose of the manure and the wheat revenues.
- *Intermediary case:* The manure can have a positive value after treatment, but with costs of storage and treatment that are greater than the final sale

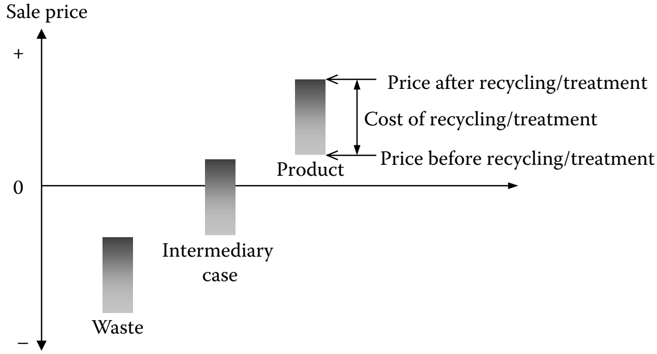


FIGURE 4.17 Financial allocation for a system with three possible classifications of waste-like coproducts, depending on the value before and after recycling or treatment.

price (i.e., the shadow price before treatment is negative). In this case, the emissions during storage and treatment are allocated between vegetable and animal production based on the ratio of the negative price to the positive price after storage and treatment (two-thirds to one-third in Figure 4.17). The emissions due to animal production are entirely allocated to milk and meat, while the emissions due to wheat production are entirely allocated to wheat.

- *Product case:* If the manure has a positive value after treatment because its sale price is greater than the costs of storage and treatment, it is a coproduct of the meat and milk production. Rather than being considered as waste treatment during wheat production, this manure is considered as valuable fertilizer. The emissions during storage and treatment are thus completely allocated to the production of wheat. In addition, a fraction of the emissions of the animal production are allocated to wheat production based on the ratio between the shadow price of manure before treatment (at the point of separation between manure and milk and meat) and the price of the sale of milk and meat. This financial causality represents the fact that part of the profit of the dairy farm is in producing farm fertilizer.

4.5.5.3 Case Study Application as Example

Based on one study of manure (Audsley et al. 1997), the price after storage reaches €3–4.5/m^3 of manure. The storage and treatment cost (assuming an initial investment of €100/m^3 of manure/year, an interest rate of 6%, and a 50-year lifetime with 5% annual payment) is €5/m^3 of manure. This situation corresponds to the intermediary case, where between 60% and 90% of the emissions associated with storage and treatment are allocated to wheat. All emissions associated with spreading the manure are allocated to wheat production.

4.5.6 Summary of Allocation

To conclude the discussion of allocation, we want to emphasize that after years of confusion on this very topic, there is finally a consensus on recommended procedures based on the ISO 14044 norm described at the beginning of this section. This is an important development, because the choice of allocation method can have a crucial influence on the final results. However, certain additional rules and complementary practices remain necessary to avoid abusive interpretations, such as those described in this section. These guidelines are summarized in the following box, adapted from WRI and WBCSD (2011).

GENERAL PRINCIPLES FOR SOLVING ALLOCATION PROBLEMS

"When faced with an allocation problem … users should avoid allocation, i.e. partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (WRI and WBCSD 2011).

The system can be expanded to include substituted products, so that the analyst can estimate and subtract the emissions that would have occurred if the coproduct were substituted by a similar item or the by same product made in a different way. To avoid arbitrary choices, this approach is generally applied if only one of the alternative products is identified as the common substitute. It is necessary to demonstrate that the chosen substitute is the effective replacement of the coproduct and the result is only valid for the selected substitution.

"The allocation process shall adhere to the general accounting principles of completeness (all emissions accounted for), transparency (clear documentation of how emissions are calculated), accuracy (a true accounting of the product's GHG inventory), and consistency (a process that is applied similarly to multiple outputs)" (WRI and WBCSD 2011).

Different types of allocation can be made as follows:

1. Allocate on a physical basis. The physical allocation (by mass, for example) can be applied only if physical causality exists (i.e., only if there is a reason that emissions would be proportional to the physical quantity considered, or if we can vary at will the ratio of coproducts in a way that corresponds to actual practice, as described in the marginal variation section).
2. Allocate on a financial basis based on the market value.
3. Allocate using value choices or best judgment based on reasonable assumptions. The allocation process has a preference for decisions based on natural science, followed by those based on other scientific approaches (e.g., social or economic science). Allocation factors (e.g., mass, energy, volume) based on value choices or arbitrary assumptions are the least preferred basis for allocation decisions. The influence of the choice or assumptions on the study results should be determined in a sensitivity analysis.

Figure 4.18 summarizes these allocation principles in the form of a decision tree.

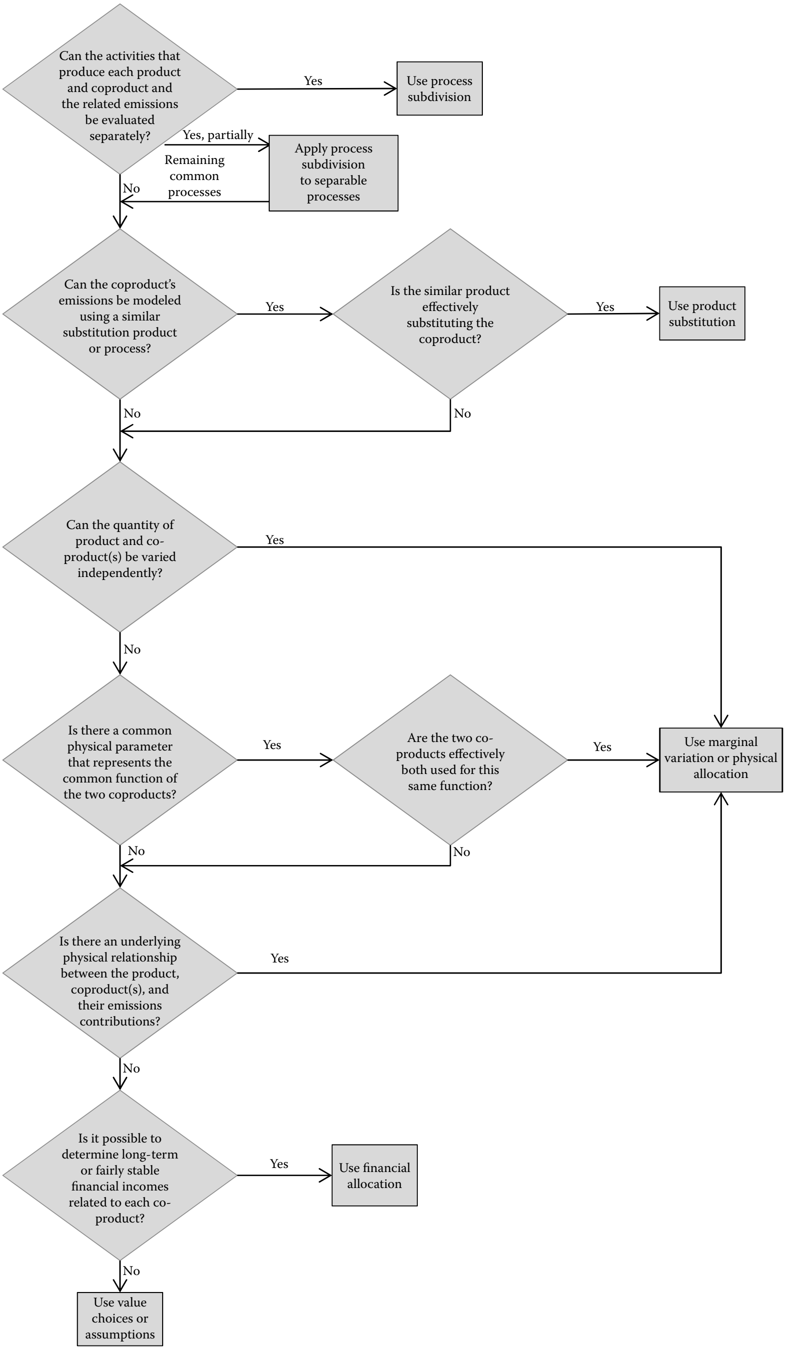


FIGURE 4.18 Decision tree for the choice of allocation method. (Adapted from WRI and WBCSD, *Product Life Cycle Accouning and Reporting Standard*, Greenhouse Gas Protocol, 2011. With permission.)

EXERCISES

Exercise 4.1: Energy and CO_2 Balance of a Gold Ring

Assume that your friend living in California has just ordered a gold wedding ring weighing 6 g. Since it is the week before the wedding(!), the ring must be flown 10,000 km by plane from the Netherlands (where it was made) to California. The manufacturing of the ring requires an electricity consumption of 2 kWh per kilogram of gold and it will eventually be buried (equivalent to being landfilled for this example). Assuming an FU of one ring over the course of one marriage, calculate the reference flows, nonrenewable energy use, and CO_2 emissions over the whole life cycle. Fill in all missing values in Table 4.14.

Exercise 4.2: Electric Light Bulbs

Based on the bulb example (Section 4.2.2), design a flowchart starting from Figure 3.7. Ensure that you have all the processes mentioned in Table 4.2.

Exercise 4.3: Hand-Dryer: Energy and CO_2 Balance

Consider the hand-drying scenarios discussed in Chapter 3. Use the reference flows and flowchart from Exercise 3.2 and the emission factors from Table 4.15. Assume that the manufacturing energy for both devices accounts for less than 1% of total life cycle energy consumption and emissions.

1. Using Table 4.15, estimate the nonrenewable primary energy used and the CO_2 emissions due to each hand-dryer scenario (fill in Table 4.16).
2. For each process and for the sum of all processes, calculate the ratio of CO_2 emissions to nonrenewable primary energy. Check if the values obtained for each ratio are consistent with typical values shown in Figure 4.2.

TABLE 4.14
Processes and Quantities for a Gold Ring Made in the Netherlands and Used in California (Exercise 4.1)

Life Cycle Stage	Process	Unit	Energy (MJ/unit)	CO_2 (kg/unit)	Reference Flow (unit/FU)	Energy (MJ/FU)	CO_2 (kg/FU)
Raw materials extraction	Gold	kg	269,000	16,500	—	—	—
Fabrication	Electricity	kWh	10.71	0.66	—	—	—
Transport	By airplane	ton-km	16.23	1.06	—	—	—
Elimination	Landfill	kg	0.20	0.01	—	—	—
Total						—	—

TABLE 4.15
Emission Factors for Hand-Drying Exercise

Database Process	Unit	Energy (MJ/unit)	CO_2 (kg_{CO2}/unit)
Electricity mix	kWh	12.4	0.703
PP (plastic)	kg	97.5	3.11
Cast iron	kg	64.3	3.9
Steel	kg	24.6	1.51
Paper	kg	17.2	0.86
Truck transport	ton-km	3.7	0.215
PP in landfills	kg	0.33	0.03
Steel in landfills	kg	0.204	0.007
Paper in landfills	kg	0.447	0.015
Incinerated paper	kg	0.292	0.018

TABLE 4.16
Calculation of the Nonrenewable Primary Energy and CO_2 Emissions According to the Process Approach for One Functional Unit

Life Cycle Stage	Process (unit)	Quantity per FU (unit per FU)	Energy per Unit (MJ/unit)	Energy per FU (MJ/FU)	Emissions per Unit (kg_{CO2}/unit)	Emissions per FU (kg_{CO2}/FU)	Check (g_{CO2}/MJ)
Materials							
Fabrication							
Transport							
Use							
Elimination							
Avoided energy							
Total							

3. Now assume that the wastepaper towels, when incinerated, produce 18 MJ of energy per kilogram burned, 20% of which is recovered as usable electricity. Calculate how much nonrenewable primary energy you avoid per kilogram of paper burned, and use this to calculate the avoided energy per FU in the table.
4. Which scenario is better for energy and CO_2? Which stages of the life cycle and which components are most important? What is the importance of the paper towel dispenser or of the electric dryer compared with the other life cycle stages?

Exercise 4.4: Hand-Dryer: Input–Output Approach

Consider the hand-drying scenarios again, but this time using the LCA I/O approach instead of the process-based approach. Assume that the consumer prices for hand-drying are

- Paper towels: $0.01/paper towel; $25/plastic dispenser
- Electric hand-dryer: $0.01/kWh; $350/dryer

Use the data in Table 4.17 to estimate energy use and CO_2 emissions (Table 4.18) for each scenario. Note that the transportation to final user is not considered here, but transportation from raw material is included in sector expenses and impacts of each sector.

Exercise 4.5: Matrix Formulation of the Inventory: Aluminum Manufacturing

For the manufacturing of aluminum, you are given the set of simplified matrices given here, describing the technology matrix (**A**), the demand vector for 1 kg aluminum (**d**), the matrix of direct emissions and extractions (**B**), and the aggregated matrix of emissions and extraction factors (**E**).

TABLE 4.17
Emission Factors for Calculations According to the I/O Approach

Database Process	Energy (MJ/$)	CO_2 (kg_{CO2}/$)
Electricity mix	93	9.9
PP (plastic)	23	1
Cast iron or steel	44	3.0
Paper	15	0.95
Landfilled paper	11	0.38

TABLE 4.18
Nonrenewable Primary Energy Calculation and CO_2 Emissions According to the I/O Approach

Life Cycle Stage	Process	Cost per FU	Energy per $	Energy per FU	Emissions per $	Emissions per FU	Check
	(unit)	(US$/FU)	(MJ/ $)	(MJ/FU)	(kg_{CO2}/ $)	(kg_{CO2}/FU)	(g_{CO2}/MJ)
Materials							
Fabrication							
Use							
Elimination							
Total							

A		Alu	Electricity	Oil	Gas
		kg	kWh	kg	l
Alu	kg	0	0	0	0
Electricity	kWh	15	0	0.3	0.25
Oil	kg	0.05	0.04	0	0
Gas	l	0	0	0	0

$(\mathbf{I}-\mathbf{A})^{-1}$		Alu	Electricity	Oil	Gas		d		$\underline{\mathbf{x} = (\mathbf{I}-\mathbf{A})^{-1}\mathbf{d}}$		
		kg^{-1}	kWh^{-1}	kg^{-1}	l^{-1}						
Alu	kg^{-1}	1	1E-19	0	0		1		**1**	Alu	kg
Electricity	kWh^{-1}	15.2	1.01	0.30	0.25		0		**15.2**	Electricity	kWh
Oil	kg^{-1}	0.66	0.04	1.01	0.01	*	0	=	**0.66**	Oil	kg
Gas	l^{-1}	0	0	0	1		0		**0**	Gas	l

B		Alu	Electricity	Oil	Gas		$(\mathbf{I}-\mathbf{A})^{-1}\mathbf{d}$		$\mathbf{b} = \mathbf{B}\,(\mathbf{I}-\mathbf{A})^{-1}\mathbf{d}$		
		kg	kWh	kg	l		1				
Energy	MJ	1.66	8.22	53.8	40.6		15.2		162	Energy	MJ
CO_2	kg	2.57	0.30	3.54	2.69	*	0.66	=	9.5	CO_2	kg
							0				

$\mathbf{E} = \mathbf{B}(\mathbf{I}-\mathbf{A})^{-1}$		Alu	Electricity	Oil	Gas		d		$\mathbf{b} = \mathbf{E} \times \mathbf{d} = \mathbf{B}\,(-\mathbf{A})^{-1}\mathbf{d}$		
		kg	kWh	kg	l		1				
Energy	MJ	162	10.5	56.9	43.2		0		162	Energy	MJ
CO_2	kg	9.5	0.45	3.67	2.8	*	0	=	9.5	CO_2	kg
							0				

Answer the following questions to see if you understand the formulation:

(a) How many kilowatt-hours of electricity are directly consumed (by Tier 1 processes) in the aluminum manufacturing process per kilogram of aluminum?
(b) How many kilowatt-hours of electricity are consumed per kilogram of aluminum, including upstream processes?
(c) What are the direct CO_2 emissions associated with 1 kWh of electricity?
(d) What are the aggregated CO_2 emissions (including upstream processes) associated with 1 kWh of electricity?
e) What are the aggregated CO_2 emissions (including upstream processes) associated with 0.5 kg of aluminum?
(f) What are the respective contributions of each the aluminum, electricity, and oil sectors to the aggregated CO_2 emissions per kilogram of aluminum?

5 Life Cycle Impact Assessment

Olivier Jolliet, Shanna Shaked,
Myriam Saadé-Sbeih, Cécile Bulle,
Alexandre Jolliet, and Pierre Crettaz

After gathering data on the raw material extractions and substance emissions associated with a product's life cycle, the third phase of a life cycle assessment (LCA) is the life cycle impact assessment (LCIA). The inventory determines the quantities of materials and energy extracted, as well as the emissions to water, air, and soil. But, how do we interpret this inventory data? How do we link these values to their environmental impacts and compare the different impacts? The impact assessment phase addresses these questions. The different steps of the impact assessment are the classification of emissions into different impact categories, characterization of midpoint impacts, and damage (end point) characterization. The impact assessment methods are simple to apply, though their development can be relatively complex. This chapter presents each LCIA step, as well as a concrete example of application. Existing methods are then described in further detail. The developments needed to improve these methods are presented in Section 5.6.

5.1 PURPOSE OF IMPACT ASSESSMENT

The inventory phase generally involves a first aggregation of data by summing emissions of each substance and each resource extraction across the life cycle, resulting in an inventory table of total emissions and extractions for each substance and resource. Even if one scenario has lower emissions of most substances, it generally has higher emissions for several others. To determine which scenario is better, it is then necessary to evaluate the magnitude of the impacts generated by each substance. Therefore, it is also necessary to have methods for aggregating emissions according to their potential(s) to cause one or more environmental impacts.

Due to the complex fate and exposure models needed to predict impacts of such a wide variety of substances, the development of these environmental impact assessment methods may involve complex models. Given the complexity of the task, some argue that it is better to compare the results of different scenarios on the basis of the inventory alone. But, considering only the inventory generally leads to an implicit weighting in which approximately the same weight is given to each pollutant, or in which some inventory flows are arbitrarily considered as more important. An impact assessment based on consistent and explicit criteria is more appropriate than an implicit evaluation, although the uncertainty is important to consider when analyzing results.

Various LCIA methods are available. Due to the many sources of uncertainty, and the different specificities of each method, there is still no reference method used by all LCA practitioners. To minimize bias in selecting and using an impact assessment method, a general framework has been proposed, as well as a set of criteria to be fulfilled (ISO 14044 2006; Udo de Haes et al. 2002; Jolliet et al. 2003a,b). Note that while the development of impact assessment methods can be quite complex, their application is usually trivial, since it consists of multiplying emissions by predefined characterization factors.

5.2 PRINCIPLES OF IMPACT ASSESSMENT

5.2.1 General Principles

How can you compare lead emissions in water with chlorofluorocarbon (CFC) emissions in air? How can you compare increases in human toxicity with contributions to climate change? In other words, how can you compare apples and oranges? Some would say that it is not apples and oranges, but apples and elephants—their impacts are so different! These elements cannot be directly added, and an apple plus an elephant does not equal two apple-elephants (Figure 5.1). But it is still possible to compare an apple and an elephant by considering criteria to which they can both be related. If you are concerned about the resistance of a floor, the weight or the weight per square meter is a good criterion. In the case of an apple weighing about 0.2 kg and an 8 t elephant, the elephant is equivalent to about 40,000 apples! Other equivalencies can be defined from other perspectives and criteria, such as their nutritional potential (in the unlikely case that an elephant is eaten) and the emissions of aromas if we focus on odors!

Which criteria should be used in an LCA to compare such different emissions and resources? Since environmental LCAs are concerned with environmental impacts, substances should be compared based on their capacity to damage the environment and the health of humans. When a polluting substance is emitted to a certain environmental medium, its concentration increases in that medium, and the substance also often transfers to other environmental media (air, water, or soil), bioaccumulates in the food chain, transforms into other substances, and is eventually ingested or inhaled by humans or other species. It ultimately impacts either human health (HH) or the quality of the environment. The path it follows is called the *impact pathway*, which encompasses all the environmental processes from the substance emission to its final impact. The *life cycle impact assessment methods* model the impact

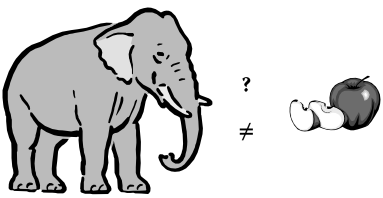


FIGURE 5.1 How to compare an apple and an elephant?

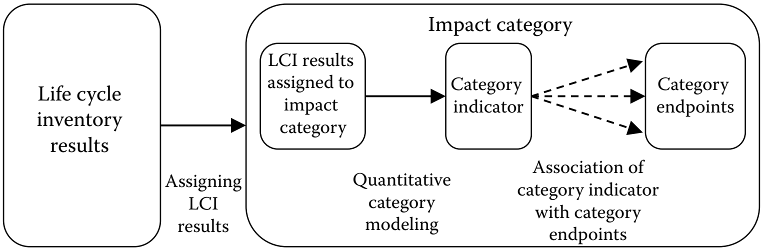


FIGURE 5.2 Impact assessment scheme to link inventory results with category end point or damage to areas of protection. (ISO, *ISO 14040 Environmental management—Life cycle assessment—Principles and framework*, 2006. With permission.)

pathways of different substances to link, as accurately as possible, each inventory data to its potential environmental damage based on these pathways (Figure 5.2).

Taking the example of global warming, the impact pathway includes the following steps: the greenhouse gas emissions generate a change in radiative forcing (first-order effect), which causes an increase in temperature (second-order effect), which has multiple effects including the rise of the sea level due to ice melting or the increase in extreme weather events (third-order effect), which eventually lead to damage to ecosystems and HH (fourth-order effect).

5.2.2 Methodological Framework: Midpoint and Damage Categories

To link the inventory data to environmental damage, a methodological framework has been developed within the Life Cycle Initiative (Section 1.3, Jolliet et al. 2004). First, all inventory results having similar effects (e.g., all the substance emissions that contribute to the greenhouse effect) should be grouped into an impact category at an intermediary level, called a *midpoint category*. For each midpoint category, we define a midpoint indicator. Each inventory flow is multiplied by a *characterization factor* to characterize its contribution to that midpoint category. The term *midpoint* expresses the fact that this point lies somewhere on the impact pathway between the inventory results and the damages. Global warming, for example, is a midpoint category representing the impact of greenhouse gases. The time-integrated change in radiative forcing is typically taken as a midpoint indicator and the contribution of each greenhouse gas to this change in radiative forcing is characterized by a global warming potential, which serves as the characterization factor, representing the contribution of each greenhouse gas emission relative to CO_2. However, others may instead use the increase in temperature as a midpoint indicator.

Each midpoint category is then allocated to one or more *damage categories*, which address the damage to different *areas of protection*, such as HH and ecosystems. The damage category is represented by a *damage indicator*, which is sometimes referred to as an end point indicator. Because each impact assessment step generally involves assumptions about how to characterize the damage contribution to the group, result uncertainty increases as we move from inventory to midpoint and from midpoint to

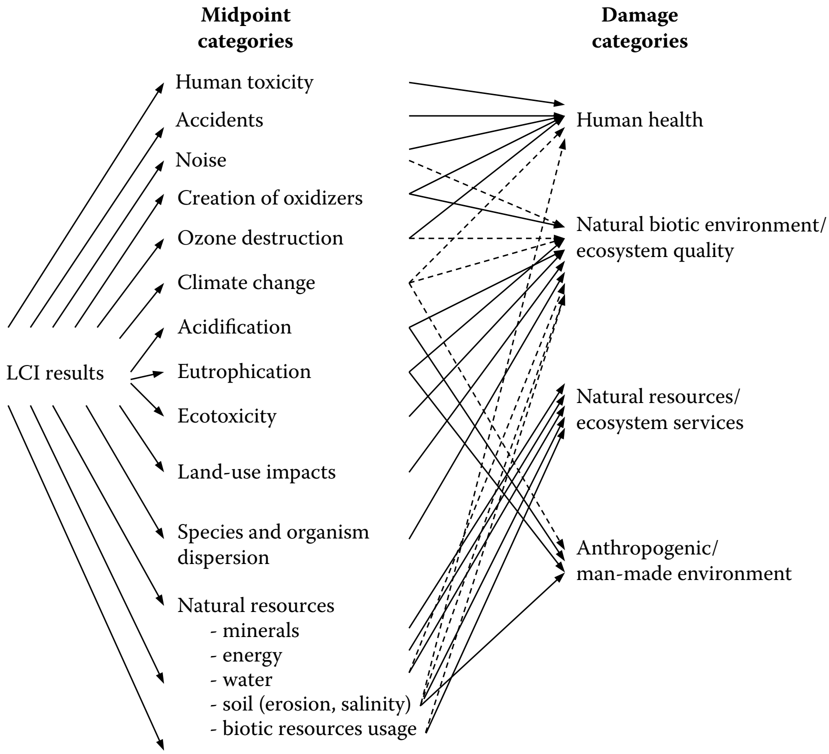


FIGURE 5.3 General structure of the UNEP-SETAC impact assessment framework. The dotted arrows represent conversions from midpoint to damage categories that are particularly uncertain. (Adapted from Jolliet, O., et al. *International Journal of LCA*, 9, 394–404, 2004. With permission.)

damage results. On the other hand, each of these grouping steps yields results that are easier to interpret. For example, a damage expressed in years of life lost is easier to perceive and interpret than the quantity of a pollutant emitted.

Figure 5.3 shows the general scheme of the methodological framework, which links each inventory result to one or more damage categories through midpoint categories (Jolliet et al. 2004). The idea of this analytical framework is that the method designer or its user can choose to stop at the midpoint level (Dutch handbook on LCA) or to go all the way to the damage level (ReCiPe and IMPACT World+ methods). Several impact assessment methods offer both options, as detailed in Sections 5.3 and 5.5).

5.2.3 Steps of Impact Assessment

Following the methodological framework described above, there are three steps in impact assessment: classification of emissions, midpoint characterization, and damage characterization.

5.2.3.1 Classification

During this step, we define a set of midpoint environmental impact categories for the types of environmental problems identified. Emissions are then classified into any relevant midpoint categories on which they have an effect. For example, carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) all contribute to global warming impacts, whereas particulate matter (PM), nitrogen oxides (NO_x), and sulfur dioxide (SO_2) all contribute to impacts of respiratory inorganics (impacts of fine particulate matter). A given substance can contribute to several impact categories, such as methane, which contributes to both climate change and the creation of photochemical oxidants. The midpoint categories in Figure 5.3 provide suggestions for potential impacts to consider, but this list is not comprehensive, and can be simplified or adjusted to match the application.

5.2.3.2 Midpoint Characterization

During midpoint characterization, emissions and extractions are weighted to represent their contribution to each midpoint category. These weighting factors are called *midpoint characterization factors*, and they express the relative importance of substance emissions (or extractions) in the context of a specific midpoint environmental impact category. These factors must be modeled and quantified in a scientifically valid and coherent manner. The inventory flows, emissions, or extractions (u_i in, e.g., kg_i/FU) are multiplied by these factors, and then summed in each midpoint category m to provide a *midpoint score* (S_m^{midpoint} in, e.g., $kg_{\text{CO2-eq}}$/FU) (Equation 5.1):

$$S_m^{\text{midpoint}} = \sum_i \left(CF_{m,i}^{\text{midpoint}} u_i \right) \tag{5.1}$$

where:

$CF_{m,i}^{\text{midpoint}}$ (in, e.g., $kg_{\text{CO2-eq}}/kg_i$) is the midpoint characterization factor of substance i in the midpoint category m

u_i is the emitted or extracted mass of substance i per functional unit as given in the inventory

The midpoint score S_m^{midpoint} is often expressed in units of equivalent mass of a reference substance.

For example, all emissions of greenhouse gases (CO_2, CH_4, NO_2, etc.) may be expressed as equivalent emissions of CO_2, based on how much 1 kg contributes to the greenhouse effect relative to 1 kg of CO_2.

For the global warming category, the Intergovernmental Panel on Climate Change (IPCC) provides characterization factors for greenhouse gases, called *global warming potentials*. Since these gases stay in the atmosphere for varying amounts of time, the global warming potential of a substance depends on the "time horizon" considered (Table 5.1). A 100-year integration over the impact of a greenhouse substance is commonly used in LCA, but this does not reflect the full impact caused over the lifetime of the substance. CO_2, for example, has an effective residence time of more than 100 years and thus contributes more to global

TABLE 5.1
Global Warming Potentials

Industrial Designation or Common Name	Chemical Formula	Lifetime (years)	Radiative Efficiency (W/m^2/ ppb)	Global Warming Potential for Given Time Horizon ($kg_{CO2\text{-}eq}/kg_i$) 20-year	100-year	500-year
Carbon dioxide	CO_2	See below	$1.4E \times 10^{-5}$	1	1	1
Methane	CH_4	12	3.7×10^{-4}	72	25	7.6
Nitrous oxide	N_2O	114	3.03×10^{-3}	289	298	153
Substances Controlled by the Montreal Protocol						
CFC-11	CCl_3F	45	0.25	6,730	4,750	1,620
CFC-12	CCl_2F_2	100	0.32	11,000	10,900	5,200
CFC-13	$CClF_3$	640	0.25	10,800	14,400	16,400
CFC-113	CCl_2FCClF_2	85	0.3	6,540	6,130	2,700
CFC-114	$CClF_2CClF_2$	300	0.31	8,040	10,000	8,730
CFC-115	$CClF_2CF_3$	1700	0.18	5,310	7,370	9,990

Source: IPCC. *Climate Change 2007: Impacts, Adaptation and Vulnerabilit, Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, UK, 2007.

warming over a 500-year time horizon. Table 5.2 illustrates how Equation 5.1 can be applied to the steel front-end panel inventory flows discussed in the Chapter 4 case study (Table 4.6). We can thus characterize the global warming midpoint score in equivalent kilograms of CO_2 ($kg_{CO2\text{-}eq}$) and express the respiratory effects of particulates in equivalent kilograms of particulate matter smaller than 2.5 microns ($kg_{PM2.5\text{-}eq}$).

5.2.3.3 Damage Characterization: Getting from Midpoint to Damage

In damage characterization, we assess the contribution of each midpoint category to one or more damage categories associated with broad areas of protection. Just as each polluting substance contributes to one or more midpoint categories, each midpoint impact may contribute to one or more damage categories. To quantify this contribution, we multiply the midpoint impact score of a category by the *midpoint-to-damage characterization factor* ($MDF_{d,m}$, e.g., in DALY/$kg_{PM2.5\text{-}eq}$) (where DALY refers to disability-adjusted life years), which estimates the damage to the area of protection d caused per unit of the midpoint reference substance of category m. The *damage characterization score* in damage category d, S_d^{damage} (e.g., in DALY/FU), is then obtained by summation (Equation 5.2):

$$S_d^{\text{damage}} = \sum_m MDF_{d,m} \times S_m^{\text{midpoint}} \tag{5.2}$$

TABLE 5.2
Determination of the Midpoint Impact Scores of the Steel Front-End Panel for the Global Warming and the Respiratory Inorganics (PM Effects) Midpoint Impact Categories Using the Characterization Factors of IMPACT World+

	Inventory Emissions (b_i)	Midpoint Characterization Factor (CF^{midpoint})	Midpoint Impact Score (S^{midpoint})
(a) Global Warming			
Units	kg_i/FU	$kg_{CO2\text{-}eq}/kg_i$	$kg_{CO2\text{-}eq}$/FU
CO_2	253.9	1	253.9
CH_4	0.15	25	3.8
N_2O	0.0013	298	0.4
Total			258.1
(b) Respiratory Inorganics (Fine Particulate Matter)			
Units	kg_i/FU	$kg_{PM2.5}/kg_i$	$kg_{PM2.5}$/FU
PM_{10}	0.0383	0.6	0.023
NO_x	0.221	0.0077	0.0017
SO_2	0.439	0.038	0.017
Other			0.0003
Total			0.042

Note: Subscript i indicates the different substances.

For example, the HH damage is a sum of the damages caused by respiratory effects of particulate matter, the HH impacts of global warming, and the impacts of other HH-related midpoint categories such as carcinogens and photochemical oxidants. Table 5.7 provides the midpoint-to-damage characterization factors for the IMPACT World+ method, expressed in damage per unit of the reference substance of each category. As an example, Table 5.3 shows how to calculate the damage to HH of the steel front-end panel, based on the midpoint scores determined in Table 5.2.

Damage scores can also be directly calculated from the inventory results u_i (Equation 5.3) by combining the midpoint and damage characterization steps for each midpoint impact category m to yield the damage characterization factor for substance i in damage category d ($CF_{d,i}^{\text{damage}}$ in, e.g., DALY/kg_i) (Equation 5.4):

$$S_d^{\text{damage}} = \sum_i \left(CF_{d,i}^{\text{damage}} u_i \right) \tag{5.3}$$

where

$$CF_{d,i}^{\text{damage}} = \sum_m MDF_{d,m} \times CF_{m,i}^{\text{midpoint}} \tag{5.4}$$

TABLE 5.3
Determination of the Damage Impact Scores on Human Health of the Steel Front-End Panel, Using the Midpoint-to-Damage Characterization Factor of IMPACT World+ Provided in Table 5.7

		Midpoint Score ($S^{midpoint}$) $kg_{j\text{-}eq}$/FU	Midpoint-to-Damage Factor(*MDF*) DALY/kg_i	Damage Impact Score (S_{damage}) DALY/FU
Global warming (<100 years)	$kg_{CO2\text{-}eq}$	258.1	8.3E–7	0.00021
PM: respiratory inorganics	$kg_{PM2.5\text{-}eq}$	0.042	0.00083	0.000035
Other				0.00088
		Total damage score $S^{damage}{}_{HH}$ (DALY/FU)		0.00113
Normalization factor N_{HH}	0.022 DALY/person-year	Normalized damage score (person-year/FU)		0.051
Weighting factor w_{HH}	€74,000/DALY	Weighted damage score (€/FU)		84

Note: IMPACT World+ normalized score and weighted score calculated according to stepwise method (Weidema 2009).

Nevertheless, it is often useful to keep the contributions of each of the midpoint categories separated for interpretation.

The calculation of the damage score involves more uncertainties than for the midpoint scores, but is still based on scientifically determined or estimated values and a shared damage category; thus, it should not be considered as a value-based weighting as defined by ISO 14044.

5.2.3.4 Optional Steps: Normalization, Grouping, and Weighting

In addition to the three steps defined in the previous subsections, the impact assessment may include three optional steps to aid interpretation and the drawing of conclusions: normalization, grouping, and weighting (Finnveden et al. 2002).

5.2.3.4.1 Normalization

The reference units of many midpoint categories, and some damage categories, are often not initially intuitive, and the meanings of the resulting impacts are thus difficult to interpret. The normalization step expresses a given impact per functional unit relative to the total impact in that category to better understand the magnitude of the damage. It thus compares the respective contribution of the considered product or service to the current total effect on a global, continental, or regional level for a given category (midpoint or damage). The results of the impact characterization are reported relative to these total reference values or normalization values.

We recommend normalizing the damage rather than the midpoint score because midpoint normalization only indicates the relative magnitude of the contribution within that midpoint category, lacking the midpoint-to-damage information about the importance of that midpoint category in the total impacts. The normalized score in the damage category d, $S_{d-\text{normalized}}^{\text{damage}}$, in, for example, person-year/FU, is expressed as (Equation 5.5)

$$S_{d-\text{normalized}}^{\text{damage}} = \frac{S_d^{\text{damage}}}{N_d} \tag{5.5}$$

where:

$S_{d-\text{normalized}}^{\text{damage}}$ is the damage score

N_d (in, e.g., DALY/person-year) is the normalization value of damage category d, which is the total damage of the current total emissions and extractions within damage category d

This total damage is calculated by multiplying the total annual global, continental, or national emissions or extractions in the region r ($u_{i\text{total},r}$ in, e.g., $\text{kg}_{\text{CO2-eq}}$/year) by their respective midpoint or damage characterization factors and dividing this by the total population P in the considered region ($P_{\text{total},r}$, person) to get a normalized score per person. The normalized value for damage category d, N_d, is thus (Equation 5.6)

$$N_d = \frac{\sum_i \left(CF_{m,i}^{\text{damage}} \times u_{i\,\text{total},r} \right)}{P_{\text{total},r}} \tag{5.6}$$

These expressions can be rearranged in various ways to directly calculate the normalized damage scores from normalized characterization factors and inventory results. The normalized score expresses the relative contribution of the considered functional unit to the total damage per person in that category, at the global, continental, or regional level. By default, IMPACT World+ provides global normalization scores for the entire world, putting the impacts of the functional unit in the perspective of total global impacts.

5.2.3.4.2 Grouping

Grouping is a qualitative or semiquantitative process that helps prioritize results by sorting or ranking, and can be done in several ways. Impact categories can be grouped according to area of protection, the types of emissions/resources, or by spatial scale (global, regional, or local). Impact categories can also be ranked based on a predefined hierarchy; for example, high, medium, and low priority. This second procedure is based on value choices that reflect the importance given by society or the user to a particular category.

5.2.3.4.3 Weighting

Rather than have multiple scores for each scenario, for every midpoint or damage category, some users just want a single score for each scenario, which is calculated by weighting the scores in each damage category based on its relative social value. Applying these *weighting factors* to each damage category (Equation 5.7) leads to a final aggregation into a single weighted environmental impact score (S_{weighted} in \$/FU):

$$S_{\text{weighted}} = \sum_{d} w_d \times S_d^{\text{damage}} \tag{5.7}$$

where w_d is the weighting factor of damage category d (in, e.g., \$/DALY).

It is recommended to only apply this weighting to the damage characterization, since the midpoint-to-damage factor already provides an effective natural science–based characterization of midpoint categories (in contrast to weighting, which is value-based) that contribute to a given area of protection (e.g., HH, ecosystem quality, ecosystem services and resources).

Weighting factors are based on social, political, and ethical values and cannot be estimated only using only natural science–based methods. Science is not capable of assessing, for example, the value of a year of life lost compared with the value of an extinct species in a given time and space. The methods that define these factors are thus generally based on monetization, surveys, or distance-to-target approaches:

- Monetization encompasses all methods that estimate weighting factors on a monetary basis. One of the most common approaches is to base these factors on the "willingness to pay" (i.e., what people are willing to pay to avoid a given damage). However, monetary values are not necessarily all additive and comparable. Weidema (2009) proposes the stepwise approach, which enables the practitioner to weight damages to HH, ecosystems, and resources for the IMPACT World+ or ReCiPe methods.
- Surveys of experts or samples of the population can also provide weighting factors based on questionnaires that reveal the perceived relative importance of damage, impact categories, or interventions (Finnveden et al. 2002). Eco-indicator 99 used this approach for its weighting factors.
- The distance-to-target approach links the weighting factors to a target value, based on policy environmental goals.

Finnveden et al. (2002) provide more details regarding good practice for choosing a particular weighting method based on the available data and study objectives. Since such weighting requires incorporating choices based on social, political, and ethical values (Finnveden 1997), it remains a debated element of LCA, especially in the context of industrial applications. Itsubo et al. (2012) recently provided interesting insights into global weighting factors, based on surveys in multiple countries. They show that developed countries tend to give more importance to ecosystem damage compared with developing regions, in which HH is the main emphasis.

Once the total damage score is calculated, Table 5.3 (bottom) shows how the IMPACT World+ normalization factors and the stepwise weighting factors are used to calculate the normalized and weighted damage scores.

5.2.4 Uncertainties and Use of Evaluation Methods of Impact

Moving from inventory to impact assessment increases the ability to interpret results, but also introduces more uncertainties, which means that differences must be greater than a minimum value to be considered significant.

As discussed in Section 6.5 on uncertainties, a difference of less than 10% is generally not considered significant for energy use or CO_2 emissions. The impact categories of respiratory inorganics, acidification, and eutrophication must have scores that differ by greater than 30% to be significant.

Toxicity categories generally require a difference of more than one order of magnitude to be considered significant, especially if each scenario has a different dominant toxic emission, or if the toxicity is due to long-term landfill emissions, which can be very uncertain. Individual characterization factors are considered substantially different when they differ by more than three orders of magnitude (Rosenbaum et al. 2008); this is large, but still limited compared with variations of over 12 orders of magnitude across different substances. Rather than use preliminary toxicity scores as ultimate rankings of dominating pollutants, they are better at identifying the chemicals that contribute to more than a certain percentage of the total score (e.g., 1%). Analysts can thus identify 10–30 chemicals to be considered in priority order and ignore the hundreds of other substances.

The following section presents a practical example to show how to use an impact assessment method.

5.3 APPLICATION EXAMPLE OF THE IMPACT WORLD+ METHOD: FRONT-END PANEL OF AN AUTOMOBILE

The application of a method to assess the impacts of a product or functional unit consists of a series of multiplications by midpoint and damage characterization factors, possibly followed by normalization values and weights. Several impact assessment methods exist and are reviewed in further detail in Sections 5.4 and 5.5. We first apply one such method here, IMPACT World+, to demonstrate such an application. By applying the IMPACT World+ method to the case of automotive components addressed in the previous chapter (Section 4.2.3), this section helps the user understand the operation and use of a method for analyzing the impact.

5.3.1 Analysis of Inventory Results

The inventory of emissions and extractions (Table 4.6) presents the quantities of substances emitted and extracted for four scenarios concerning automobile front-end panels. Though most inventory flows of the steel scenario are higher, no scenario has all the emissions and extractions at the highest or lowest values, so the inventory results alone do not directly enable ranking the environmental performance of the four front-end panels considered. It is therefore essential to have an analytical method to compare the impact and put into perspective the importance of these different emissions and extractions.

5.3.2 General IMPACT World+ Framework and Classification

The detailed emission inventory of emissions and extractions (Table 4.6) are classified into midpoint and damage categories according to Figure 5.4. The impact assessment accounts for all 500 calculated emissions and extractions, even those not listed in Table 4.6.

As a midpoint and damage impact assessment method, IMPACT World+ first applies *midpoint characterization factors* to determine impacts at midpoint levels. IMPACT World+ accounts for 30 midpoint indicators, which are typically regrouped into 10–15 categories for more comprehensible results. While they can be reported and interpreted separately, IMPACT World+ puts these impacts into perspective by calculating the damage of each midpoint impact on any of the *three areas of protection*: HH, ecosystem quality, and ecosystem services and resources. The latter area of protection includes the impacts to human society that have no direct health consequences, such as abiotic (nonliving) resource use and depreciation of ecosystem services. Acknowledging that some users may want to freely select impact categories

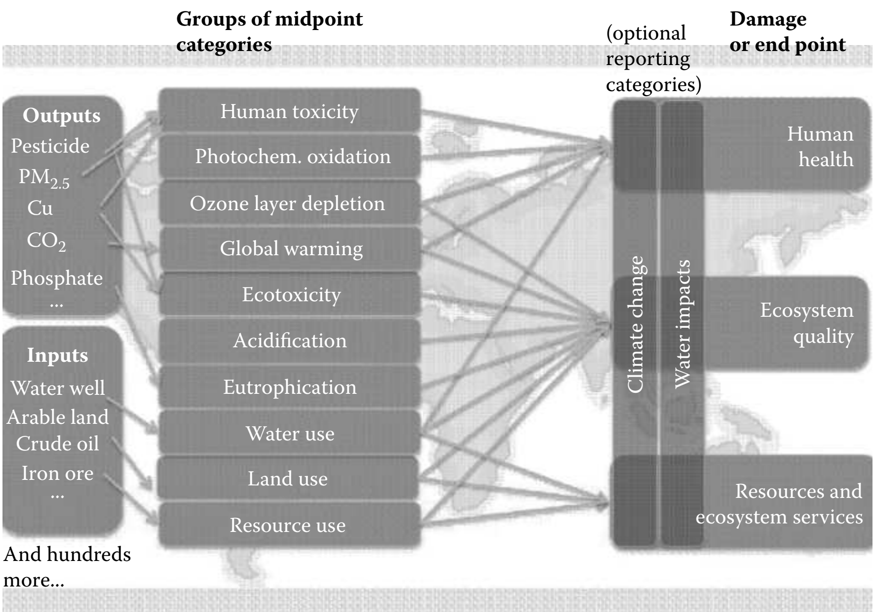


FIGURE 5.4 The IMPACT World+ global framework, from inventory flows to damage to areas of protections.

of particular interest, optional grouping can be made for reporting purposes, separating out the impacts associated with climate change (e.g., carbon footprint) or water (water footprint) without double counting.

5.3.3 Midpoint Characterization of Front-End Panel Scenarios

Within each midpoint category, the midpoint characterization factor ($CF_{m,i}^{\text{midpoint}}$) is used to convert the emitted or extracted substance amount to the equivalent reference substance amount for that category (Figure 5.4). Taking the example of greenhouse gases (Table 5.1), the reference substance is CO_2, and the carbon footprint (CF) of methane (CH_4) is its global warming potential of 25, which means that 1 kg of methane is equivalent to 25 kg of CO_2.

The global warming score of each front-end panel is obtained by multiplying the inventory emissions of each greenhouse gas (Table 4.6) by their global warming potentials and then summing over the product for each substance (Table 5.2a). The same calculations are made for impacts of particulate matter (Table 5.2b), as well as every other midpoint category.

At the midpoint level, the score of each scenario can be expressed as a percentage of the score of the highest scenario in the category (Figure 5.5). The steel panel generates the highest impact in most midpoint categories, except for aquatic ecotoxicity and ionizing radiation, for which the virgin aluminum scenario dominates (Figure 5.5). The "recycled aluminum" scenario generally has the lowest scores, except in the aquatic eutrophication, noncarcinogenic effects, and ionizing radiation categories, where the "composite" front-end panel has the lowest impacts.

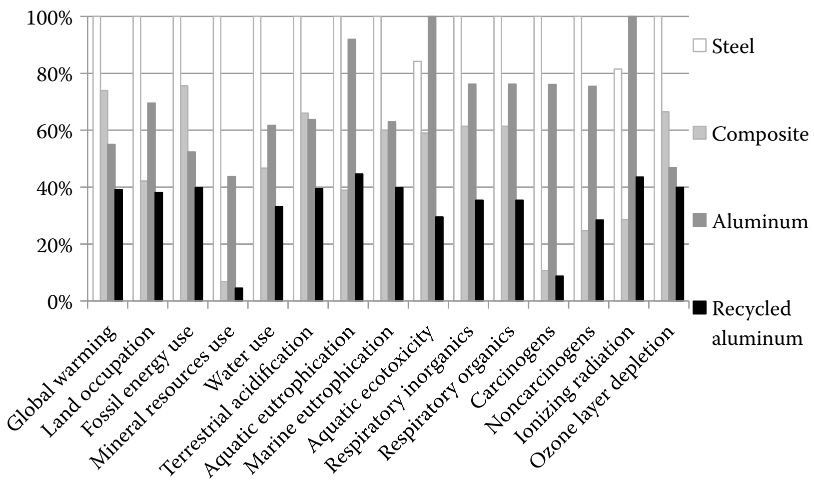


FIGURE 5.5 Comparison of the midpoint characterization scores for the four front-end panels, according to IMPACT World+.

5.3.4 Damage Characterization of Front-End Panel Scenarios

The IMPACT World+ method characterizes aggregated damage in three categories:

1. Impacts on HH include global warming, water use, respiratory effects, carcinogenic and noncarcinogenic effects, ionizing radiation, and the destruction of the stratospheric ozone layer. Table 5.3 gives an example of how to calculate damage scores for global warming and particulate matter effects on HH.
2. Impacts on ecosystem quality include the effects on terrestrial ecosystems (acidification and land use), on aquatic ecosystems (marine acidification, ecotoxicity, ionizing radiation, as well as freshwater and marine eutrophication of water), and the effect of global warming on ecosystems.
3. Resource use includes the extraction of minerals and nonrenewable energy. Due to the uncertainties at damage level, results are displayed on a log scale to avoid overinterpreting small differences among scenarios.

For the *HH impacts*, Figure 5.6 shows that in the case of these four front-end panel scenarios, the impacts due to respiratory organics (also called *photooxidant formation*), ionizing radiation, and ozone layer depletion are two to three orders of magnitude less than other HH impacts. This is common in many assessments and consistent with a systematic analysis carried out by Bulle et al. (2013) for the 4000+ ecoinvent processes.

An important and influential value is the time horizon to consider when calculating impacts. If one only considers impacts over the next 100 years, this reduces impacts by close to an order of magnitude for global warming and carcinogens (bottom of error bars).

Since damage scores have an uncertainty of at least two orders of magnitude, these results suggest that only five categories need to be considered for hotspot

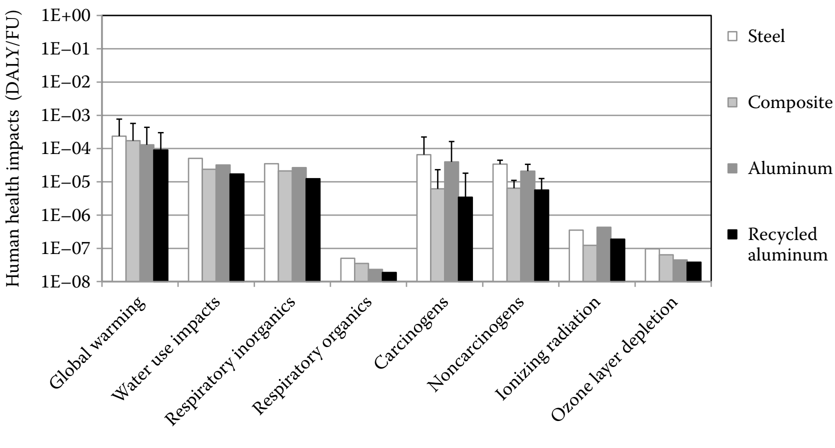


FIGURE 5.6 Comparison of the damage on human health for the four front-end panels, according to IMPACT World+. Error bars represent impacts occurring after 100 years.

(preliminary and targeted) analysis: global warming, carcinogens and noncarcinogens, respiratory inorganics (or particulate matter), and water use.

Mineral, fossil, and water use are modeled based on an extraction–consumption–competition–adaptation approach, meaning that on top of accounting for resource extraction, the impacts also account for resource consumption, competition, and adaptation. Boulay et al. (2011a,b) modeled this cause–effect chain for water use up to HH damage. For the water-use category, impacts may depend on the region considered for the analysis, and the method applied here represents a global default. If the analysis occurs in a developed country, adaptation capacity renders the impact negligible. However, in a region with high water scarcity and low income, water-use impacts will be higher than in this default calculation, as discussed by Boulay et al. (2015) for a laundry and detergent case study. Comparison of the four front-end panels shows that the recycled aluminum scenario ranks better or equal to other scenarios in all relevant categories, whereas the steel panel, due to its larger weight and higher associated emissions while driving, leads to the highest impact on HH.

For *ecosystem damage impacts*, Figure 5.7 shows that global warming and ocean acidification are correlated and half an order of magnitude larger than most other impacts, which are relatively similar in magnitude. Ionizing radiation is negligible. Land occupation has low impacts compared with other categories, because this transportation case study does not involve land-use intensive processes in its system boundaries.

In ranking the ecosystem impacts of each scenario, the recycled aluminum scenario ranks better or equal to the other scenarios in all relevant categories. The steel scenario ranks worst on all ecosystem impacts except water use, for which aluminum manufacturing is worst.

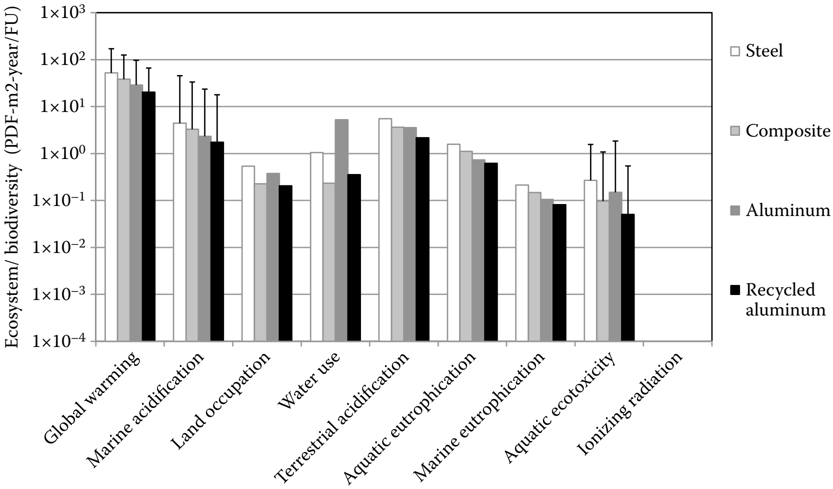


FIGURE 5.7 Comparison of the damage impacts on ecosystem quality for the four front-end panels, according to IMPACT World+. Error bars represent impacts occurring after 100 years.

For impacts on ecosystem services and resources (not shown), the ranking for the subcategory "fossil resources" is similar to the midpoint results (Figure 5.5). This still interim fossil resources category of IMPACT World+ differentiates the energy vectors as a function of the marginal reduction in the availability of a nonrenewable resource (Fatemi et al. 2013), yielding, for example, a lower cost for coal ($\$5 \times 10^{-4}$/MJ) compared with petroleum resources ($\$5 \times 10^{-3}$/MJ); it is still undergoing testing.

5.3.5 Normalization of Front-End Panel Scenarios

Damage scores can be normalized using the global total scores in each damage category: 0.022 DALY/person-year of HH damage; 8,500 PDF-m^2-year/person-year of ecosystem quality damage; and 63,100 MJ/person-year or $315/person-year in ecosystem services and resources, assuming a future increase in energy price due to resource scarcity of $\$5 \times 10^{-3}$/MJ. The bottom of Table 5.3 illustrates for the steel front-end panel how the normalized impacts on HH are calculated.

Figure 5.8 shows the normalized damage score of each scenario in each damage category, which can be interpreted as the contribution of the functional unit compared with the annual impact of a person on HH, ecosystem, or energy resources. The impacts on HH and ecosystems due to global warming can be distinguished from other impacts. About two-thirds of the HH and ecosystem impacts occur after 100 years, but the ranking among scenarios remains independent of time horizon. The recycled aluminum scenario has the lowest impact in all categories, and the steel leads to the highest impacts.

Despite normalized scores all being expressed in the same unit, they should not be summed across areas of protection. Doing so implies a default weight of 1 to each area of protection, which would mean that the total global impacts on HH,

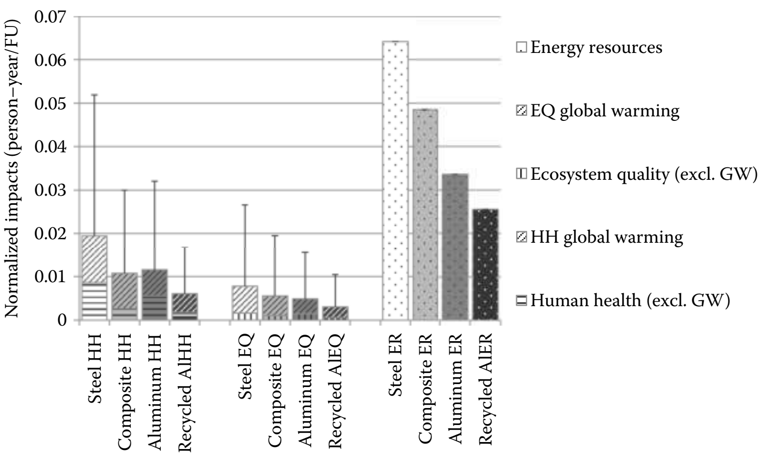


FIGURE 5.8 Comparison of the normalized damage scores of the four front-end panels for human health (HH), ecosystem quality (EQ), and resource and ecosystem services (RE), according to IMPACT World+. Error bars represent impacts occurring after 100 years.

ecosystem quality, and ecosystem services and resources are all considered equivalent. The next section discusses potential weighting schemes.

5.3.6 Weighting of Impacts for Front-End Panel

To compare the overall impact of different scenarios, it is sometimes desirable to aggregate the damage scores of all areas of protection into a single indicator. To do so, we must weigh the damage scores as a function of their relative importance. IMPACT World+ does not provide standardized weights, leaving the user to select their own factors. IMPACT World+ can, however, be coupled with the economic weighting factors of Weidema (2009), which are as follows: €74,000/DALY for HH, €0.14/PDF-m^2-year for ecosystem quality, and €0.86/$ for resources.

For the steel front-end panel, the last row of Table 5.3 illustrates the calculation of the weighted HH damage. Applying this approach to the other three scenarios and summing over all three areas of protection, our case study yields the single weighted scores in Figure 5.9. This confirms that recycled aluminum has the lowest impact and steel the highest, with aluminum and composite being intermediary.

5.4 OVERVIEW OF THE MAIN IMPACT ASSESSMENT METHODS

On the one hand, the example of applying an impact assessment to the front-end panel in Section 5.3 demonstrates that the application of these methods is relatively straightforward. On the other, because of the potential breadth and complexity of impact assessment methods and the many possible applications, there is still no single consensus method. This section presents an overview of the major impact assessment methods (Table 5.4) and their historical development, with more details on each method in Section 5.5. More information on each method is available via the UNEP-SETAC Life Cycle Initiative website (Appendix I).

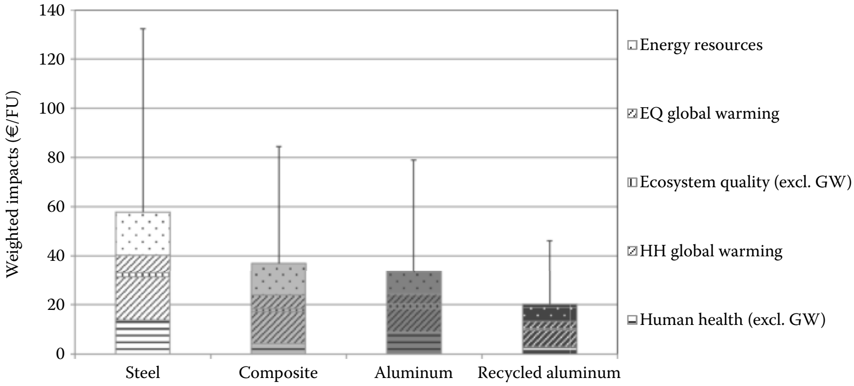


FIGURE 5.9 Single weighted damage score for each of the four front-end panels, accounting for human health, ecosystem quality, and resource and ecosystem services, according to IMPACT World+. Error bars present results occurring after 100 years.

TABLE 5.4
Key Characteristics of the Main Impact Assessment Methods

Methods		**Ecological Scarcity (Frischknecht et al. 2008; 2013)**	**Dutch Handbook on LCA (Guinée et al., 2001)**	**TRACI (Bare et al. 2003)**	**IMPACT 2002+ (Jolliet et al. 2003a,b)**	**LIME (Itsubo and Inaba, 2003)**	**ReCiPe (Goedkoop et al. 2009)**	**EU-LCA (Hauschild et al. 2013)**	**IMPACT World+ (Bulle et al. 2013)**
Evaluation level		Policy objectives	Midpoint	Midpoint	Midpoint and damage	Midpoint and damage	Midpoint and damage	Midpoint and damage	Midpoint and damage
Normalization and weighting		Both based on actual environmental flows and policy objectives	Normalization	Normalization of damage for United States	Normalization of damage. Weighting up to user	Normalization and Japanese weighting	Both (default weighting: 0.4 human health; 0.4 ecosystems; 0.2 resources)	Normalization not yet available	World normalization. Stepwise weighting (Weidema 2009) optional
Covered midpoint categories	Global warming	X	X	X	X	X	X	X	X
	Oceanic acidification						?		X
	Stratospheric ozone depletion		X	X	X	X	X	X	X
	Human health impacts of particulate and toxics	X	X	X	X	X	X	X	X
	Indoor and worker impacts								X
	Accidents		X						
	Ionizing radiation	X	X		X		X	X[a]	X
	Photooxidant formation	X	X	X	X	X	X	X	X

	Acidification	X	X	X	X	X	X	X	X
	Eutrophication	X	X	X	X	X		X	X
	Ecotoxicity	X	X	X	X	X	X	X	X
	Land use	X	X		X	X	X	X	X
	Energy use	X	X	X	X	X	X	X	X
	Mineral extraction				X	X	X	X	X
	Water use	X[b]			X			X	X
	Soil quality								X
	Use of biotic resources		X			X			(X)
Covered damage categories	Human health			X	X	X	X	X	X
	Natural biotic environment (ecosystems)			X	X	X	X	X	X
	Natural abiotic resources			X	X	X	X	X	X
	Natural biotic resources (e.g., tuna)					X			(X)
	Man-made biotic resources (e.g., crops)					X			
	Man-made abiotic resources (e.g., buildings)								

[a] Including effects on ecosystems.
[b] Based on Frischknecht, R., Steiner, R. and Jungbluth, N. (2008). *The Ecological Scarcity Method: Eco-Factors 2006*, Umwelt-Wissen Nr. 0906, Bundesamt für Umwelt (BAFU), Bern.

One of the first LCA impact assessment methods was the critical volumes method (Bus 1984), which took the first step of grouping emissions by emission compartment (air, water, and soil). This method does not account for the persistence or fate of pollutants and is therefore no longer valid. CML 92 (Heijungs 1992) was the first method to focus on the effects of emissions, and has thus been the basis for many further developments. Fate, however, was not accounted for in the evaluation of toxics, as it followed an approach similar to that of the critical volumes method. To overcome this deficiency, Huijbregts (1999) recalculated the impacts of toxics by considering the fate of pollutants, and these calculations have been integrated into the midpoint characterization method described in the Dutch handbook on life cycle assessment (Guinée et al. 2002).

Already, 20 years ago, the Environmental Priorities Strategy (EPS) method (Ryding et al. 1993; Steen 1996) suggested assessing impacts in terms of damage. It had an excellent analytical framework, but the basis of coefficient calculations was sometimes not very transparent. It was thus updated by Steen (1999), giving rise to the EPS 2000d method.

Eco-indicator 95 (Goedkoop 1995) and its update, Eco-indicator 99 (Goedkoop and Spriensma 1999), played pioneering roles in being fully damage-oriented methods, with Eco-indicator 99 becoming one of the references in the field. ReCiPe (Goedkoop et al. 2009) then built on and replaced Eco-indicator 99 and the Dutch handbook on LCA to combine midpoint and damage approaches.

The EDIP97 method (Wenzel et al. 1997) is an approach that stops at midpoint level, and was created to help industries develop environmentally friendly products. Its update, EDIP2003 (Hauschild and Potting 2004), considers spatial differentiation in the modeling of characterization factors.

The ecological scarcity method (Braunschweig et al. 1998; Frischknecht and Büsser Knöpfel 2013) compares the impacts of different emissions based on political targets. It goes beyond environmental impacts to account for areas where environmental pressure is high and therefore indicates the risks of increased environmental costs for companies. Developed in Switzerland, alternative versions currently exist for different countries such as the Netherlands, Colombia, and different regions of Japan (Miyazaki et al. 2004).

The U.S. EPA has developed the TRACI method (Bare et al. 2003; Bare et al. 2006), which assesses impacts at the midpoint level and is closely related to the risk assessment methods of the U.S. EPA. It has been updated to include USEtox for the toxicity-related categories.

In Japan, LIME (Life cycle Impact assessment Method based on Endpoint modeling) quantifies Japanese environmental impacts (Itsubo and Inaba 2003), including damage calculations that are coherent and original.

Following the critical surface time method (CST95), IMPACT 2002+ (Jolliet et al. 2003a,b) allowed impact assessment at both midpoint and damage levels. It was innovative in the assessment of damage to HH and ecosystem quality by adapting the latest concepts of risk analysis to the specificities of LCA. This method has now been replaced by the IMPACT World+ method, which provides factors for each continent at midpoint and damage level.

Finally, the European impact assessment method (Hauschild et al. 2013) has operationalized several recommendations from the Life Cycle Initiative (Jolliet et al. 2004). The recommendations in this guidance document are made based on an analysis of a wide range of existing characterization models used in LCIA. In each impact category, the first step of the analysis was a preselection of existing models. If a method was used in multiple LCIA methodologies, only the most recent and up-to-date version of that method was considered (Hauschild et al. 2010a,b). The second step was the development of general recommendations for each category and drafting of assessment criteria to be used in the evaluation and comparison of the preselected methods. Hauschild et al. (2011) describe the selection criteria for each impact category. The final document presents recommendations for each impact category, implementing the consensus USEtox model (UNEP-SETAC toxicity model) presented by Hauschild et al. (2008) and Rosenbaum et al. (2008). Though recommended for use by the Joint Research Centre (the European Commission science agency), this method lacks somewhat in consistency because indicators at midpoint level are not necessarily compatible across categories.

We recommend always using two or three impact assessment methods in parallel, choosing from among the latest methods in Table 5.4. The quality of the method used is generally more important than its geographic relevance.

In the following section, we summarize the key elements of the most widely used methods (ecological scarcity, EPS, Eco-indicator 99, Dutch handbook on LCA) and those that are the most up-to-date in different parts of the world (TRACI-US, LIME-Japan, ReCiPe 2008, the European LCIA method, and IMPACT World+).

5.5 DESCRIPTION OF THE MAIN IMPACT ASSESSMENT METHODS

Sections 5.5.1 through 5.5.4 briefly describe methods that played an important role in LCIA development and may still be used in certain studies, but which have since been replaced by the more recent methods described in the subsequent Sections 5.5.5 through 5.5.12.

5.5.1 Critical Volumes: An Outdated Approach

The principle of this method (Bus 1984) is to calculate, for each pollutant to air or water, the equivalent polluted volume based on a critical threshold. The equivalent volumes determined for each pollutant are then grouped by medium (air, soil or water). S^m is the critical volume (in units of cubic meters) of medium m (air, water, or soil) (Equation 5.8):

$$S^m = \frac{u_i^m}{C_i^m} \tag{5.8}$$

where:

u_{im} is the emission of substance i in grams

C_{im} is the concentration threshold value of substance i in medium m in grams per cubic meter

The threshold values used for C_i^m generally come from the Swiss regulations on air quality protection (LRV 1985) and waste water discharge (LRV 1975). For example, the maximum allowable concentration of NO_x is 0.03 mg/m^3, which means that 1 m^3 of air is considered polluted for every 0.03 mg of NO_x emitted. The results are given in cubic meters of critical air, cubic meters of critical water, cubic meters of solid waste, and megajoules of energy equivalent.

Although this method has the advantage of simplicity, it is unsuitable on many points. Comparing an emission to a threshold concentration value does not take into account the persistence and fate of pollutants. The method does not allow comparison between air pollutants and water pollutants and does not account for transfer into the food chain. It assumes that a single pollutant is contained in a given volume, when in reality many pollutants may be contained in the same volume. The volumes obtained are, therefore, only used for comparison and have no connection with reality.

This method therefore does not follow the structure defined by the Life Cycle Initiative (Section 1.3) and should not be used as such or within other approaches such as water footprints. Its approach of disregarding the fate of substances and their degradation or dilution may substantially underestimate the effects of long-lived pollutants (by around a factor of 1000, depending on the pollutant).

5.5.2 EPS 2000d Method

The EPS method (Environmental Priority Strategies in Product Design) was developed in Sweden (Ryding et al. 1993; Steen 1996), and updated by Steen a few years later (1999). It aims to provide a monetary value for different types of damage by using the "willingness to pay" concept described in Section 5.2.3. Impacts are expressed in environmental load units (ELU) and can be aggregated into a single indicator. This method, which includes the classification, characterization, and weighting steps, played an important historical role by paving the way for the assessment of damage.

5.5.3 Eco-indicator 99

Building on the original Eco-indicator 95 (Goedkoop 1995) method based on distances to target, the Eco-indicator 99 method (Goedkoop and Spriensma 1999) was one of the first methods to consistently measure damage to resources, HH, and ecosystem quality. The methodology of Eco-indicator 99 was developed "top down," starting with damage (to HH, ecosystem quality, and mineral resources and fossil fuels) to identify the most important impacts and then relate these to inventory emissions.

Human health damage is expressed in equivalent years of life lost (DALYs), and includes respiratory and carcinogenic effects, global warming, destruction of ozone, and ionizing radiation.

Damage to ecosystem quality is expressed as a percentage of extinct species in a certain area due to the environmental burden, which includes ecotoxicity and acidification and eutrophication, along with the impacts of land use and transformation based on empirical data describing the presence of vascular plants.

Extraction of mineral and fossil resources is characterized by the excess energy required for future extractions. A limiting assumption for this model is that all emissions and land uses are assumed to occur in Europe.

The scores in each category are then normalized by the total European score per European, as determined by surveys. The results can be used as a default value but should not be considered representative of the European average.

Eco-indicator results can be presented for three different social perspectives, each of which uses different time horizons, uncertainty management, and other subjective factors. The individualist perspective considers only proven effects on short timescales, believing that technology can avoid many problems. The egalitarian perspective considers all possible effects on very long timescales, believing that problems can lead to catastrophe. The hierarchist perspective is somewhere in between the other two, balancing the short- and long-term perspectives.

Eco-indicator 99 was an attractive method, especially its ability to compare and combine mineral and fossil resources. But, some points were controversial, such as only characterizing energy and fossil resources through the additional energy required for their future extraction, without accounting for the direct use of energy. Because of irreversible dissipation, the energy content of fossil resources also becomes unavailable and should also be considered, as in, for example, IMPACT2002+.

Although Eco-indicator 99 is still used, it is now de facto replaced by the ReCiPe method described in Section 5.5.9.

5.5.4 Dutch Handbook on LCA

The CML 92 method (Heijungs 1992) was at one point one of the most used methods in Europe. It was updated to CML 2002, giving rise to the handbook on LCA (Guinée et al. 2001, 2002), a real cookbook of operational guidelines for performing LCA step by step. This method stops at the midpoint level and is structured as follows.

5.5.4.1 Classification and Characterization

The emissions are classified in different impact categories, themselves divided into three groups:

- Group A includes the basic impact categories, namely abiotic resource depletion, impacts of land use (loss of soil resources), climate change, stratospheric ozone depletion, human toxicity, ecotoxicity (freshwater and marine aquatic ecotoxicity, terrestrial ecotoxicity), the formation of photo-oxidants, and acidification and eutrophication.
- Group B includes more specific impact categories, namely, land-use impacts on biodiversity, sediment ecotoxicity (freshwater and marine), effects of ionizing radiation, unpleasant odors in air, noise, waste heat, and accidents.
- Group C lists other impact categories, namely, the reduction of biotic resources, soil desiccation, and other unpleasant odors (e.g., smelly water).

The characterization factors for human toxicity and ecotoxicity were developed by Huijbregts (1999) using the USES-LCA 2.0 model. This model takes into account the fate of substances in view of their persistence, biodegradation, and transfer between different media (air, water, soil). Exposure is also considered through data such as the volume of air breathed in, the volume of drinking water, and the consumption of fish, meat, vegetables, and dairy produce.

5.5.4.2 Normalization and Evaluation

Scores for each impact category are then normalized by the total global score for this effect to give the contribution of each category to the total global impact. The coefficients of this method can be downloaded (see website details in Appendix I). Like Eco-indicator 99, the CML 2002 method remains popular, but has been officially replaced by the ReCiPe method described in Section 5.5.9.

5.5.5 Ecological Scarcity Method

As described by Frischknecht et al. (2008), the "ecological scarcity" method assesses impacts of life cycle inventories based on the "distance-to-target" principle. If the current emissions of a substance are close to or exceeding critical maximum acceptable flow values, then that substance has a higher ecofactor and, thus, a higher impact score.

Under this method, the total impact score S is calculated from the inventory of substance emissions u_i (in tons) as follows (Equation 5.9):

$$S = \sum_i CF_i \times u_i \tag{5.9}$$

where CF_i is the ecofactor of substance i, expressed in ecopoint per unit of pollutant emission or resource extraction (Equation 5.10):

$$CF_i = CF_i^{\text{midpoint}} \times \frac{c}{F_{ni}} \times \left(\frac{F_{ai}}{F_{ci}} \right)^2 \tag{5.10}$$

where:

CF_i^{midpoint}	is an optional characterization factor, such as a global warming potential of 23 for methane (in equivalent kilograms of CO_2 per kilogram of substance i)
F_{ni},	used for normalization, is the annual flow of substance i in Switzerland in tons per year
F_{ci}	is the critical maximum acceptable flow of substance i in the reference area in tons per year, which is derived from science-based targets of Swiss environmental policy
F_{ai}	is the actual flow of substance i in the reference area in tons per year
c	is a constant, 1E12 (ecopoint per year)

The method was first published in 1990, updated in 1998 (Brand et al. 1998) and adjusted to the structure of the ISO standard in 2006 (Frischknecht et al. 2008), with an enlarged set of substances, additional factors for freshwater use, and the use of the long-term maintenance of soil fertility as the new target for heavy metal emissions to air and to soil.

A new 2013 version (Frischknecht and Büsser Knöpfel 2013) has been published with a few adaptations. New ecofactors were determined for polycyclic aromatic hydrocarbons (PAH) and radioactive isotopes emitted to air, as well as for oil to sea and radioactive isotopes to surface water. For freshwater use, the regionalized ecofactors introduced in the 2006 version have been expanded to all countries and continents to be applied to consumed water rather than water withdrawal. For resources, new ecofactors were introduced for abiotic resources and land use in several biomes based on the land-use impacts upon plant biodiversity. Finally, new ecofactors have been introduced for noise of various means of transportation (i.e., truck, car, train, aircraft), which are adaptable to individual noise levels. This method has the advantage of presenting only one end result and, contrary to the critical volumes method, it is internally consistent because it compares emissions to critical emission flows. This method is useful to apply in conjunction with other methods or other midpoint damage-oriented methods. It should, nevertheless, be noted that the critical flows are difficult to determine for a large number of substances and that target values are based not only on the potential damage, but also on what is achievable.

5.5.6 TRACI, the U.S. EPA Method

The U.S. EPA has developed TRACI (Tool for the Reduction and Assessment of Chemical and other environmental Impacts) (Figure 5.10), which characterizes potential effects, under U.S. conditions, for the following midpoint impact categories: ozone depletion, global warming, acidification, eutrophication, tropospheric ozone

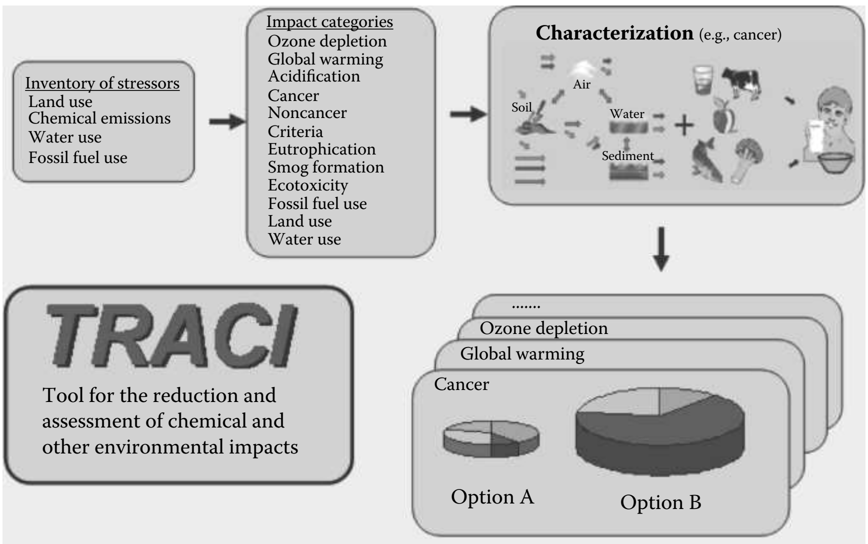


FIGURE 5.10 Structure of the TRACI impact assessment method. (From EPA, 2015, http://www.epa.gov/nrmrl/std/traci/traci.html.)

(smog) formation, ecotoxicity, human particulate effects, human carcinogenic effects, human noncarcinogenic effects, and fossil fuel depletion. Location-specific methodologies within the United States were developed for the following impact categories: acidification, smog formation, eutrophication, carcinogenic effects, noncarcinogenic effects, and criteria air pollutants. TRACI has adopted the USEtox characterization factors for human carcinogenic and noncarcinogenic categories, as well as for ecotoxicity.

TRACI impact categories were only characterized at the midpoint level. However, normalization factors were published in a second stage (Bare et al. 2006; Laurent et al. 2011a,b) to provide and communicate information on the relative significance of the indicator results, and prepare for additional procedures, such as grouping, weighting, or life cycle interpretation. The TRACI characterization factors are available in spreadsheet format, on request at http://www.epa.gov/nrmrl/std/sab/traci/.

5.5.7 IMPACT 2002+

IMPACT 2002+ (Jolliet et al. 2003a,b) is an impact assessment method providing results both at midpoint and at the damage level. The inventory results are first grouped into 14 midpoint impact categories, with similar impact mechanisms or pathways. This method introduced original developments in calculating human toxicity and ecotoxicity. For several other categories, IMPACT 2002+ incorporated or adapted elements of the Eco-indicator 99 method (Section 5.5.3) and the Dutch handbook on LCA (Section 5.5.4). These midpoint categories are then assigned to four categories of damages, representing the changes in environmental quality. IMPACT 2002+ considers the overall effects of an emission or extraction integrated over a long-term or infinite time horizon.

In IMPACT 2002+, the normalized damage factor is obtained by dividing the calculated impact by the total impact of all European emissions and extractions contributing to the studied category, preferably at the damage level, but also available at the midpoint level. The standard normalized score is therefore expressed in equivalent person-years per functional unit. IMPACT 2002+ does not provide standardized weighting factors, but the user can enter their own factors to weight the normalized scores. By assigning a default weight of 1 to each category, the user considers that the total European impacts on HH, ecosystem quality, climate change, and resources are all equivalent. More general information about IMPACT 2002+ is given in Jolliet et al. (2003a,b). The general structure of IMPACT 2002+ is to group midpoint impacts into four damage categories: HH, ecosystem quality, climate change, and resources (Figure 5.15).

5.5.7.1 Human Health

Human toxicity (either carcinogenic or noncarcinogenic), ionizing radiation, depletion of the ozone layer, respiratory effects of inorganic particulate matter, and photooxidant formation are the midpoint categories contributing to HH damage. The effects of toxic substances on humans and ecosystems were modeled using IMPACT 2002, a multimedia and multipathway exposure model developed for western Europe by Pennington et al. (2005). This model also allows for spatial differentiation of 135 European regions. For human toxicity, the HH damage is directly calculated at damage level, then back-calculated in terms of the reference substance for this midpoint category, namely vinyl chloride. The midpoint-to-damage characterization factors convert any midpoint HH impacts from kilograms of equivalent substance to HH damage in units of effective years of life lost (DALYs). A normalization value of 0.0068 DALY/person-year is applied for HH, corresponding to a reduction in life expectancy of 0.5 year per person over their lifetime. This value is calculated as the sum of all impacts that contribute to HH, excluding those caused by climate change.

5.5.7.2 Ecosystem Quality

The midpoint categories aquatic ecotoxicity, terrestrial ecotoxicity, aquatic acidification, terrestrial eutrophication and land use contribute to the loss of ecosystem quality. Ecosystem damage is expressed in the potentially disappeared fraction of species (PDF) over one square meter in one year. The damage score for ecosystem quality is divided by a normalization value of 13,700 PDF-m^2-year/person-year. This normalization value was determined by combining all ecosystem impacts due to aquatic and terrestrial ecotoxicology, eutrophication, aquatic and terrestrial acidification, and land use.

5.5.7.3 Climate Change

For the climate change category, the IPCC (2007) provides substance characterization factors for multiple time horizons, each accounting for the contribution to warming integrated over a set time. Most LCA methods use a 100-year integration. This does not account for the full impact caused by a CO_2 emission, which takes centuries to effectively decay (IPCC 2007). Thus, IMPACT 2002+ uses the longest time horizon of 500 years when calculating characterization factors (Table 5.1). The normalization value for climate change corresponds to the amount of greenhouse gases emitted per person per year in Europe, namely 9950 $kg_{CO2\text{-}eq}$/person/year.

5.5.7.4 Resources

The two midpoint categories contributing to the resources damage category are the consumption of nonrenewable primary energy and the extraction of minerals. Because energy resources are expressed in terms of nonrenewable primary energy dissipated, the characterization factors include the surplus energy required to extract future resources as well as the overall nonrenewable primary energy content of fossil fuels (based on the calorific value of fossil fuels and on the ecoinvent value of 450 MJ/kg uranium for nuclear resources). Unlike nonrenewable primary energy, minerals do not disappear, but are simply distributed into the economy and the environment and can often be reused. Mineral use is thus aggregated with energy use based on the additional mining energy consumed in the future due to lower ore grades, expressed in additional megajoules per extracted unit. The normalization value for resources is based on the annual consumption of nonrenewable primary energy per person in western Europe, which is 152,000 MJ/person/year.

The IMPACT 2002+ method has been replaced by IMPACT World+, as described in Section 5.5.11. It still remains an often-used method today.

5.5.8 LIME: The Japanese Method

In response to the need for an LCIA method adapted to Japan, LIME (Figure 5.11) was developed to quantify environmental impacts induced by environmental loading in

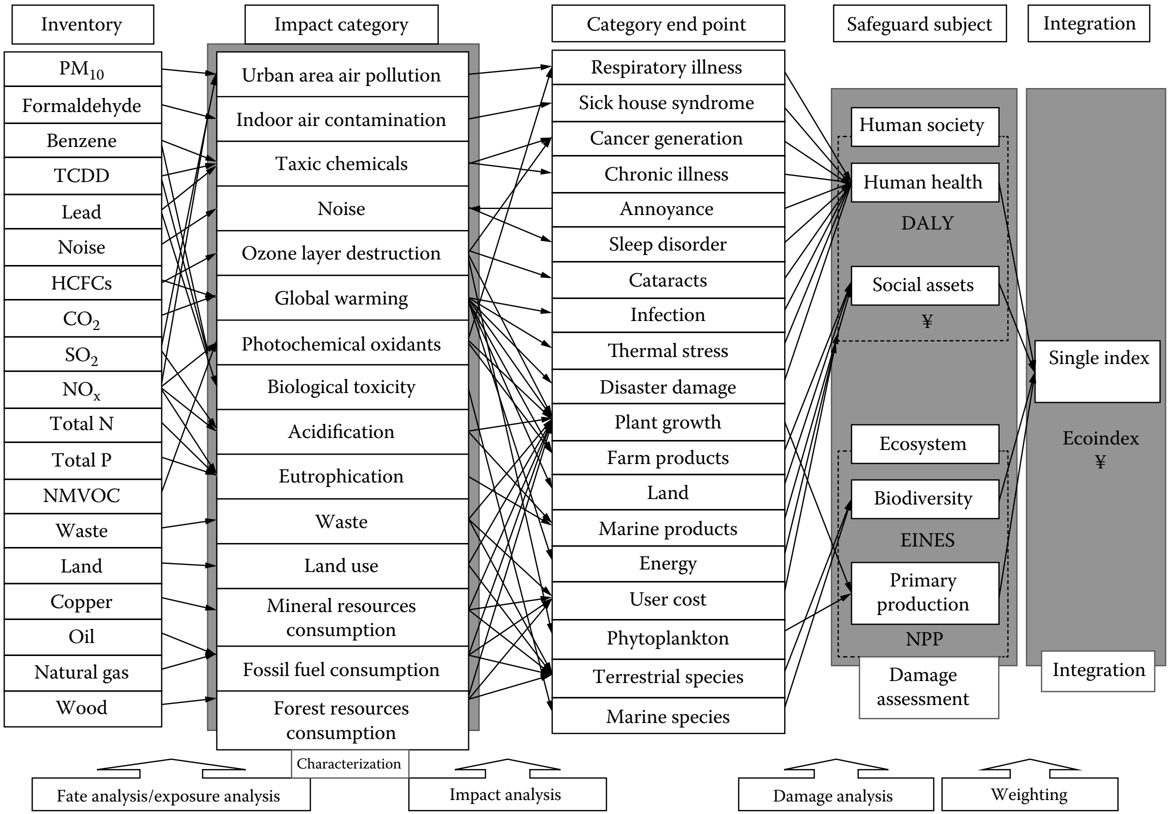


FIGURE 5.11 General structure of the LIME2 method. (From Itsubo, N. and Inaba, A. LIME2 Life-cycle Impact assessment Method on Endpoint modeling, Summary, JLCA newsletter, March 2012, Life-Cycle Assessment Society of Japan. http://lca-forum.org/english/pdf/No12_Summary.pdf).

Japan (Itsubo and Inaba 2003). With a strong emphasis and originality on a consistent assessment of environmental impacts at the damage level, LIME specifically aimed at developing characterization factors and damage factors that reflect Japan's environmental conditions, along with weighting factors that reflect Japan's social values.

Figure 5.11 illustrates the conceptual structure of LIME, which is similar to the IMPACT 2002+ method discussed in the previous subsection, aggregating an inventory of emissions and resource use into various levels of impact. One important LIME midpoint impact category is urban air pollution, which can better account for urban conditions. The 11 LIME midpoint impact categories are further aggregated into the following four areas of protection: HH, social welfare, biodiversity, and plant production, respectively, expressed in terms of DALYs, Japanese yen, EINES (expected increase in number of extinct species), and NPP (net primary production).

The final weighting steps integrate the four damage categories into a single index using conjoint analysis, a technique mostly applied in market research and environmental economics. Two types of weighting factors were obtained by using this method: willingness to pay and dimensionless indicators.

5.5.9 ReCiPe 2008

ReCiPe 2008 (Goedkoop et al. 2009) arose from the desire to merge the midpoint approach of the CML method (Guinée et al. 2002; Section 5.5.4) with the damage approach of the Eco-indicator 99 method (Goedkoop and Spriensma 1999; Section 5.5.3). Collaboration with RIVM (Dutch National Institute of Public Health) and the University of Nijmegen ensured access to knowledge and models over a wide range of environmental issues, ranging from acidification to climate change. This synthesis work resulted in the ReCiPe LCIA method, with impact category indicators and characterization factors at the midpoint and damage levels (Table 5.5 and Figure 5.12).

The model structure depicted in Figure 5.12 is similar to methods presented previously in linking the life cycle inventory (left) to a midpoint indicator (middle) and damage indicator (right). An added feature of the ReCiPe method is that results are presented for three different social perspectives, based on subjective value choices, such as time horizon and uncertainty management (see Section 5.5.3 for a description of these perspectives taken from the Eco-indicator approach).

Eighteen impact categories are addressed at the midpoint level, leading to 18 characterization factors (Table 5.5). A general criterion used to define these impact categories and indicators is that midpoint impact categories should have a standalone value in a midpoint-oriented LCIA method, but that they should also be usable as an intermediate step in a damage-oriented method.

The last three columns of Table 5.5 link each midpoint category to relevant damage categories. ReCiPe uses the following three damage categories: damage to HH expressed in DALYs, damage to ecosystem diversity (ED) expressed in loss of species per year, and damage to resource availability (RA) expressed in U.S. dollars. ReCiPe 2008 characterization factors have been tabulated in a Microsoft Excel spreadsheet available at www.lcia-recipe.info.

TABLE 5.5
ReCiPe Midpoint Categories, including Assignment to Damage (End Point)

Midpoint Impact Category			**Damage Impact Category**[a]		
Name	Abbreviation	Unit	HH	ED	RA
Climate change	GWP	kg (CO_2 to air)	+	+	
Ozone depletion	ODP	kg (CFC-11 to air)	+	–	
Terrestrial acidification	TAP	kg (SO_2 to air)		+	
Freshwater eutrophication	FEP	kg (P to freshwater)		+	
Marine eutrophication	MEP	kg (N to freshwater)		–	
Human toxicity	HTP	kg (14DCB to urban air)	+		
Photochemical oxidant formation	POFP	kg (NMVOC to air)	+	–	
Particulate matter formation	PMFP	kg (PM_{10} to air)	+		
Terrestrial ecotoxicity	TETP	kg (14DCB to industrial soil)		+	
Freshwater ecotoxicity	FETP	kg (14DCB to freshwater)		+	
Marine ecotoxicity	METP	kg (14DCB to marine water)		+	
Ionizing radiation	IRP	kg (U^{235} to air)	+		
Agricultural land occupation	ALOP	m^2-year (agricultural land)		+	–
Urban land occupation	ULOP	m^2-year (urban land)		+	–
Natural land transformation	NLTP	m^2 (natural land)		+	–
Water depletion	WDP	m^3 (water)			–
Mineral resource depletion	MDP	kg (Fe)			+
Fossil resource depletion	FDP	kg (oil)			+

Source: Goedkoop, M., et al., *ReCiPe 2008. A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level*. The Hague, the Netherlands: VROM, 2009. http://www.pre-sustainability.com/download/misc/ReCiPe_main_report_final_27-02-2009_web.pdf

[a] HH refers to human health, ED to ecosystem diversity, and RA to resource availability; "+" means that a quantitative connection has been established for this link in ReCiPe 2008; "–" means that although this is an important link, no quantitative connection could be established. In the abbreviations, P is for "potential".

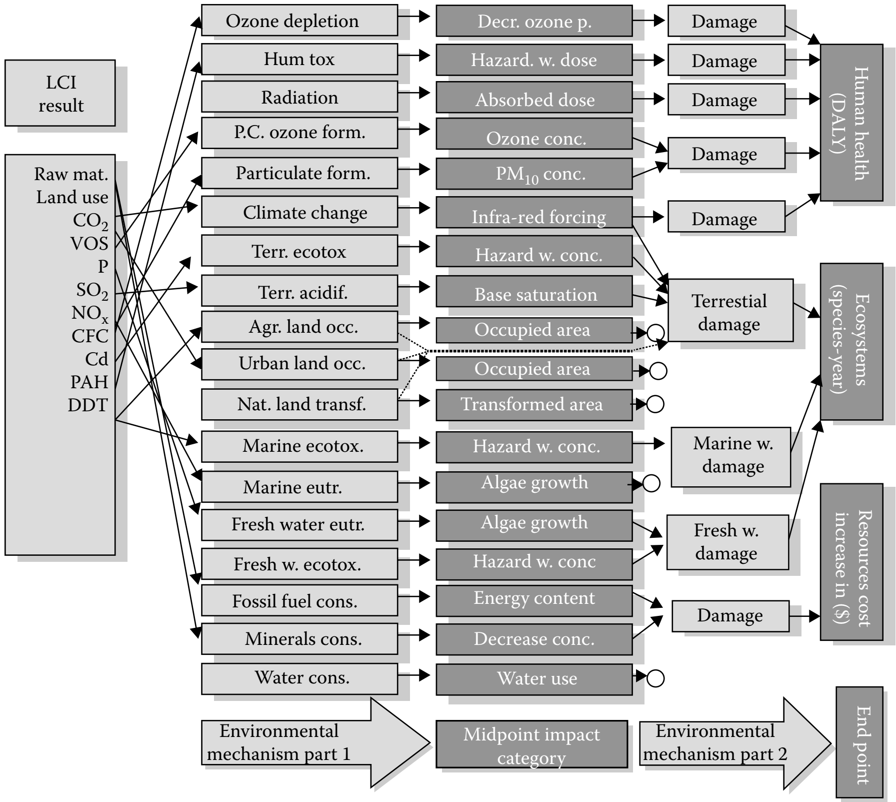


FIGURE 5.12 General structure of the ReCiPe 2008 method. (From Goedkoop, M., et al., *ReCiPe 2008. A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level*. The Hague, the Netherlands: VROM, 2009. http://www.pre-sustainability.com/download/misc/ReCiPe_main_report_final_27-02-2009_web.pdf. With permission.)

5.5.10 New European Method

Developed under the auspices of the International Reference Life Cycle Data System (ILCD), the ILCD LCIA method focuses on the following points (Hauschild et al. 2010a,b; 2011, 2013; see Table 5.6):

1. Analysis of existing LCIA methodologies and related approaches
2. Definition of evaluation criteria for recommended LCIA methods and general recommendations for characterization models and areas of protection
3. Guidance on recommended LCIA characterization framework, models, and factors
4. Documentation of recommended characterization factors in the ILCD data system

TABLE 5.6
Summary of Recommended Methods and Their Quality Classification for Each Midpoint Impact Category and Calculation from Midpoint to Endpoint

	Recommendation at Midpoint		From Midpoint to Damage	
Impact category	Recommended default LCIA method and indicator	Classification[a]	Recommended default LCIA method and Indicator	Classification
Climate change	IPCC radiative forcing expressed as global warming potential 100 years (GWP100)	I	De Schryver et al. (2009): Link to DALY (HH) and PDF-m^3-year (ED)	III
Ozone depletion	1999 WMO Ozone Depletion Potential (ODP)	I	ReCiPe model for human damage, in DALY	III
Human toxicity, cancer effects	USEtox comparative toxic unit for humans (CTU_h)	II/III	DALY calculation adapted from Huijbregts et al. (2005)	II/III
Human toxicity, noncancer effects	USEtox comparative toxic unit for humans (CTU_h)	II/III	DALY calculation adapted from Huijbregts et al. (2005)	III
Particulate matter/ respiratory inorganics	Humbert et al. (2011): Intake fraction for fine particles ($kg_{PM2.5\text{-}eq}$/kg)	I/II	DALY calculation adapted from van Zelm et al. (2008)	I/II
Ionizing radiation, human health	Frischknecht et al. (2005): Human exposure efficiency relative to U235	II	DALY	III
Ionizing radiation, ecosystems	AMI comparative toxic unit for humans (CTU_h)	III	None identified, possible to use PDF-m^3-year	–
Photochemical ozone formation	ReCiPe tropospheric ozone concentration increase (POCP)	II	DALY calculation from van Zelm et al. (2008)	II
Acidification and terrestrial eutrophication	(Seppälä et al. 2006, Posch et al. 2008): Accumulated exceedance (AE)	II	PDF	III
Eutrophication, aquatic	Struijs et al. (2008): Residence time of nutrients in freshwater (P) or marine end compartment (N)	II	Damage to freshwater ecosystems: PDF-m^3-year	III

TABLE 5.6 (Continued)
Summary of Recommended Methods and Their Quality Classification for Each Midpoint Impact Category and Calculation from Midpoint to Endpoint

	Recommendation at Midpoint		From Midpoint to Damage	
Ecotoxicity	USEtox comparative toxic unit for ecosystems (CTU_e)	II/III	None identified, possible to use PDF-m^3-year	–
Land use	Milà i Canals et al. (2007): Soil organic matter	III		III
Resource depletion, water	Frischknecht et al. (2008): Water use related to local water scarcity	II	None identified	–
Resource depletion, mineral, fossil and renewable	CML 2002 scarcity	II	ReCiPe surplus costs	III

Source: Hauschild, M., et al., *International Journal of Life Cycle Assessment*, 18, 683–697, 2013. http://dx.doi.org/10.1007/s11367-012-0489-5.

Note: Depletion of renewable resources is included in the analysis but none of the analyzed methods is mature for recommendation.

[a] I: Recommended and satisfactory; II: Recommended, some improvements needed; III: denoted as interim, that is, the most appropriate among the existing approaches but not yet ready for recommendation.

The ILCD method provides global recommendations and characterization factors for the modeling of the most common impact categories, linking emissions and resources consumed over the life cycle to the indicators at midpoint and end point level.

The recommendations and factors are mostly based on existing LCIA models and characterization methods, supplemented by a selection of environmental models. Recommendations are given for climate change, ozone depletion, human toxicity (including ionizing radiation impacts), particulate matter/respiratory inorganics, photochemical ozone formation, acidification, eutrophication, ecotoxicity (including ionizing radiation impacts), land use, resource depletion, and a number of other impacts (i.e., noise, accidents, erosion, desiccation, and salinization). The overall framework assesses these impacts relative to the three areas of protection: HH, natural environment, and natural resources. The recommended methods and associated characterization factors are each classified by quality:

Class I: Recommended and satisfactory
Class II: Recommended, some improvements needed
Class III: Denoted as interim; that is, the most appropriate among the existing approaches but not yet ready for recommendation

Though developed in Europe, the ILCD LCIA method aims to be globally applicable due to the global nature of goods and services, crossing national and geographic borders over a life cycle. The ILCD recommendations and their classification are summarized in Table 5.6 for midpoint categories and for the link from midpoint to damage.

5.5.11 IMPACT World+

IMPACT World+ was developed in response to the need for a regionalized impact assessment method covering the entire world, and implementing state-of-the-art characterization modeling approaches. Developed as a joint major update to IMPACT 2002+, EDIP, and LUCAS (a Canadian-specific impact assessment method; Toffoletto et al. 2007), it includes uncertainty information to account for both spatial variability and model uncertainty. It regionally assesses the impacts of any georeferenced emission/resource use, providing continent-specific characterization factors. It also calculates the uncertainty associated with an unknown emission location based on the geographical variability of characterization factors at a given geographical scale.

We wish to highlight three key features of IMPACT World+:

1. IMPACT World+ has multiple novel ways of characterizing midpoint level impacts. It is the first LCIA method to include the consensus-based USEtox model for toxicity (Hauschild et al. 2008; Rosenbaum et al. 2008) and water-use impacts (Boulay et al. 2011a,b) consistently in a damage-oriented method. It also includes major modeling improvements, such as the inclusion of ecosystem services in land use (Saad et al. 2013), the inclusion of acidification in an improved atmospheric fate model (Roy et al. 2012 a,b), improved water and mineral resource use with the introduction of an extraction–consumption–competition–adaptation approach, more accurate respiratory effects based on new epidemiologically derived factors (Humbert et al. 2011; Gronlund et al. 2015), and spatialized eutrophication with a $0.5° \times 0.5°$ grid covering the world (Helmes et al. 2012). IMPACT World+ also divides some midpoint indicators into subcategories, with multiple distinct pathways for affecting human toxicity and ecotoxicity. It also specifically accounts for indoor emissions (Wenger et al. 2012) and pesticide residues (Fantke et al. 2011, 2012). Thirty midpoint indicators are thus accounted for in the IMPACT World+ method.
2. The damage calculation is similar to that presented in other methods, where each midpoint impact can cause damage to up to three areas of protection: HH, ecosystem quality, and ecosystem services and resources (which includes societal impacts with no direct health consequences, along with depreciation of ecosystem services). This allows the midpoint contributions to be put into perspective. Table 5.7 summarizes the midpoint-to-damage factors for the main midpoint categories.
3. IMPACT World+ allows users to optionally group certain midpoint categories associated with either climate change or water use, while avoiding double counting with other impact categories.

TABLE 5.7
Main IMPACT World+ Midpoint Categories and Midpoint-to-Damage Factors Relating Each to Damage to Human Health (HH), Ecosystem Quality (EQ), and Ecosystem Services and Resources (ER)

Midpoint Impact Categories	Damage Categories	HH	EQ	ER
Name	Unit	DALY	PDF-m^2-year[a]	$
Global warming	$kgCO_{2\text{-eq}}$	$8.3 \times 10^{-7} + 2.0 \times 10^{-6}$ LT[b]	0.185 + 0.43 LT	
Marine acidification	$kgCO_2$		0.0165 + 0.152 LT	
Land occupation, biodiversity	ha-year arable eq		6000	
Fossil energy use	MJ deprived			0.005[c]
Mineral resources use	kg deprived			Interim
Water use	m^3 deprived	1.8×10^{-4}	0.0020	Interim
Terrestrial acidification	$kgSO_{2\text{-eq}}$		8.32	
Aquatic eutrophication	kg PO_4 P-lim eq		55.3	
Marine eutrophication	kg N N-lim eq		12.5	
Aquatic ecotoxicity (USEtox)	CTU_e*		1	
Toxicity cancer (USEtox)	CTU_h	11.5		
Toxicity noncancer (USEtox)	CTU_h	2.7		
Ionizing radiations	$Bq_{C\text{-14 eq}}$	2.1×10^{-10}	1.9×10^{-10}	
Respiratory inorganics (PM)	$kg_{PM2.5\text{-eq}}$	0.00083		
Respiratory organics	$kg_{NMVOC\text{-eq}}$	3.9×10^{-8}		
Ozone layer depletion	$kg_{CFC\text{-11 eq}}$	0.00176	-	

Note: For example, global warming causes 8.3×10^{-7} DALY (disability-adjusted life years) per kilogram of CO_2 equivalent emitted.

[a] Potentially disappeared fraction of species over 1 m^2 in 1 year.

[b] LT stands for long-term impacts beyond 100 years, so the top number represents impact for the first 100 years.

[c] This midpoint to end point damage factor is substance specific.

*CTU refers to USEtox comparative toxic units, corresponding to potentially affected fraction of species-cubic meter-days per kilogram for ecosystem impacts (CTU_e) and to cases of cancer and noncancer for HH impacts (CTU_h).

The front-end panel case study described in Section 5.3 illustrates the use of IMPACTWorld+.

Normalization is performed at the global level, as described theoretically in Section 5.2.3. IMPACT World+ uses the total global scores of 0.022 DALY/person-year for HH damage, 8,500 PDF-m^2-year/person-year for ecosystem quality damage, and 63,100 MJ/person-year or $313/person-year in resources or ecosystem services, assuming a future increase in energy price due to resource scarcity of $0.005/MJ.

IMPACT World+ does not provide standardized weights, leaving the user with the opportunity to select their own factors. It can be coupled with the weighting factors of the stepwise method derived from budget constraints (Weidema 2009), which attributes the following weights: €74,000/DALY for HH, €0.14/PDF-m^2-year for ecosystem quality and €1/€ for resources.

All publications supporting the creation of IMPACT World+ are provided at www.impactworldplus.org, together with midpoint and damage characterization factors.

5.5.12 USEtox

Developed by a team of researchers from the Task Force on Toxic Impacts under the UNEP-SETAC Life Cycle Initiative, USEtox estimates the fate, exposure, and effects of chemicals. The UNEP-SETAC initiative supports the development, evaluation, application, and dissemination of USEtox to improve understanding and management of chemicals in the global environment.

The USEtox model is not a comprehensive LCIA method but an environmental model focusing on the characterization of human and ecotoxicological impacts of LCIA and comparative risk assessment (CRA). It is presently used to calculate midpoint damage in the European ILCD method, and it has been fully integrated in the midpoint–end point IMPACT World+ method.

Prior to USEtox, a number of different models around the world had been developed to estimate human and ecotoxicological impacts, varying in scope, modeling principles applied, and substances covered. The situation for the LCA practitioner who wished to include chemical-related impacts was that (a) there would probably be many substances in the life cycle inventory for which no characterization factor was available from any of the models, and (b) published characterization factors would often vary substantially among the models.

This unsatisfactory situation led the UNEP-SETAC Life Cycle Initiative to launch a comparison and harmonization of existing characterization models to (Hauschild et al. 2008):

1. Identify which differences in the old characterization models caused the observed differences in their characterization factors
2. Develop a scientific consensus about good modeling practice, based on the identified influential differences
3. Harmonize the old characterization models, removing unintended differences
4. Develop a scientific consensus model based on the learnings from the comparison of the characterization models with the following characteristics
5. Be parsimonious (as simple as possible, as complex as needed), containing only the model elements which were identified as the most influential in the comparison of the existing characterization models
6. Be transparent and well documented

7. Fall within the range of the existing characterization models; that is, not differ more from the old characterization models than these differ among themselves
8. Be endorsed by the modelers behind all participating models

The result of this scientific consensus model development is the USEtox model (Rosenbaum et al. 2008) implemented in Microsoft Excel. The USEtox model calculates characterization factors for aquatic ecotoxicity (Henderson et al. 2011) and for human toxicity (Rosenbaum et al. 2011), focusing on carcinogenic impacts and noncarcinogenic impacts for chemical emissions to urban air, rural air, freshwater, sea water, agricultural soil, and natural soil (Figure 5.13). The unit of the characterization factor for freshwater aquatic ecotoxicity is the potentially affected fraction of species-3-days per kilogram of emissions and for human toxicity is cases per kilogram of emissions, where both are expressed as comparative toxic units (CTU) to stress the comparative nature of the characterization factors.

The fate component is the same for ecotoxicity and human toxicity, and then combined with a human exposure model to describe the transport from environmental compartments to humans via inhalation and ingestion.

In addition to the fate factors and exposure factors, effect factors are required to calculate human-toxicological characterization factors. The effect factor is the change in lifetime disease probability due to change in lifetime intake of a pollutant (cases/kg). USEtox determines effect factors for carcinogenic and noncarcinogenic chemicals separately. Data for effects after inhalation and oral exposure are also

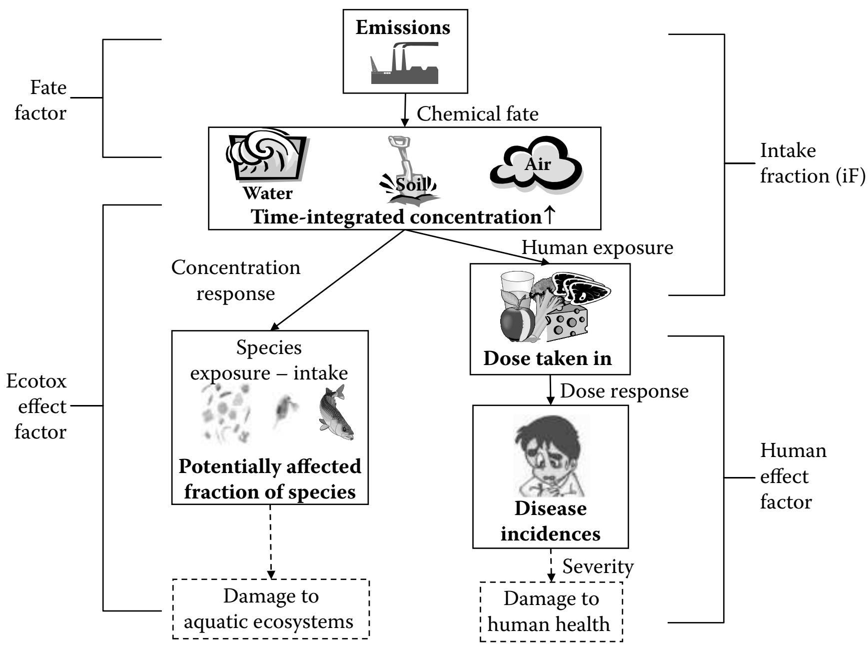


FIGURE 5.13 Main steps of the USEtox assessment.

determined separately. The software, user manual, and results for more than 1000 substances for human toxicity and 2500 substances for aquatic ecotoxicity are available at http://www.usetox.org/. Though not an LCIA method on its own, USEtox is likely to become the default model to screen the toxicity of chemicals in LCIA, with more detailed models applied for complementary advanced and spatialized studies.

5.6 FUTURE DEVELOPMENTS

Since the early critical volumes method, considerable advances have been made in fate modeling and exposure of substances for human toxicity and ecotoxicity (IMPACT World+, ReCiPe), and the first consensus methods are becoming available (USEtox). There are still many impact assessment features that are currently in development or that will be needed in the future.

5.6.1 Further Spatial Differentiation

The fate, exposure, and toxicity of substances depend on multiple spatial aspects. The population exposure to a substance will, of course, be greater in a densely populated area. Also, an emission to water will have very different impacts if it is into a narrow river, upstream of a large lake (high dilution volume, high residence time of water), or into a river that quickly empties into an ocean.

By comparing two models that treated Europe either as one single region or as 130 subregions, Pennington et al. (2005) showed that for the purposes of LCA, a description of the one-box European average is generally satisfactory as a first approach. The cases of emissions near a lake or into a river leading directly to the ocean are the exceptions and should be differentiated in the future.

More generally, LCA developers are continuing to identify a limited number of key characteristics that can cause significant variation in impacts, such as emissions at ground level versus at the height of a chimney, and emissions in areas with high versus low population densities.

5.6.2 Methods for Higher Resolution Life Cycle Impact Assessment

Impacts can differ greatly depending on the characteristics of the emission region, yet products are having increasingly global supply chains, with emissions spanning the world. IMPACT World+ has therefore made an important step forward toward providing country- and region-specific characterization factors. Further developments pursue this effort further to provide more detailed spatialization for toxicity.

The LCImpact European project has, for example, provided advanced characterization in several impact category domains; for example, comparing subcontinental versions of USEtox with the multiregional IMPACT World+ model that accounts for transboundary air advection (Kounina et al. 2014; Shaked 2011). Different scenarios can be run in a model such as IMPACTWorld to provide guidance to decision-makers on the optimal methods for decreasing HH impacts and increasing sustainability in global trade. Such a consumer-oriented analysis provides a unique viewpoint in comparison with other assessments that are typically centered around production. Such a model can also track the fate and exposure of chemicals traveling through food exports, which preliminary studies have shown can be as important as air transport. The LCImpact project will generate a full impact assessment method, incorporating such developments along with other innovations.

At the interface between risk assessment and LCIA, the Pangea multiscale model (Wannaz et al. 2015) enables spatial coverage from a 10 km scale up to a global scale, with high resolution in user-defined regions of special interest.

5.6.3 Substances and Impact Categories

Existing methods only study up to 1000 substances, so UNEP-SETAC is continuing to study the almost 100,000 substances listed in EINECS (European Inventory of Existing Commercial Substances) and the more than 65 million of substances registered in the Chemical Abstracts Service (CAS) system.

Moreover, certain impact categories need to be further developed and refined, with the following new developments to be prioritized:

- Worker exposure is often neglected in LCA but may play a dominant role.
- Near-field consumer exposure may be significant, especially during the use stage, to products such as cosmetics, detergents, and chemicals embedded in furniture or food packaging. USEtox-compatible approaches are presently being developed to provide product-specific exposure models.
- Effects of land use and land transformation are being included.
- Resource consumption, competition, adaptation, and substitution after disposal or use are being further developed, rather than simply considering their extraction from the environment.

Further development of water impacts are needed at midpoint and damage levels. For water resources, as a baseline evaluation, the water stress index (WSI) developed by Pfister et al. (2009) can be used to evaluate potential impacts on HH related to water use. The WSI combines the water consumed (also called *blue water*, in units of cubic meters), with the regional water availability to provide an evaluation of water in competition. Impacts from pollutants emitted into water are already accounted for in the ecotoxicology impact categories and should not be double counted. In the frame of the UNEP-SETAC Life Cycle Initiative, further work is presently carried out within the Water Use in LCA (WULCA) task force to extend this scarcity index to account for both human and ecosystem water needs.

5.6.4 Harmonization of Life Cycle Impact Assessment: The Life Cycle Initiative Flagship Project for LCIA Global Guidance

Initiated in 2012, the LCIA guidance flagship project of the UNEP/SETAC Life Cycle Initiative is providing global guidance and building consensus on environmental LCIA indicators.

As described by Jolliet et al. (2014), the project focuses in the first stage on highly relevant impact categories, such as global warming, health effects of particulate matter emissions, land use, and water use. Consensus on these can be reached by focusing first on selected pathways for which there is common agreement; for example, for biodiversity impacts due to land occupation. Earlier LCIA work carried out within the UNEP-SETAC Life Cycle Initiative, such as USEtox and WULCA, is being used as a starting point for further improvement. In the second stage, the project will address human toxicity, acidification, eutrophication, ecotoxicity, and energy resources. It will also provide recommendations on how to integrate these individual indicators in a consistent framework, ensuring consistency of the indicator selection process and assessment across impact categories. The focus is to reach consensus on midpoint indicators first, while positioning and relating these indicators within a consistent midpoint–end point framework.

EXERCISES

Exercise 5.1: Impact Calculation of Global Warming Impacts

Using the 100-year global warming potentials in Table 5.1, calculate the total impact in the global warming category due to the following emissions associated with 1 kg of the following textile materials: wool, cotton, and nylon (Tables 5.8 through 5.10).

1. What are the relative contributions of CH_4 and N_2O to the total global warming score of these textile materials?

TABLE 5.8
Calculation of Global Warming Impact of Greenhouse Gas Emissions Associated with 1 kg Wool at Farm

Emissions	Unit	GWP	Unit	Equivalent	Unit
61	kg_{CO2}/FU		kg_{CO2}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
0.058	kg_{N20}/FU		kg_{N20}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
1.4	kg_{CH4}/FU		kg_{CH4}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
			Total:		$kg_{CO2\text{-}eq}$/FU

TABLE 5.9
Calculation of Global Warming Impact of Greenhouse Gas Emissions Associated with 1 kg Cotton at Farm

Emissions	Unit	GWP	Unit	Equivalent	Unit
1.2	kg_{CO2}/FU		$kg_{CO2}/kgCO_2$		$kg_{CO2\text{-}eq}$/FU
0.0036	kg_{N20}/FU		$kg_{N20}/kgCO_2$		$kg_{CO2\text{-}eq}$/FU
0.0022	kg_{CH4}/FU		$kg_{CH4}/kgCO_2$		$kg_{CO2\text{-}eq}$/FU
			Total:		$kg_{CO2\text{-}eq}$/FU

TABLE 5.10
Calculation of Global Warming Impact of Greenhouse Gas Emissions Associated with 1 kg Nylon at Farm

Emissions	Unit	GWP	Unit	Equivalent	Unit
6.5	kg_{CO2}/FU		kg_{CO2}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
0.00074	kg_{N20}/FU		kg_{N20}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
0.049	kg_{CH4}/FU		kg_{CH4}/kg_{CO2}		$kg_{CO2\text{-}eq}$/FU
			Total:		$kg_{CO2\text{-}eq}$/FU

TABLE 5.11
Inventory of the Hand-Dryer and Paper Towel Scenarios

Scenarios	Functional Unit	CO_2 ($kgCO_2$/unit)	CH_4 ($kgCH_4$/unit)	N_2O (kgN_20/unit)	PM_{10} ($kgPM_{10}$/unit)	$PM_{2.5}$ ($kgPM_{2.5}$/unit)	NOx (kgNOx/unit)	SOx (kgSOx/unit)
Air-dryer	1 pair of hands dried	1.1×10^{-2}	2.3×10^{-5}	7.2×10^{-8}	1.5×10^{-6}	3.6×10^{-10}	1.4×10^{-5}	4.80×10^{-5}
Paper towels		1.0×10^{-3}	1.8×10^{-7}	4.1×10^{-7}	2.4×10^{-8}	4.2×10^{-6}	3.1×10^{-5}	3.7×10^{-7}

TABLE 5.12
Characterization Factors, Midpoint CF_i

	Comparable Unit	CO_2 ($kgCO_2$)	CH_4 ($kgCH_4$)	N_2O (kgN_20)	PM_{10} ($kgPM_{10}$)	$PM_{2.5}$ ($kgPM_{2.5}$)	NOx (kgNOx)	SOx (kgSOx)
GWP	$kg_{CO2\text{-}eq}$	1	25	298	0	0	0	0
Respiratory inorganics	$kg_{PM2.5}$	0	0	0	0.6	1	0.0077	0.038

TABLE 5.13
Midpoint to Damage; Normalization, and Weighting Factors

	Midpoint-to-Damage Factor	Normalization Factor	Weighting Factor
	MDF	N_k	w_k
Respiratory inorganics	(DALY/$kg_{PM2.5}$)	(DALY/person-year)	(€/DALY)
	0.00083	0.0216	74,000
Global warming	(DALY/kg_{CO2})	(DALY/person-year)	(€/DALY)
	8.30×10^{-7}	0.0216	74,000

TABLE 5.14
Impact Assessment (form)

Air-Dryer – Respiratory Inorganics

	Emission Per FU	Midpoint Characterization Factor	Midpoint Score per FU
Respiratory Inorganics	**(kg/FU)**	**($kg_{PM2.5\text{-}eq}$/kg)**	**($kg_{PM2.5\text{-}eq}$/FU)**
Substance 1: PM_{10}			
Substance 2: $PM_{2.5}$			
Substance 3: NOx			
Substance 4: SOx			
Total midpoint impact ($kgeq_{PM2.5}$/FU):			
Total damage (DALY/FU):			
Normalized damage (pt/FU):			
Weighted damage (€/FU)			

Paper Towels – Respiratory Inorganics

	Emission Per FU	Midpoint Characterization Factor	Midpoint Score Per FU
Respiratory Inorganics	**(kg/FU)**	**($kg_{PM2.5\text{-}eq}$/kg)**	**($kg_{PM2.5\text{-}eq}$/FU)**
Substance 1: PM_{10}			
Substance 2: $PM_{2.5}$			
Substance 3: NOx			
Substance #4: SOx			
Total midpoint impact ($kgeq_{PM2.5}$/FU):			
Total damage (DALY/FU):			
Normalized damage (pt/FU):			
Weighted damage (€/FU)			

Air-Dryer – Climate Change

Climate change	Emission Per FU (kg/FU)	Midpoint Characterization Factor ($kg_{CO2\text{-}eq}$/kg)	Midpoint Score Per FU ($kg_{CO2\text{-}eq}$/FU)
Substance 1: CO_2			
Substance 2: CH_4			
Substance 3: N_2O			
Total midpoint damage ($kg_{CO2\text{-}eq}$/FU):			
Total end point damage (DALY/FU):			
Normalized damage (pt/FU):			
Weighted damage (€/FU)			

Paper Towels – Climate Change

Climate change	Emission Per FU (kg/FU)	Midpoint Characterization Factor ($kg_{CO2\text{-}eq}$/kg)	Midpoint Score per FU ($kg_{CO2\text{-}eq}$/FU)
Substance 1: CO_2			
Substance 2: CH_4			
Substance 3: N_2O			
Total midpoint damage ($kg_{CO2\text{-}eq}$/FU):			
Total end point damage (DALY/FU):			
Normalized damage (pt/FU):			
Weighted damage (€/FU)			

2. Rank the materials according to their global warming scores per kilogram.
3. Discuss what additional information and parameters you need to account for to achieve a fair comparison based on a common functional unit.

Exercise 5.2: Human Health Impacts Due to Respiratory Inorganics and Climate Change

Use the inventory data in Table 5.11 for the hand-dryer comparison (originally described in Exercises 3.2 and 4.2) to estimate the HH impacts for each scenario due to respiratory inorganics and global warming (using GWP 100 years), including midpoint, damage, normalized, and weighted scores. Use the factors available in Tables 5.12 and 5.13 and the available form (Table 5.14).

1. Which scenario leads to a higher score for the respiratory inorganics and for global warming?
2. Which of the midpoint, damage, normalized, and weighted scores can be summed across respiratory inorganics and global warming?
3. What is the best scenario in terms of total weighted HH score?

6 Interpretation

This chapter discusses interpretation, the fourth phase of an LCA. The majority of interpretation is performed after goal definition, inventory, and impact assessment, but should also be applied throughout each phase in an iterative process. This chapter defines the interpretation principle and identifies its key points. Interpretation is illustrated here with an example that compares a desktop computer to a laptop computer. One section is devoted to quality control, followed by a section dedicated to the calculation of uncertainties, an important topic that is still being developed in LCA. This chapter also elaborates upon the social and economic perspectives of LCA to create a more complete assessment.

6.1 INTERPRET! INTERPRET! INTERPRET!

The purpose of the interpretation phase is to identify the life cycle stages at which intervention can substantially reduce the environmental impacts of the system or product, as well as analyze the uncertainties involved. This LCA phase thus enables the analyst to evaluate results, draw conclusions, explain the limitations of the study, and make recommendations, all based on the results of the preceding inventory and impact assessment phases. This phase should provide clear and usable information for decision-making.

To achieve these goals, the interpretation phase involves identification of critical points in the life cycle (e.g., where much of the impact occurs), as well as assessment of the quality and robustness of results using a series of checks (e.g., quality control, sensitivity analysis, and uncertainty analysis). The results from previous phases must be combined with information on data quality, methodological choices (such as allocation rules, limits of the system, and models used), value choices (which could differ for the study and the analysis), and data from similar studies if they exist.

LCA results requiring months of work are often only rapidly and superficially interpreted. To avoid this misallocation of time, we recommend thorough interpretation at all possible levels as follows:

- Interpretation must be conducted in a systematic way for each LCA phase, including after the goal and scope definition, after the inventory of pollutant emissions, after midpoint and damage characterization, and after the evaluation of overall impact. Interpretation is particularly useful for discussing and analyzing the results of the complete inventory before moving on to the impact assessment.
- The contributions of each stage of the life cycle should be compared and analyzed, including extraction and preparation of raw materials, and energy, transportation, manufacturing, use, and disposal.

- The contributions of each system component should be reviewed, such as the CPU, monitor, keyboard, and computer peripherals when analyzing a computer.
- Finally, the respective contributions of each pollutant and extracted substance should be analyzed, identifying which emissions and extractions generate the most impact for each impact category.

6.2 IDENTIFICATION OF ACTION PRIORITIES

The goal of interpretation is to examine various ways of reducing environmental impacts and then identify priorities for taking action. The inventory and impact assessment results are used to identify key points of environmental impact, and improvements are then identified to reduce resource consumption, energy demand, or emissions (Heijungs et al. 1992).

A first step is to focus on the life cycle stages and groups of processes that generate the greatest impact. However, the grouping of processes can be arbitrary and depends on how the system is modeled. For example, the product's manufacturing stage may be divided into five substeps, whereas the use stage is often considered as a single step. In such a case, each individual manufacturing substep appears small, even if the stage as a whole is significant. One should be careful to include large sets of individual processes that each appear to have small impacts but lead to substantial impacts when summed.

In the process of interpretation, we can also focus on life cycle stages that have the highest potential to reduce impacts with limited investment. In some win-win cases, both impacts and costs can be reduced at the same time. In other cases, even a limited low-cost intervention can be extremely efficient in reducing impacts. For example, changing uninsulated doors to well-insulated doors does not drastically decrease impacts, but also may cost little enough to be a very efficient improvement per unit area.

The importance of interpretation is demonstrated by the common practice of companies to study and invest in changes related to their own business operations, even when most of the environmental impacts occur upstream or downstream from these operations. Some agribusiness industries put 80% of environmental efforts into production sites that represent only 20% of the impacts, without analyzing the other 80% of impacts that occur upstream or downstream of the production site. A comprehensive interpretation over the entire life cycle of the product enables companies to improve their own operations as well as their upstream and downstream impacts.

Interpretation also allows optimization of investment over the life cycle, based on a cost–benefit analysis (Section 6.8.3).

6.3 INTERPRETATION EXAMPLE: DESKTOP VERSUS LAPTOP COMPUTER

To demonstrate how to apply interpretation throughout the LCA process, we discuss the comparison of a desktop and a laptop computer. This example, used for instructional purposes, is based on the studies of Tekawa (1998) and Atlantic Consulting

(1998), and the data should not be used as a basis for new projects. These data are then compared with the more recent, extensive, and reliable computer data from the ecoinvent database.

6.3.1 Goal and Scope Definition

The comparison presented here is one meant to inform the development of a sustainable, ecofriendly computer, by performing a life cycle assessment of two different types of PCs. The two studied scenarios are a desktop computer with a cathode ray tube screen and a laptop with a liquid crystal display screen, where each is used for 10,000 hours (Table 6.1). It should be noted that the secondary functions of these two computers are significantly different, with different key characteristics for each computer in term of flexibility of use and adaptation ability. However, since a laptop is often primarily used at a fixed location, it is considered in this study as a replacement of a desktop computer.

The considered function is the processing of information, and the functional unit(FU) is a PC operating for 10,000 hours (Table 6.1).

6.3.1.1 Definition of System Boundaries

The system boundaries are defined to account for all processes that induce more than 2% of the total resource consumption (Figure 6.1). The infrastructure of the building used to manufacture the computer is thus not taken into account. The disposal of the laptop battery was not included in the initial study due to lack of data availability.

6.3.2 Inventory

The emissions inventory (Table 6.2) shows that the desktop computer has consistently higher emissions and energy use than the laptop (by at least a factor of two). The mass of CO_2 emitted over the life cycle of the desktop computer is more than

TABLE 6.1
Functional Unit and Reference Flows for Computer Comparison

Product or System	Function/ Service	Functional Unit	Duration of Service	Reference Flows	Key Environmental Parameters
Scenario 1: desktop PC	Information processing: word processing, calculations, drawing, etc.	1 PC with speed of 200 MHz; medium usage	2,000 h/ year over 5 years	1 desktop with cathode ray tube screen 160 W electricity use	Useful lifetime (until it becomes obsolete) Electricity consumption rate
Scenario 2: laptop PC				1 laptop with liquid crystal display screen 33 W electricity use	

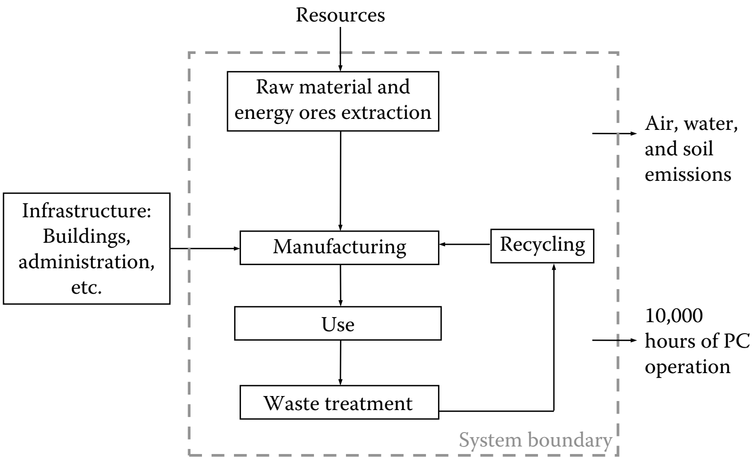


FIGURE 6.1 System boundary for personal computer comparison.

TABLE 6.2
Partial Emissions Inventory

	Desktop (26 kg)	Laptop (3 kg)
Resource Use		
Nonrenewable primary energy (MJ)	23,000	8,500
Emissions to Air (kg)		
CO_2	860	322
CH_4	1.9	0.7
HC	1.5	0.6
NO_x	2.0	0.7
SO_2	5.0	2.1
Pb	0.00011	0.000039
Emissions to Water (kg)		
Pb	0.00018	7.0×10^{-6}

30 times the mass of the computer itself. The energy consumption of the laptop, and thus the many emissions associated with energy consumption, is considerably lower. Because a secondary function of the laptop is to work when not plugged in, its various components are optimized to decrease energy use. In interpreting this inventory, however, it is essential to recall that the system boundary does not include the laptop battery, and thus may neglect key elements such as heavy metal emissions that can greatly affect the overall environmental impact.

As would be expected from the energy-saving design of the laptop, less than half of its nonrenewable primary energy consumption is due to use, compared with use being responsible for three-quarters of the energy consumption in the desktop

(Figure 6.2). The laptop production, however, requires almost as much energy as the desktop production. In both scenarios, the distribution and waste treatment stages consume less than 2% of the total energy.

To further compare the energy consumption in the two scenarios, the manufacturing stage can be divided into the different computer parts, each of which involves material production, parts production, and assembly. Figure 6.3 compares these manufacturing stages for computer monitors and printed circuit boards. The energy for materials production is significant only for the desktop monitor, so this is the only component that has the potential to save energy by being recycled. For other

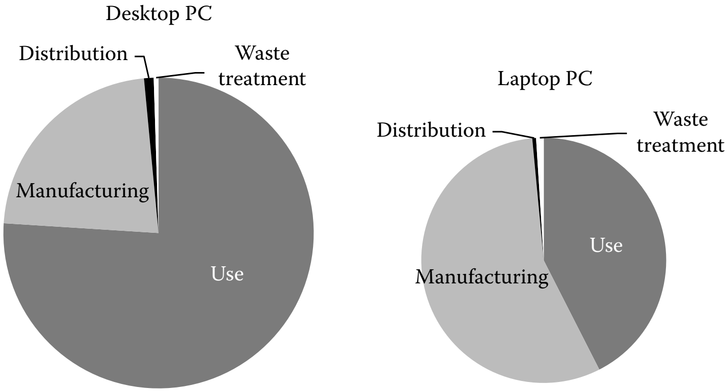


FIGURE 6.2 Primary nonrenewable energy consumption at each stage in the PC life cycle, scaled to indicate absolute consumption in each scenario.

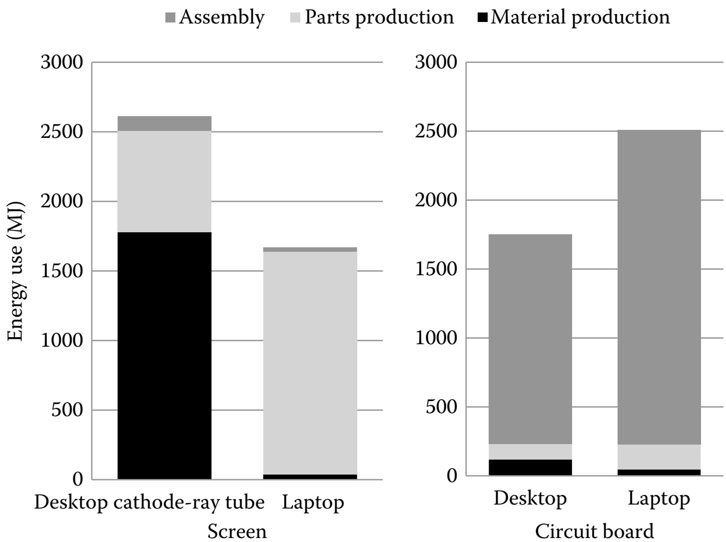


FIGURE 6.3 Primary nonrenewable energy consumption for production of monitors and circuit boards.

components, any recycling benefits would be primarily to prevent the dispersal of toxins into the environment (such as heavy metals).

Both computers also use a large amount of energy for production of electronic components and assembly of circuit boards. The importance of the circuit board assembly stage is likely due to its manufacture and assembly in clean rooms, which require substantial infrastructure and high consumption of energy for air conditioning and ventilation.

6.3.3 Impact Assessment

The desktop computer clearly dominates all the considered impact categories (Figure 6.4), with the contribution of the monitor representing more than half of the impact in every considered category. Although the laptop impact is less than 40% of the desktop impact in all categories, note that the emissions and impacts associated with the laptop battery have not been taken into account and could substantially increase impacts on human health or ecosystems. Furthermore, the service life of both the desktop and laptop has been set at 5 years when defining each scenario, yet a desktop computer generally has a longer service lifetime than a laptop due to its sturdiness and immobility. To further interpret the implications of these assumptions, a sensitivity analysis (Section 6.6.2) on the lifetime of computers could be undertaken.

The interpretation of the impact assessment phase must also note other contributions that have not been accounted for in this study, such as air transportation. An estimated 5% of the impact of a computer is due to the transportation by air during its manufacture (Kaenzig 2003). The impact of air transportation due to a laptop as a passenger carry-on is also not included here, but would further increase this impact. For long-distance transportation by air, every extra kilogram causes additional consumption of 0.004 L of jet fuel per 100 km. If the laptop were transported 40,000 km

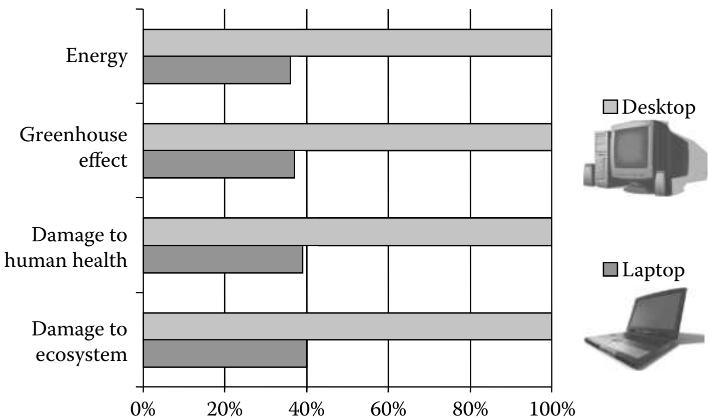


FIGURE 6.4 Impact assessment based on the critical surface-time method. Results are normalized to the desktop PC scenario.

per year by plane, the 200,000 km of transportation over the life of the laptop would induce an additional 954 MJ of energy consumption, or 20% of nonrenewable primary energy.

6.3.4 Assessment Based on Updated Data and Method

This study has been updated using the ecoinvent v.2 inventory data for computer and computer equipment. The laptop battery manufacturing and disposal is included in this update. Figure 6.5 shows relative impacts comparable to those in Figure 6.4, where the laptop does not exceed 40% of the impact of the desktop in most impact categories, except for ozone layer depletion.

Although the relative impacts of the updated analysis are similar to those of the old analysis, the absolute values for primary energy and greenhouse gas emissions differ, demonstrating the importance of using recent data. The new ecoinvent based study estimates 7300 MJ/FU of nonrenewable primary energy and emissions of 370 $kg_{CO2\text{-}eq}$/FU, compared with 23,000 MJ/FU and 860 $kg_{CO2\text{-}eq}$/FU in the old study.

Using the updated ecoinvent data, the impact categories with the highest normalized damages are greenhouse effects, nonrenewable primary energy, respiratory effects, and inorganic carcinogenic effects (Figure 6.6). In all these categories, the laptop clearly has lower impacts.

For both types of PC, the impact assessment identifies the contributions of each pollutant to each impact category, such as the human health category (Figure 6.7). Human health impacts are dominated by primary particulates ($PM_{2.5}$) and secondary particulates (sulfur dioxides transformed into sulfates and nitrogen oxides converted to nitrates). These are common dominant sources of human health impacts.

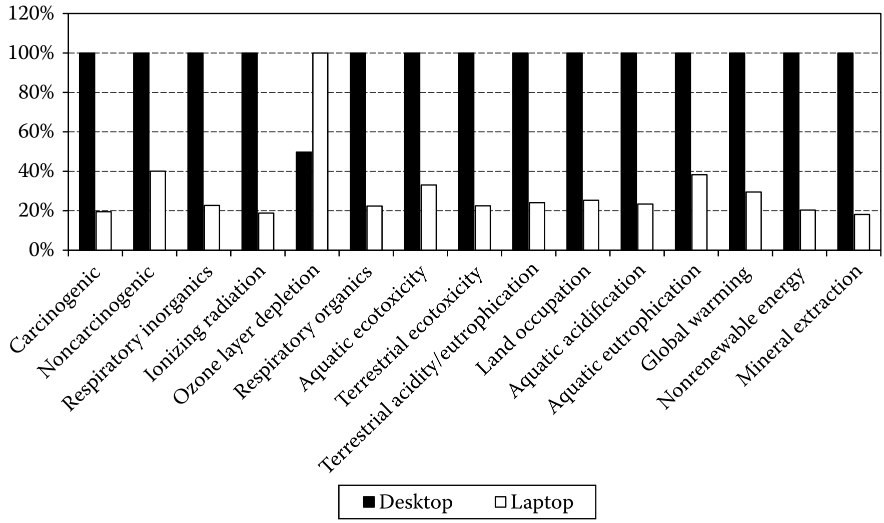


FIGURE 6.5 Comparison of relative midpoint impacts of a desktop and laptop PC, based on the IMPACT 2002+ impact assessment method.

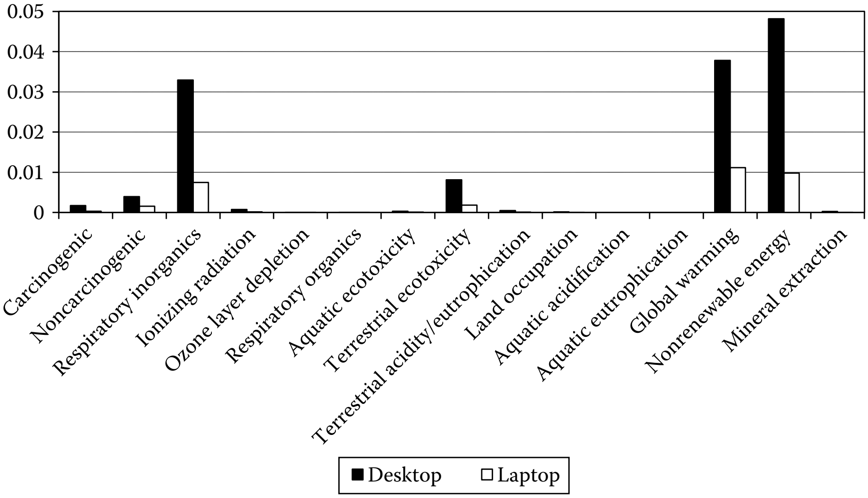


FIGURE 6.6 Normalized endpoint impact scores for the desktop and laptop PC scenarios, based on the IMPACT 2002+ impact assessment method.

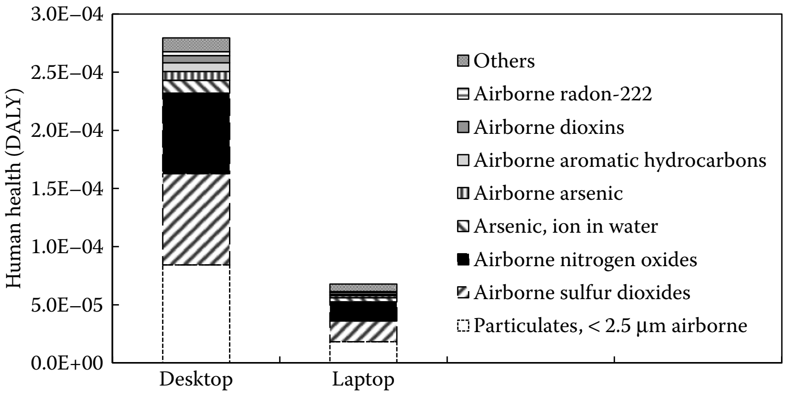


FIGURE 6.7 Contribution of different pollutants to the human health damage, using the IMPACT 2002+ impact assessment method.

6.4 QUALITY CONTROL

Because of the intensive use of data in a life cycle assessment, the interpretation phase includes double-checking and verification at critical points, looking at important data sets and key assumptions. This section presents a series of procedures to ensure the validity of LCA results. Some checks are done at specific life cycle assessment phases, and some are performed throughout the study. The main point of quality control is to verify the consistency of results and look into anything that is unexpected. In the case of the slightest unexpected or surprising result, never let it go—either a mistake has been made (which is usually the case) or you have the opportunity to learn something interesting and new. The analyst has no option but to understand and explain the discrepancy.

6.4.1 Controls at Every Phase of LCA

6.4.1.1 Goal and Scope Definition: System Modeling

A properly conducted study requires a transparent and understandable representation of the system. For this, a systematic flow chart depicts each scenario and systematically numbered modules (see Figure 8.1) to avoid forgetting processes.

6.4.1.2 Inventory Analysis: Unit Control

Errors too often come from careless mistakes, especially when different units are involved in large data sets. To minimize errors, the analyst must systematically check units for each calculation, always carefully accounting for the factor of 1000 when converting between such units as grams, kilograms, tons, megajoules, and gigajoules. Moreover, it is not always legitimate to add two quantities that have the same units. For example, although the normalized scores for human health and ecosystems are both expressed in person-years per FU, they cannot be summed directly without implicitly or explicitly assuming weighting factors for the total normalized impacts of these two categories.

6.4.1.3 Inventory Analysis: Mass Balance

One way to verify inventory results is to check the mass balances of certain elements. The carbon balance is most commonly calculated (see the example in Section 8.2.4) but balances of nitrogen, phosphorus, and heavy metals can also be checked.

6.4.1.4 Inventory Analysis: Energy and CO_2 Balances "by Hand"

Several key steps are needed to establish consistent energy and CO_2 balances for each unit process and across the entire system. First, for each major foreground unit process, we check the assumptions, intermediary flows, energy use per reference unit, CO_2 emissions per reference unit, and finally the contribution of each intermediary flow to the energy and CO_2 emissions per FU (Tables 4.2 and 4.3). Particular attention should be paid to the electricity mix chosen. Indeed, electricity mixes have very different energy efficiencies and CO_2 emission factors depending on their region or country of origin (Section 4.2.2). Because this choice can greatly influence the LCA results, it is important to clarify the assumptions under which calculations have been made.

When calculating the CO_2 balance, do not forget to account for the use stage and the end of life, since a large part of CO_2 emissions occur during the use stage through combustion and during the waste disposal stage through combustion or eventual decomposition. This aspect is often overlooked when using an ecoinvent data set in which combustion is not included and can thus result in substantially underestimating CO_2 emissions.

LCA software programs (e.g., SimaPro) are extremely useful in performing the large sets of calculations necessary for a full LCA, but can also lead to errors when not properly handled. It is thus advisable to check that this software provides the same results as those obtained "by hand" or spreadsheet calculations for a few key substances or flows. If differences concerning energy, CO_2, or NO_x exist, they must be understood and explained.

6.4.1.5 Inventory Analysis: Comparing CO_2 and Energy

One way to check the consistency of the inventory results is to compare energy consumption and CO_2 emissions for each submodule and for the entire FU.

First, the ranking of scenarios based on CO_2 emissions should in most applications be equivalent to the ranking based on nonrenewable primary energy consumption. If they are different, either an error has been made or something new can be learned from this difference.

The ratio of CO_2 emissions to nonrenewable primary energy usage (gCO_2/MJ) is calculated for each life cycle stage. The calculated ratios are then checked against the values of the stage's dominant processes and materials, using the typical values in Figure 4.2 to check orders of magnitude. This step helps the analyst check whether the results are totally absurd, or whether certain major stages in the life cycle were not taken into account.

One example of the importance of such a step is when petrochemical materials and fuels are considered. Figure 4.2 shows that when the whole life cycle is accounted for, including precombustion, usage, combustion, and waste treatment, petrochemical materials and fuels have ratios around 60 gCO_2/MJ. Thus, an emissions-to-energy ratio of 6–10 gCO_2/MJ for diesel or fossil fuels means that the inventory has only accounted for the fuel emissions before combustion. A value of 30 gCO_2/MJ for a plastic means that emissions during the elimination stage were neglected. It is, therefore, important to check this ratio to avoid omissions of life cycle stages and other calculation errors.

6.4.1.6 Inventory Analysis: Comparison of Inventory Results with Other Studies

Similar studies should obviously be taken into account when available. If the inventory results differ from the results of previous studies, these differences and their causes must be highlighted and explained, such as in a comparative table. The origins of these differences must be identified, such as underlying assumptions and FUs, system boundaries, reference flows, coefficients of energy consumption, and CO_2 emissions per flow unit.

6.4.1.7 Impact Assessment: Toxicity Check

Because the calculation of human toxicity and ecotoxicity is still under development and can vary between impact assessment methods, the impact analysis should be performed using several different impact assessment methods, with careful consideration for the contribution of each pollutant. The results obtained by different methods often give different orders of magnitude, and these differences must be explained. It is also crucial to test the robustness of results using a sensitivity analysis (Section 6.5.1).

6.4.1.8 Impact Assessment: Rules for Proper Use of LCA Software

When using LCA software, keep checking that all expected emissions and extractions are considered in the given impact assessment method. Sometimes an emission is not properly included simply because its name is not strictly identical between the inventory results and the impact assessment method (e.g., "nonmethane volatile organic compounds" vs. "nonmethane hydrocarbons," or "PM" vs. "particles" vs. "particulate matter"). Since a given chemical can have hundreds of different names, checks should be based on the chemical's Chemical Abstracts Service (CAS) number when possible.

6.4.1.9 Project Management: Recommended Use of Spreadsheets

Because of the large amounts of data considered, some rules must be followed when using spreadsheets to minimize errors and increase transparency. First, no cell containing a formula should contain a number. All data required for calculation should be entered in separate cells, including unit conversion factors and constants. For example, 1000 g/kg should be in a separate cell and documented as a unit conversion factor rather than entering "1000" as part of a formula. If a given constant is later updated (i.e., a new toxicity study updates the effect factor of particulates on human health), this update will then automatically be reflected in all the formulas in which this constant is used. Second, the units should be clearly indicated for each variable and the units of the final results should be checked. Finally, each calculation must be documented, including the assumptions made, explanation of the variables, and origins of values.

6.4.1.10 Project Management: Rules for Project Documentation

Many LCA projects and data end up as unusable by anyone other than the creator, because the associated computer files are too poorly annotated. To enable further use of these projects, some precautions should be taken.

First, the main results of the report should include enough information for a reader to trace these back to their original spreadsheet(s). This may be a table or separate document listing the files that contain the data for each figure of the report.

Each of these files should be clearly documented, including a descriptive name, the creator, and the date of the last major change.

Similarly, to allow use of the individual processes in future studies, each process should be given an understandable name, with the author and source of information specified. Processes should also be entered in such a way that they can be verified by matching the data format of the original data source.

In summary, the data obtained during the study must be sufficiently and clearly annotated, and the assumptions must be well defined and expressed in a transparent manner for a later use by someone outside the project.

6.4.2 Critical or Peer Review to Check for a Comprehensive and Consistent Study

Once these more practical controls are complete for each life cycle assessment phase, you should check that the assumptions, methods, and data are consistent with the objectives of the study and that the results are comprehensive enough to support a conclusion based on the objectives listed in the goal and scope definition phase.

Called a “critical review,” this ensures that the methods used to perform the life cycle assessment are consistent with ISO 14040, that they are valid from a technical and scientific point of view, and that the data used are appropriate and reasonable regarding the objective of the study. It ensures that the interpretations reflect the identified limitations and goals of the study and that the study report is transparent and consistent. A critical review can be carried out internally or externally, but is always performed by an expert independent of the study. ISO 14040 (pp. 9–11) provides more details on the procedure, and the SETAC code of practice (SETAC 1993) provides key elements of the issues to be discussed in a critical review.

The critical review task should be budgeted for as approximately 5%–10% of the total LCA cost. It is preferable to involve the reviewers as soon as possible to be able to take their comments into account before the end of the project. In addition to the ISO and SETAC guidelines listed above, Klöpffer (2005) helps detail the mandate and the modalities of such a review.

6.5 OVERVIEW OF UNCERTAINTY, VARIABILITY, AND DATA QUALITY

6.5.1 General Principles and Types of Uncertainty

How much confidence can we have in the results of an LCA study? In the case of a comparative study, how do we know when the difference between two scenarios is significant? To answer these questions, it is, firstly, important to know where these uncertainties come from, what types of uncertainty they are, and how to take them into account.

Because of the many judgments made within an LCA, analysis of uncertainty and sensitivity concerning various parameters is critical to understanding the robustness of results. We describe here the types of uncertainty (Huijbregts 1998), how uncertainty of input data can be characterized, and then how to assess uncertainty propagation through the system (Section 6.6). These assessment methods are then illustrated through the instructional example of the front-end panel of a car (Section 6.6.5).

6.5.1.1 Uncertainties in the Four LCA Phases

When conducting an LCA, uncertainties arise based on the choices made during the goal and scope definition, the inventory data of the inventory phase, and the characterization factors and pollutant transport models of the impact assessment phase. Table 6.3 and the subsequent sections briefly describe the various types of uncertainty that occur in these LCA phases.

6.5.1.2 Parameter Uncertainty

Parameter uncertainty is expressed as a distribution of the possible values of a parameter: the probability distribution function (Hogg and Tanis 1993). Examples of such probability distributions are the normal, lognormal, or uniform distributions (Figure 6.8).

In the case of a lognormally distributed variable, the probability distribution function can be characterized by a geometric mean μ and the squared geometric standard

TABLE 6.3
Types of Uncertainty Occurring at Each Phase of the Life Cycle Assessment

		Type of Uncertainty					
		Parameter and Input Data	**Model**	**Choice**	**Spatial Variability**	**Temporal Variability**	**Technological/ Population Variability**
LCA Phase	Goal and scope			FU, system boundaries			
	Inventory	Inaccurate or no input flows and emission factors	Linear instead of nonlinear modeling	Allocation methods, technology level	Variations in intermediary flow provenance	Temporal evolution of emission factors	Differences in technology among factories
	Choice of impact categories and classification		Undefined impact categories, unknown contributions	Leaving out known impact categories, choice of characterization method(s)	Spatial level of detail in method and factors		
	Midpoint and damage characterization	Uncertainty on environmental model parameters	Regional differences in emissions factors	Time horizon considered (e.g., 100 vs. 500 years)	Regional differences in environmental sensitivity	Variation due to seasonal change in temperature	Differences in human exposure patterns
	Normalization and weighting	Inaccurate normalization data	Weighting criteria are not operational	Choice of weighting method	Change in normalization data		Variations in social preferences

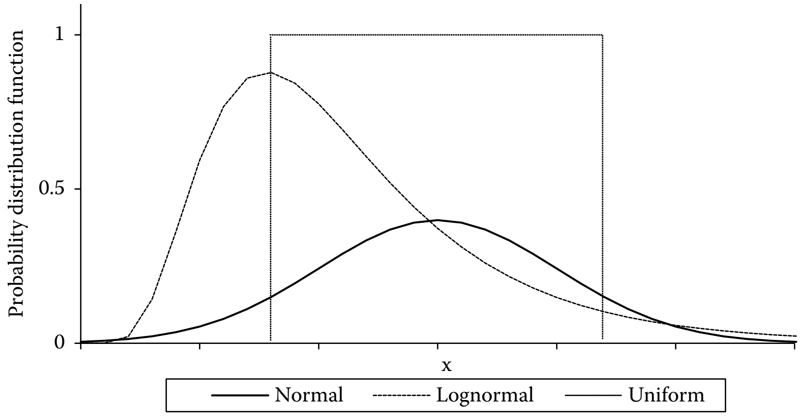


FIGURE 6.8 Schematic representation of normal, lognormal, or uniform distributions, which can be used to characterize parameter uncertainties.

deviation GSD^2. The GSD^2 of a lognormal distribution defines the 2.5th and 97.5th percentiles, which bound the 95% confidence interval around the geometric mean μ (Equation 6.1):

$$\text{Prob}\left(\frac{\mu}{GSD^2} < x < \mu \times GSD^2\right) = 0.95 \tag{6.1}$$

A GSD^2 of 2 means that the parameter has a 95% probability of falling between 0.5 and 2 times the value of the geometric mean μ.

In the specific context of LCA, the lognormal distribution is often applied by default because the parameter values sometimes vary over several orders of magnitude. In this case, the lognormal distribution has the advantage of automatically excluding several impossible scenarios, such as negative emissions or negative uses of processes, which are meaningless in most cases and could lead to erroneous uncertainty estimates.

6.5.1.2.1 Model Uncertainty

To encompass the entire system, some of the system characteristics must be simplified in the modeled system. For example, impact assessment models often assume linearity of environmental processes, without considering possible nonlinear effects. Moreover, certain parameters are calculated based on correlations with associated uncertainties. Bioconcentration factors in foods, for example, are used to estimate the concentrations of substances in the food chain, and they are often calculated based on correlations with the substance's octanol-water partition coefficient (K_{ow}). The best strategy to reduce this type of uncertainty is to take the time to search for and obtain measured rather than extrapolated parameters, focusing on the most influential parameters identified during the screening phase.

6.5.1.2.2 Uncertainty Due to Choices and Assumptions

Despite the LCA rules and guidelines, analysts still need to make certain choices or assumptions, such as selection of the FU, the system boundaries, or the allocation procedure; these can each greatly impact the final result and thus introduce uncertainty.

6.5.1.2.3 Data Variability

In addition to the intrinsic uncertainties discussed above, technological and environmental processes may vary in space and time. When using lower levels of spatial or temporal resolution, or if the location, temporal dynamics, or exact technology is not specified, then this variability can lead to additional uncertainty on the results.

- *Spatial variability:* Inventories and impacts associated with system processes can vary greatly with where these processes occur, yet most LCAs only account for spatial variability in a limited way. This is partly because the spatial distribution of emissions associated with an FU is often unavailable. Every FU involves global supply chains, which make a spatial analysis of associated impacts highly complex. Therefore, calculations for the impact assessment often use generic factors (at a continental level), but new impact assessment methods such as IMPACT World+ provide the option of having factors spatially differentiated by country or by emission type. These more specific factors can only be used if the type or location of inventory data is specified.
- *Temporal variability:* Technologies, their performance, and their impacts change over time. Due to the large data sets considered in an LCA, it is often difficult to properly account for the change in impacts based on when the technology was developed.
- *Technological variability:* Differences between similar processes (due, for example, to the use of different technologies among factories producing the same product) may be another source of variability in the life cycle.

In addition to these uncertainty types, errors can occur at any phase of the life cycle and need to be identified by systematic verification and checking of results (see Section 6.4).

6.5.2 Data Quality and Uncertainty Distribution for Input Data

6.5.2.1 Probability Distribution of an Individual Variable

Figure 6.8 illustrates three types of probability distributions used in the quantification of uncertainty on the parameters of a life cycle assessment. In LCA, uncertainties are often important and represented by lognormal distributions, thus avoiding the possibility of negative emissions. It is rare, however, to actually have sufficient measurements (over 30) to parameterize a lognormal (or Gaussian) distribution. In such a case, uncertainties are estimated based on qualitative indicators that are then transformed into semiquantitative distributions.

6.5.2.2 Quality Indicators

Several quality indicators have been developed (Weidema and Wesnaes 1996; Weidema 1998), including the following:

- The reliability of data is based on the measurement method used and the verification procedures.
- The completeness depends on the representativeness of the available data and the number of companies considered over a given time period.
- The temporal, geographical, and technological correlations indicate whether the place, time, and technology of the collected data correspond well to the process studied.

Each inventory data point is given a qualitative score between 1 (best) and 5 (worst) for each of these indicators. Table 6.4 presents the criteria for assigning scores.

To transform these qualitative indicator scores into a quantitative score, an uncertainty factor can be assigned to each of the pedigree matrix scores using Table 6.5.

A supplementary factor characterizes the *base uncertainty* (U_B), which is specific to certain demands of energy and resources and certain pollutant emissions (Table 6.6). This factor is low for CO_2 emissions because they are mainly due to well-understood combustion processes, but it is relatively high for substances such as heavy metals whose emissions vary with multiple parameters.

The square of the geometric standard deviation (95% confidence interval) of the considered value is then calculated based on Equation 6.2 (as derived from the generalized Equation 6.3 in Section 6.6.4):

$$GSD^2 = \exp\sqrt{\ln(U_R)^2 + \ln(U_C)^2 + \ln(U_G)^2 + \ln(U_T)^2 + \ln(U_L)^2 + \ln(U_S)^2 + \ln(U_\mathrm{B})^2} \tag{6.2}$$

where U indicates the uncertainty factors based on reliability (U_R), completeness (U_C), geographic correlation (U_G), temporal correlation (U_T), technological correlation (U_L), sample size (U_S), and the base uncertainty (U_B).

Taking the example of a process requiring aluminum, let us assume that the data on the necessary quantity of aluminum required by this process have the following characteristics:

- Peer-reviewed and based on measurements (quality score of 1 for reliability)
- Representative of a small number of enterprises and for an adequate time period (quality score of 2 for completeness)
- Obtained less than 3 years prior to the current study (temporal correlation score of 1)
- Coming from a geographical area having similar conditions to the conditions of the case study (geographical correlation score of 4)

TABLE 6.4
Data Quality Indicators with Five Levels of Quality as Described in a Pedigree Matrix

Quality Score	1	2	3	4	5
Reliability	Verified data based on measurements	Verified data partially based on assumptions or nonverified data based on measurements	Nonverified data partially based on qualified estimates	Qualified estimate (e.g., by industrial expert)	Nonqualified estimate
Completeness	Representative data from all sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from > 50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	Representative data from only some sites (< 50%) relevant for the market considered or from > 50% of sites but from shorter periods	Representative data from only one site relevant for the market considered or from some sites but from shorter periods	Representativeness unknown or data from a small number of sites and from shorter periods
Temporal correlation	Less than 3 years of difference to the time period of the data set	Less than 6 years of difference to the time period of the data set	Less than 10 years of difference to the time period of the data set	Less than 15 years of difference to the time period of the data set	Age of data unknown or more than 15 years of difference to the time period of the data set
Geographical correlation	Data from area under studied	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from unknown area or distinctly different area (North America instead of Middle East; OECD-Europe instead of Russia)
Further technological correlation	Data from enterprises, processes. and materials under study	Data from processes and materials under study (i.e., identical technology), but from different enterprises	Data from processes and materials under study but from different technology	Data on related processes or materials	Data on related processes on laboratory scale or from different technology
Sample size	>100, continuous measurements	>20	>10	>=3	Unknown

Source: Ciroth, A. et al. 2013. *International Journal of Life Cycle Assessment*. With permission.

TABLE 6.5
Default Uncertainty Factors (Dimensionless) Applied to Quality Matrix

Indicator Score	Abbreviation	1	2	3	4	5
Reliability	U_R	1.00	1.05	1.10	1.20	1.50
Completeness	U_C	1.00	1. 02	1.05	1.10	1.20
Temporal correlation	U_T	1.00	1.03	1.10	1.20	1.50
Geographical correlation	U_G	1.00	1.01	1.02	—	1.10
Technological correlation	U_L	1.00	—	1.20	1.50	2.00
Sample size	U_S	1.00	1.02	1.05	1.10	1.20

Source: Frischknecht, R. et al., 2004. *International Journal of Life Cycle Assessment*, 10, 3–9. With permission.

- Exactly corresponding to the type of desired aluminum (technological correlation score of 1)
- Obtained from a sample of unknown size (quality score of 5 for sample size)

For a demand of materials, the basic uncertainty factor is 1.05 (Table 6.6). Having determined the default uncertainty factors corresponding to the quality scores defined for each data characteristic (Table 6.5), the variance of the aluminum quantity is then

$$GSD^2 = \exp^{\sqrt{\ln(1.00)^2+\ln(1.02)^2+\ln(1.00)^2+\ln(1.02)^2+\ln(1.00)^2+\ln(1.20)^2+\ln(1.05)^2}} = 1.21$$

Note that the final GSD^2 (dimensionless) does not depend on the required quantity of aluminum.

Once the uncertainty over each individual component is determined as above, these individual uncertainties are combined using Monte Carlo methods or a Taylor series expansion. This yields the overall uncertainty of the inventory flows or impacts per functional unit, as described further in Section 6.6, which also discusses the essential concept of comparative uncertainty based on which parameters are common among scenarios.

6.6 ASSESSMENT AND MITIGATION OF UNCERTAINTY

Table 6.7 describes the appropriate methods for evaluating or mitigating each type of uncertainty and variability. Each method is described in more detail in the subsections below.

6.6.1 Semiquantitative Approaches and Expert Judgment

6.6.1.1 LCA Standardization

For uncertainties associated with choice, standardization processes such as the ISO 14040 series promote sets of default choices that limit inconsistencies between studies and thus reduce the influence of different choices.

TABLE 6.6
Ecoinvent Base Uncertainty Factors (U_B, dimensionless) Applied to the Inputs and Outputs of the Technosphere and the Elementary Flows

Input/Output		Input/Output	
Uncertainty on Intermediary Flows: Demand of:		**Pollutants Emitted to Air:**	
Thermal energy	1.05	CO_2 [combustion, process]	1.05
Electricity	1.05	SO_2 (sulfur dioxide) [combustion]	1.05
Semifinished products	1.05	NO_x, NMVOCs, CH_4, N_2O, NH_3[a] [combustion]	1.50
Materials	1.05	CH_4, NH_3 [agricultural]	1.20
Transportation services (ton-km)	2.00	N_2O, NO_x [agricultural]	1.40
Waste treatment services	1.05	Individual VOCs [process]	2.00
Infrastructure	3.00	CO (carbon monoxide) [combustion]	5.00
Uncertainty on Elementary Flows: Resources:		Individual hydrocarbons, TSM[b] [combustion]	1.50
Primary energy carriers	1.05	TSM [process]	1.50
Metals, salts	1.05	PM_{10}[c]	2.00
Land use, occupation	1.50	$PM_{2.5}$	3.00
Land use, transformation	2.00	PAHs[d] [combustion]	3.00
Pollutants Emitted to Water:		Heavy metals [combustion]	5.00
BOD, COD, DOC, TOC[e]	1.50	Inorganic emissions, others [process]	1.50
Inorganic compounds (NH_4, PO_4, NO_3, Cl, etc.)[f]	1.50	Radionuclides (e.g., radon-222) [process]	3.00
Individual hydrocarbons, PAHs	3.00	Pollutants emitted to soil:	
Heavy metals	5.00	Oil, total hydrocarbons [process]	1.50
Heavy metals [agricultural]	1.80	Pesticides [agricultural]	1.20
Pesticides [agricultural]	1.50	Heavy metals [process, agricultural]	1.50

Source: Frischknecht, R. et al., 2004. *International Journal of Life Cycle Assessment*, 10, 3–9. With permission.

Note: Factors Apply to Combustion, Process and Agricultural Emissions, unless Otherwise Specified in Square Brackets.

[a] Respectively, nitrates (NO, NO_2), nonmethane volatile organic compounds, methane, and ammonium.

[b] Total suspended matter.

[c] Particulate matter smaller than 10 μm.

[d] Polycyclic aromatic hydrocarbons.

[e] Respectively, biological oxygen demand, chemical oxygen demand, dissolved organic carbon, and total organic carbon, defined in Section 4.2.2.

[f] Respectively, ammonia, phosphate, nitrate, chlorine, etc.

TABLE 6.7
Methods for Calculating and Reducing Different Types of Uncertainty

	Types of Uncertainty					
	Parameter Uncertainty	Model Uncertainty	Uncertainty Due to Choices	Spatial Variability	Temporal Variability	Technological Variability
LCA standardization			*			
Expert judgment/independent reviewers	*	*	*	*	*	*
Sensitivity study	**	**	**	**	**	**
Scenario analysis			**		*	
Correlation and regression analysis	*	**				*
Nonlinear modeling		**				
Dynamic or spatialized modeling		*		***	***	
Taylor expansion	***			**	*	**
Probabilistic simulation and Monte Carlo analysis	****			**	*	**
Measurement comparison and additional data collection	****	****				****

Note: The number of stars reflects the adequacy of the method to address the considered type of uncertainty.

6.6.1.2 Expert Judgment and Default Uncertainty Estimates

Many LCAs do not include a comprehensive and detailed uncertainty analysis. Expert judgment has led to generally accepted default rules that can be used and adapted when no other uncertainty analysis is available.

- For energy and CO_2, any difference less than 10% can be considered insignificant at first glance.
- For respiratory inorganic effects or acidification and eutrophication, the difference between two scenarios should typically be greater than 30% to be significant. These percentages have to be adjusted depending on the required quality of the study and according to the impact category.
- For toxicity characterization, the calculation of impacts often involves more uncertainty, requiring a difference of at least one to two orders of magnitude between scenarios to be considered significant. This is especially true if the dominant emissions differ among scenarios, or if they correspond to the long-term emissions of a landfill, where the impacts are highly uncertain. For impact categories with such high uncertainties, it is often appropriate to represent differences in results on a log scale rather than a typical linear scale.
- As described by Rosenbaum et al. (2008), the large uncertainties over the characterization factors for carcinogenic effects, noncarcinogenic effects, and ecotoxicity (one to three orders of magnitude) should be interpreted within the context of the large variation between characterization factors of chemicals (up to 12 orders of magnitude). This means, for example, that contributions of 1%, 5%, or 90% of the total human toxicity impact can be considered equivalent but significantly higher than the impacts due to an emission that contributes less than 1% or less than one-millionth of the total impact.

Failing to account for these variations in characterization factors is the principal cause of misinterpretation of toxicity results in the different impact assessment methods. In calculating life cycle toxicity impacts, we can then identify substances that contribute to, for example, at least 1% of the total score. This generally leads to the identification of 10–30 chemicals to focus on, while the remaining 400 substances can be ignored as not significant to this case study. The 10–30 important substances are not necessarily in the accurate order of their impacts, and further analysis can thus be performed to decrease uncertainties for each substance, including over the relevant stage of the life cycle, the processes responsible for each emission, and the respective importance of the fate, exposure, and effect in the total impact of this substance.

LCA software increasingly integrates methods of uncertainty propagation, which enables a more detailed analysis of uncertainties using a variety of methods described in the following subsections.

6.6.2 Sensitivity Study

The goal of a sensitivity analysis is to test the robustness of results and their sensitivity to data, assumptions, and models used. To do this, we move beyond preconceptions to identify the key parameters that most influence the outcome.

One option is to vary each input parameter by a certain percentage and then examine the resulting percentage variation of the model results. If the goal of the LCA is to compare two products, the sensitivity analysis should relate parameter variations to the difference between the two scenarios. Because many scenarios have common processes, variations in certain processes will not cause significant variations in the differences between scenarios. The variation in differences between scenarios is generally smaller than the independent variation within a given scenario, so this independent variation can be misleading. Another type of sensitivity analysis is to vary the parameters between their reasonable minimum and maximum values and to analyze the impact on the final result.

6.6.2.1 Scenario Analysis

A final option, called a scenario analysis (Huijbregts 1998), is to study the effect of certain assumptions on the outcome of the assessment. For example, we test the importance of the method of allocation (Chapter 4) by comparing how different substitution assumptions or other allocation methods (e.g., financial allocation) affect the final results (see Section 8.5 for another example). In addition, different impact assessment methods involve different assumptions about the aggregation and weighting of inventory results, which introduces uncertainties that can be quantified by a sensitivity study on results from three different impact assessment methods.

From an ecodesign perspective, a sensitivity study identifies the main factors that can be adjusted to improve the environmental performance of a product.

6.6.3 Model Improvement Strategies

6.6.3.1 Nonlinear Modeling

Many LCA models assume linear responses in nonlinear phenomena, which introduces uncertainties that nonlinear modeling can help mitigate. In dose–response modeling used in impact assessment, for example, a default linear dose–response curve is often used, yet in reality, the slope of the dose–response curve may vary depending on background concentrations. With enough available data, this model uncertainty can be reduced by fitting a nonlinear model to the data and then providing factors as a function of background concentration. The danger of such an approach is that nonlinear models can have asymptotes. If an LCA predicts such high impacts that, for example, all species will disappear, a nonlinear consequential approach could suggest that decreasing the load slightly yields no improvement. In such a case, a simpler linear relationship yields the more realistic result that decreasing the load leads to the long-term reduction of impacts.

6.6.3.2 Dynamic or Spatialized Modeling

Model refinements that explicitly account for spatial or temporal variability are increasingly available for both the inventory (e.g., ecoinvent 3.0) and impact assessment phases (e.g., IMPACT World+). Thus, studies with enough information on the location and timing of emissions can reduce the uncertainty of results by accounting for local emission or characterization factors, rather than using generic aspatial or steady-state factors. Chapter 5 discusses the spatial aspects of IMPACT World+ in much greater detail.

When studies consider a limited time horizon (e.g., a 100-year time horizon for global warming), it becomes important to account for the temporal dynamics of both releases and impact assessment factors. In the case of greenhouse gas emissions, six different methods are proposed to account for the temporal dynamics (Brandão et al. 2013). The Levasseur et al. (2010, 2012) approach, in particular, is both rigorous and easy to apply, since the authors provide a freely available Excel tool to account for the temporal dynamic of releases (see website in Appendix I). This tool helps ensure consistency between the dynamics of emissions and impacts in the case of global warming and will thus reduce uncertainties associated with time horizons.

6.6.4 Monte Carlo Analysis and Taylor Series Expansion in LCA

6.6.4.1 Monte Carlo

Because inventory and impact assessment models can be very complex with large amounts of data, it is difficult to algebraically propagate parameter uncertainties to uncertainties on the model results. A Monte Carlo analysis uses a data-intensive method to estimate the uncertainty on the final results by running through thousands of simulations based on the possible input parameter values. When applied correctly, it also determines the significance of a difference between two scenarios.

A Monte Carlo analysis (Figure 6.9) first identifies each model input parameter (p_1, p_2, p_3, etc.) and its probability distribution (e.g., normal, lognormal, or uniform as shown in Figure 6.8), by, for example, using the pedigree method presented in Section 4.3.3. The model result S_i is then calculated by randomly selecting a value for each input parameter ($p_{1,i}$, $p_{2,i}$, $p_{3,i}$, etc.) based on its probability distribution. Repeating this operation many times (typically 1,000 to 100,000 times) yields a set of results S_i that can be statistically analyzed to define the distribution of the final result S.

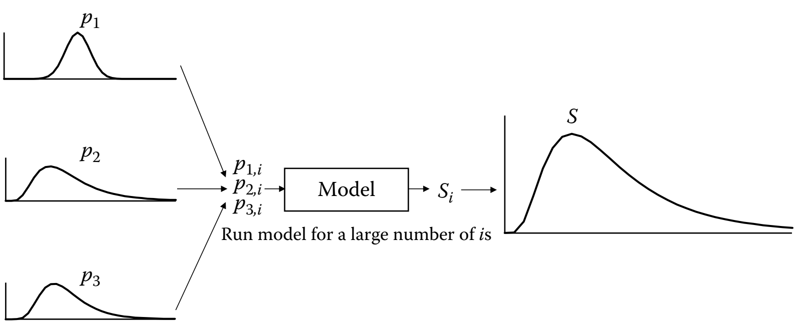


FIGURE 6.9 Uncertainty propagation using Monte Carlo analysis, propagating parameter uncertainties to uncertainties on the model results. The distribution of each model result S is based on a set i of input parameter values (e.g., $p_{1,i}$, $p_{2,i}$, $p_{3,i}$) that are randomly selected from the distribution of each parameter and then repeated for a large number of is.

An LCA is often used to find the difference between two or more scenarios, so one might be tempted to estimate the uncertainty over this difference by calculating the results probability distribution of the results for each scenario and then finding the probability distribution of the difference. This would only be acceptable if every parameter of each scenario were independent, which is never the case in an LCA. Different scenarios always involve common variables, such as the electricity or fuel consumption that occurs in any production line. The characterization factors are also not independent, as they are equivalent for a given pollutant regardless of the scenario.

We use a concrete example to demonstrate this dependence. Imagine that, for a given electricity mix, approximately 5 g of particulate matter are emitted per kilowatt-hour of electricity produced, with an uncertainty of ±4 g (so emissions can range from 1 to 9 g/kWh). In the case of independence between the electricity mixes, we would calculate a distribution of impacts for each scenario based on this range of particulate emission factors. We find that the largest difference in impacts between the two scenarios occurs when Scenario B uses the highest possible particulate emission value (9 g/kWh), and Scenario A uses the lowest emission value (1 g/kWh). In the case of both scenarios using the same electricity mix, it does not make sense to compare results where Scenario B uses a different emission factor for that mix than Scenario A. For variations of such shared input parameters, the two scenarios vary in parallel and the uncertainty is overestimated if the Monte Carlo method is applied successively to each scenario.

To properly compare two scenarios, one must compare the difference or the ratio between the scenarios (Figure 6.10). For a given set of input parameter values, we calculate the score of Scenario A (S_A), the score of Scenario B (S_B), and the relative difference $(S_B - S_A)/S_A$. Repeated simulation using an array of input parameter values yields a distribution of scenario differences, from which we can calculate the probability that $(S_B - S_A)/S_A$ is less than or greater than zero.

The reality may often prove to be between the full independence assumption—that tends to overestimate the uncertainty—and the full dependence assumption that

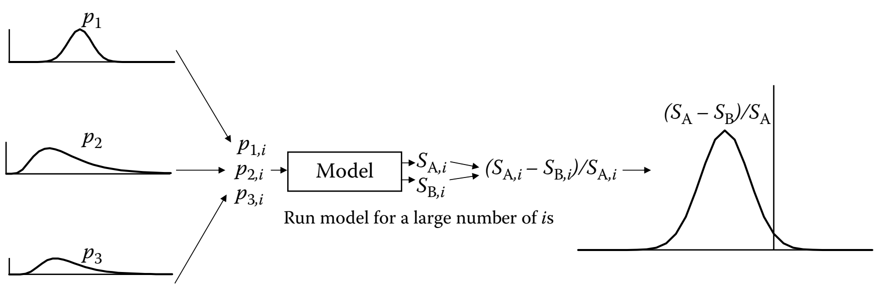


FIGURE 6.10 Distribution of the difference between two scenarios based on Monte Carlo analysis, and accounting for shared input parameters. Parameter values $p_{1,i}$, $p_{2,i}$, and $p_{3,i}$ are randomly selected from the distributions of each parameter and used to calculate relative differences between model results for Scenarios A and B, $(S_{A,i} - S_{B,i})/S_{A,i}$. This yields a distribution of the percentage difference between the two scenario results, assuming the same parameter value for any shared input parameters. When most of the curve lies left of the x-axis, there is a relatively high probability that Scenario B has a bigger impact than Scenario A.

assumes that the same electricity mix is always used in different scenarios—that may underestimate the uncertainty if different electricity sources are eventually used. It is therefore useful to make both calculations, assuming independence and full dependence, to provide an upper and a lower limit to the uncertainty.

6.6.4.2 Analytical Uncertainty Propagation Using Taylor Series Expansion

As an alternative to a more computationally intensive Monte Carlo simulation, Morgan and Henrion (1990) proposed an analytical and transparent uncertainty propagation method using a Taylor series expansion, which was later adapted, described, and applied to a multimedia fate model by MacLeod et al. (2002) and to LCA by Ciroth et al. (2004) and Heijungs (2010). Hong et al. (2010) further developed the Taylor series expansion method, applying it to lognormally based uncertainty analysis to be relevant for scenario comparisons. They applied the developed method to an automobile case study (see Sections 4.2.3 and 4.4.3) to explicitly estimate uncertainty propagation simultaneously in life cycle inventory (LCI) and LCIA. This Taylor series expansion method calculates the geometric standard deviation (*GSD*, defined in Section 6.5) of the output as a function of the *GSD*s of the input parameters.

6.6.4.2.1 Single Scenario

In the case where all input factors are independent from one another, the *GSD* of the output *S* is calculated as a function of the *GSD* of each input parameter *p* as follows (MacLeod et al. 2002, Equation 6.3):

$$\left(\ln GSD_S\right)^2 = s_1^2\left(\ln GSD_{p_1}\right)^2 + s_2^2\left(\ln GSD_{p_2}\right)^2 + \ldots + s_n^2\left(\ln GSD_{p_n}\right)^2 \qquad (6.3)$$

where the relative sensitivity (s_i) of the model output to the input parameter *i* describes the relative change in the model output (ΔS) due to the relative change in this input parameter *i* (Δp_i) from the mean (Equation 6.4):

$$s_i = \frac{\Delta S/S}{\Delta p_i/p_i} \qquad (6.4)$$

6.6.4.2.1 Scenario Comparison

When comparing Scenarios A and B, the geometric standard deviation of the ratio of the two scenarios ($GSD_{\mathrm{A/B}}$) can be expressed as a function of the difference in sensitivity between Scenarios A and B to each input parameter p_i ($s_{\mathrm{A}_i} - s_{\mathrm{B}_i}$) and the *GSD* of each input parameter (GSD_{pi}), as in Equation 6.5:

$$\left(\ln GSD_{\frac{\mathrm{A}}{\mathrm{B}}}\right)^2 = \sum_i \left[\left(s_{\mathrm{A}_i} - s_{\mathrm{B}_i}\right)^2 \times \left(\ln GSD_{pi}\right)^2\right] \qquad (6.5)$$

The degree of confidence that the impact of Scenario A is lower than B is based on the probability that A/B < 1. For a lognormal distribution, this probability can be calculated according to Hong et al. (2010, equation 9, p. 503).

As the case study in the next subsection will show, the Taylor series expansion method gives results similar to those of a Monte Carlo analysis. This is especially the case when one particular process or impact dominates the total impact. When the total impact is a combination of several similarly important components, the two methods deviate more from one another.

6.6.5 Application of Monte Carlo and Taylor Series to Case Study

We will use the automobile front-end panel case study (Section 4.2.3) to illustrate the Monte Carlo and the Taylor series expansion methods, comparing the climate change impacts of steel with that of virgin aluminum. We find that the two different methods predict similar impacts and uncertainties, with uncertainties influenced by which parameters are common and which are considered independent (as described in the previous subsection).

Figure 6.11 presents a simplified flow chart for the steel and aluminum *base cases*. The width of each arrow is proportional to climate change impacts of that process over the product life cycle. Gasoline is a major contributor to the climate change impact in both the aluminum and steel scenarios, and aluminum primary production is another major contributor in the aluminum scenario.

Each input parameter of the LCI has an uncertainty characterized by a lognormal distribution, as defined by its data pedigree within the ecoinvent database (Section 4.3.2, Frischknecht et al. 2005). The LCA software SimaPro 6.0 provides the corresponding GSD^2 for each parameter in each of the ecoinvent unit processes, with 74% of all processes being characterized as lognormally distributed and 26% undefined. SimaPro 6.0 was used to calculate the sensitivity of climate change impacts to each parameter, as well as to run Monte Carlo simulations with 1000 iterations on these climate change impacts. To reflect the 35% uncertainty on the global warming potential (GWP) indicated by the IPCC 2007 report, this study used a squared geometric standard deviation (GSD^2) of 1.35 as a close approximation. For gasoline consumption, we run a low-uncertainty scenario (GSD^2 of gasoline = 1.03) and a high-uncertainty scenario ($GSD^2 = 1.77$) assuming independence between steel and aluminum (i.e., a high gasoline consumption in the steel scenario does not imply a high consumption in the aluminum scenario). For petrol extraction and treatment impacts, we also run a low ($GSD^2 = 1.1$) and a high ($GSD^2 = 2.0$) uncertainty scenario, but assume that these vary in parallel in the steel and the aluminum scenarios.

This study first used the Taylor series expansion method to calculate the two lognormal probability distribution functions of climate change impacts of the steel and aluminum scenarios (Figure 6.12). At first, one might assume that the probability that the aluminum scenario has a higher impact than steel is the fractional area of overlap between the two distributions, which would be the case if the two scenarios used entirely independent sets of parameters. However, since the impact scores of two LCA scenarios are generally based on multiple common LCI processes and LCIA characterization factors, we cannot compare the impacts as if they were independently calculated.

To address the issue of common parameters, we can use a Monte Carlo simulation to calculate the difference in climate change impacts between scenarios based

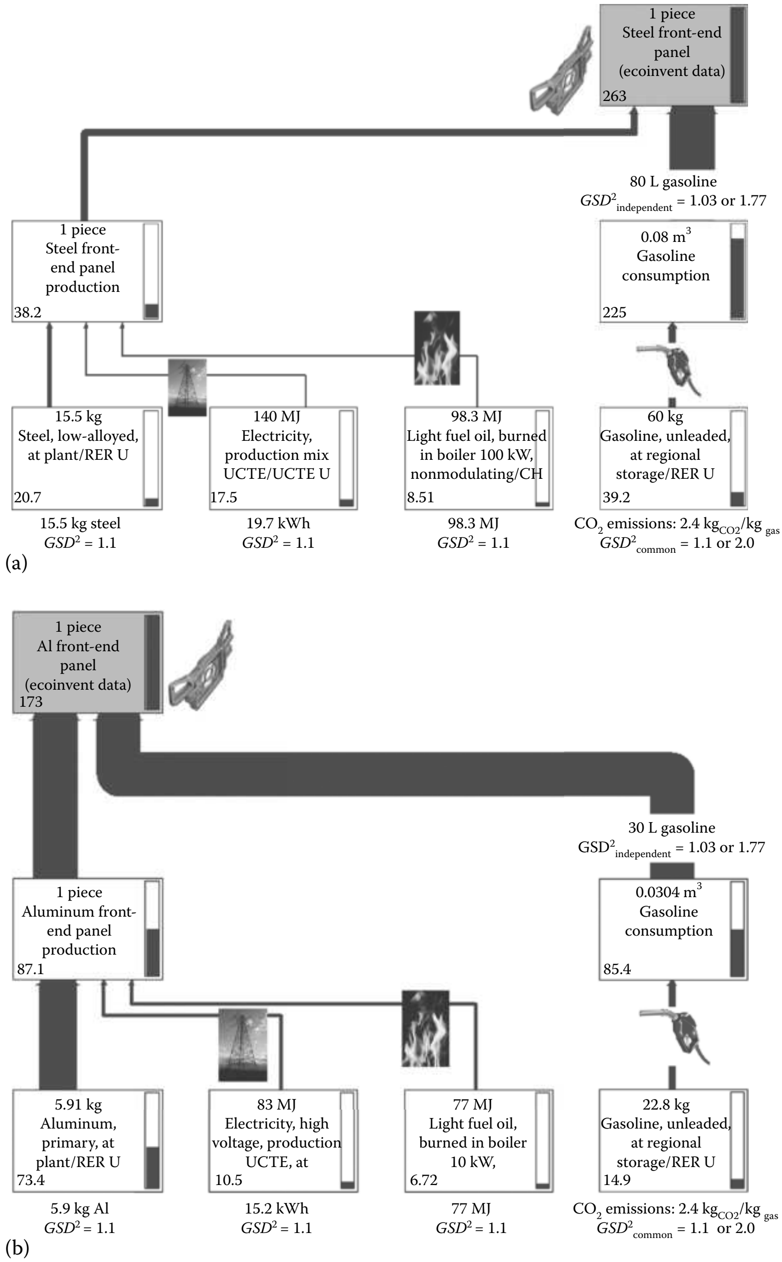


FIGURE 6.11 Flow chart and climate change impacts of the (a) steel and (b) aluminum scenarios according to Hong et al. (2010). Associated squared geometric standard deviations on the uncertainty of the most influential parameters are given as GSD^2.

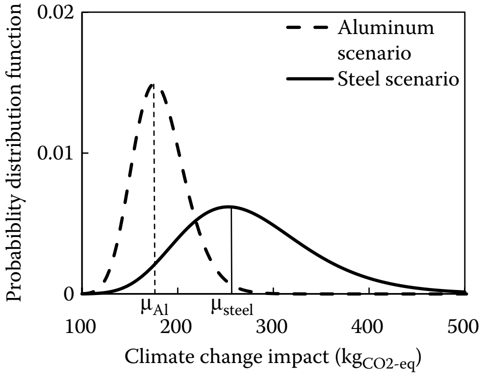


FIGURE 6.12 Lognormally distributed climate change impacts (in units of equivalent kilograms of CO_2) of the steel and aluminum scenarios, using the high uncertainty value on the independent parameter of gasoline consumption ($GSD^2_{\text{independent}} = 1.77$).

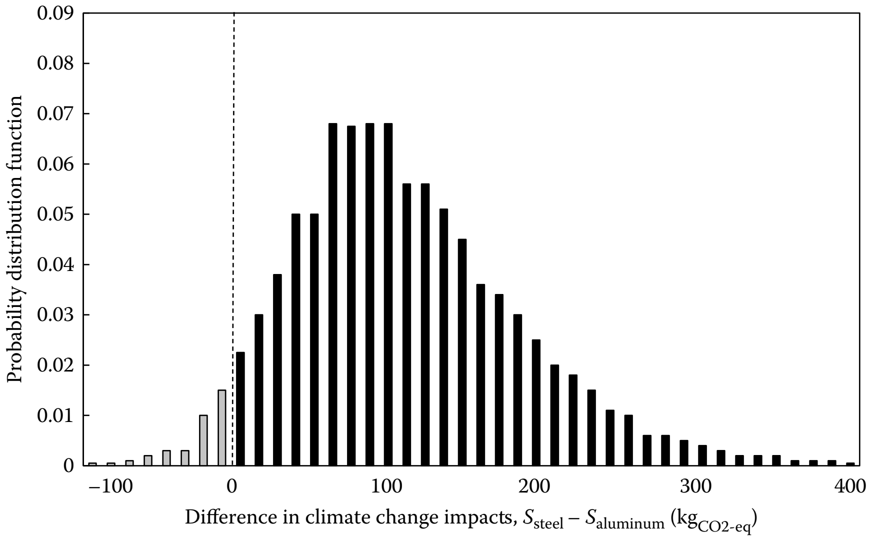


FIGURE 6.13 Probability distribution of the difference between steel and aluminum climate change impacts, using Monte Carlo analysis (independent case with $GSD^2_{\text{gasoline}} = 1.77$ varying independently between the two scenarios).

on the same set of parameters. We repeat the operation a large number of times (e.g., 10,000 times) to determine the probability distribution function of the difference, and ultimately calculate the probability that the steel scenario has a higher climate change impact than the aluminum scenario: P(steel − aluminum > 0). This probability is equal to 7.5%, corresponding to the light gray bars in Figure 6.13.

The Taylor series expansion method can also account for common parameters and calculate the probability that one scenario has a greater impact than another, in a very similar way (Equation 6.5). Rather than plotting the probability distribution function of the differences between the two scenarios, we use the ratios of the two impacts. The probability that the impact of aluminum is greater than that of steel—P(steel/aluminum > 1)—is found to be 7.5% (shaded area of Figure 6.14), which is virtually equal to the probability calculated with the Monte Carlo method.

In addition to calculating the probability that one scenario has a higher impact than the other, the analytic Taylor series expansion method provides the explicit contributions of each parameter to the overall uncertainty (Figure 6.15). For the steel scenario, gasoline consumption contributes most to uncertainty in the climate change impact, followed by light fuel oil consumption**.** These processes contribute more moderately to the uncertainty in the climate change impacts of the aluminum scenario, which also has substantial contributions from electricity production and aluminum primary production. By so easily identifying the sources of uncertainty, future work can be better focused on minimizing these uncertainties and better predicting impacts and comparisons.

This automobile case study illustrates application of the Monte Carlo analysis and Taylor series expansion method to calculate uncertainties in inventory (Figure 6.11) and impact assessment (Figure 6.12), predicting probabilities in comparing the impacts of different scenarios. It addresses the importance of accounting for dependencies on common parameters in LCA, both for common LCI processes and common LCIA characterization factors.

The probability distributions obtained with the Taylor series expansion method are very close to those from a classical Monte Carlo simulation, while being

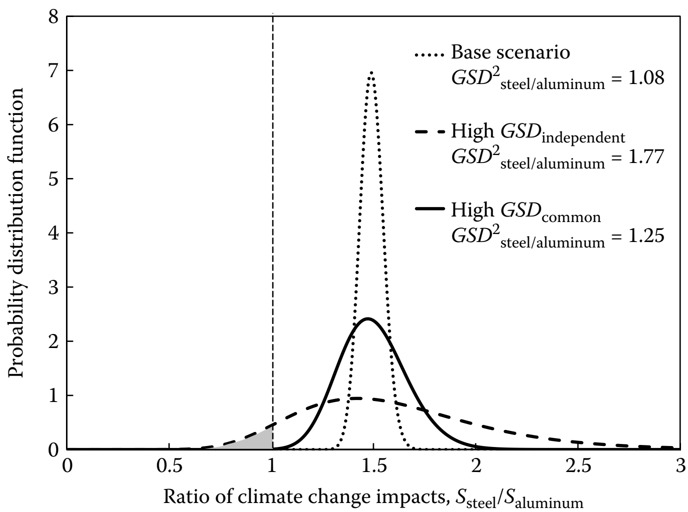


FIGURE 6.14 Probability distributions for the ratio of steel impact score (S_{Steel}) to aluminum impact score ($S_{aluminum}$) according to the Taylor series expansion for the base case (dotted line), for the case where the gasoline impact is considered an independent parameter with high uncertainty (dashed line), and for the case where the gasoline impact is considered a common parameter with high uncertainty (solid line).

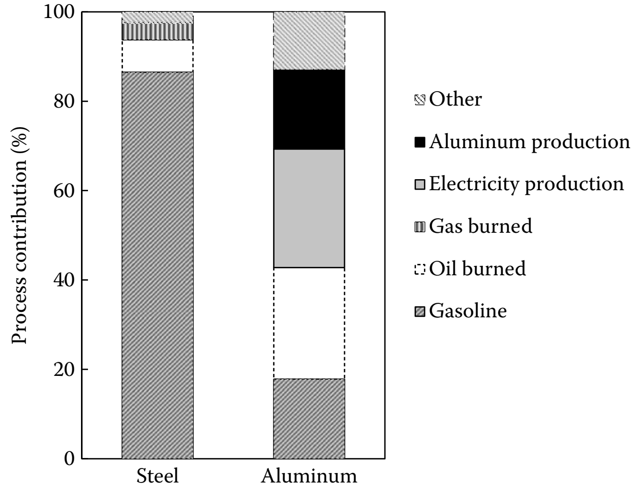


FIGURE 6.15 Process contribution to the overall uncertainty of climate change scores calculated by the Taylor series analytical approach for steel and aluminum.

significantly easier to obtain. Moreover, the Taylor series expansion method provides each parameter's contribution to uncertainty in a very transparent way. Note that results between the two methods are generally similar when there is one dominating process or impact, but may differ more when multiple processes are somewhat equally combined to yield the total impact. One limitation of this application of the Taylor series method is that the input and output parameters must be assumed to be lognormally distributed. Lognormal distributions are quite applicable to LCA, because emission factors, characterization factors, and other parameters are usually positive and can vary over orders of magnitude. Being confined to lognormal distributions for all input and output variables, however, is a significant limitation with implications that must be further explored.

Finally, application of both these methods illustrates that although an LCA can have high absolute uncertainties, it is powerful in comparative settings, where, for example, characterization factors are common among scenarios.

6.6.6 Comparison to Measurements and Additional Data Collection

The final results of an LCA cannot be verified by comparison to measurements, since the impact of an FU throughout its life cycle is not measurable. Certain elements of an LCA, however, can be verified experimentally, either at the unit processes level for inventory, or for the assessment of certain environmental processes and impacts. For the impact of toxic substances and their distribution in the environment, it is possible to evaluate multimedia models (transfer of air, water, soil, food) by comparing model-predicted substance concentrations resulting from the known total emissions of a substance to concentrations measured in water, air, soil, or the food chain.

This approach of comparing modeled concentrations to measured values was used to validate the impact assessment model, IMPACT 2002, based on one of the dioxin congeners (Margni 2003). In this study, the substance properties were defined, and emission data were collected from the literature to be used as the model input. The verification of the IMPACT 2002 model focused on three complementary levels: A first comparison involved the environmental concentrations (air, water, soil), a second was related to the concentrations linked to ingestion (plants, meat, milk, eggs, fish), and the final comparison was related to the fraction ingested by the population (intake fraction). The results were compared with exposure estimates found in the literature.

Another way to improve the quality of the models and parameter estimates is by collecting additional data on the parameters that contribute most to the uncertainty of the results. Correlation and regression analysis can be used to identify and thus prioritize the contribution of individual parameters to the total uncertainty (Huijbregts 1998).

6.7 LCA SOFTWARE

LCA studies clearly involve a large amount of data, making hand calculations tedious, and specialized software valuable for interpretation. Chapters 4 and 5 describe software programs specific to inventories and impact assessment, and many of these same programs are used for interpretation.

The main commercial software programs available for conducting LCA are SimaPro (Goedkoop et al. 2003) and GaBi (GaBi 2003). SimaPro is well designed to simply present and interpret the inventory and impact assessment results, and to easily review detailed contributions of each unit process. V. 7 allows for simultaneous analysis using both the process-based and input–output approaches (Chapter 4), while estimating uncertainty propagation with the Monte Carlo method. GaBi uses more aggregated processes based on industrial data, and is thus particularly relevant for industrial applications in the automotive and electronics sectors and for modeling nonlinear processes. The Quantis Suite software has been developed recently to achieve the balance of a company as a whole and over all of its life cycle (Section 7.3.7).

Several free LCA software programs are also available. A first open-source free program is being created within the framework of the openLCA project (Appendix 1) to provide a modular software program for life cycle analysis and sustainability assessments. Initially, it will begin with a basic framework for LCA calculations of results and uncertainty, along with a tool to convert among different data formats. The Open-IO project (Ciroth 2007) has already released a U.S. input–output database specifically for openLCA. Second, Brightway2 is a powerful recently developed tool allowing analysts to quickly perform cutting-edge calculations and visualizations (Appendix I). Third, CMLCA (Chain Management by Life Cycle Assessment; Heijungs and Frischknecht 2005) is a program intended to support the technical aspects of LCA. Although its user interface is not very flexible, it can be used for rich data analysis, including a complete matrix algebra tool with matrix inversion, as well as integrated methods for sensitivity analysis and uncertainty assessment. CMLCA supports fully hybrid inventories that consist of both process-based and IO-based data, though the comprehensive IO database is not free. The EORA input–output database also offers several assessment tools on its website that can be highly useful in interpreting multiregional IO results (see website listed in Appendix I). Finally, there are various non-LCA-specific tools that use a life cycle approach, such as the carbon tool of the Association Bilan Carbone (BC Bilan Carbone 2010), originally developed by the French Ministry of the Environment (ADEME).

Appendix II provides additional details on these various programs and other available software.

6.8 ENVIRONMENTAL EVALUATION AND SOCIOECONOMIC EVALUATION

LCA can be undertaken for various decision-making applications (Chapter 3), such as whether to accept or select a certain product, and whether to choose a certain policy. Although this book focuses on environmental considerations, decision-makers often also consider economic and social dimensions. These aspects are not part of traditional LCA, which focuses on the environmental assessment, but it is important to see how they can be evaluated consistently with the LCA approach. This section thus first describes life cycle costing and the cost–benefit analysis, followed by a short overview of accounting for social aspects.

6.8.1 Life Cycle Costing

6.8.1.1 Introduction

Just as we can account for the extracted materials or emitted substances over the life cycle of a product or a service, it is also possible to track and analyze the financial flows over its life cycle. This economic evaluation of a product or service over all stages of its life cycle requires knowledge of costs at every stage. The concept of life cycle costing (LCC) can be defined similarly to that of environmental life cycle assessment (Rebitzer 2002). According to Blanchard and Fabrycky (1998), "the life cycle costing refers to all costs associated with the system for a given life cycle." Swarr et al. (2013) provide an LCC code of practice, presenting a comprehensive model of all costs incurred by producers, ownership costs of consumers, and the real costs imposed on other affected stakeholders, with consistent system boundaries aligned with the requirements of ISO 14040.

As is the case for an LCA, financial flows must be considered not only for the production stage, but also for the use and other stages of the life cycle (Figure 6.16). For the total cost of production, the producer must account for the costs of research and development, as well as the actual manufacturing. This is then added to the costs of wages and benefits to yield the sale price of the product that the consumer sees. Since this sale price should include all of the preceding costs, it may not be necessary to examine the upstream costs over the production chain in all detail. The user also generally pays for costs associated with using and disposing of the product, such

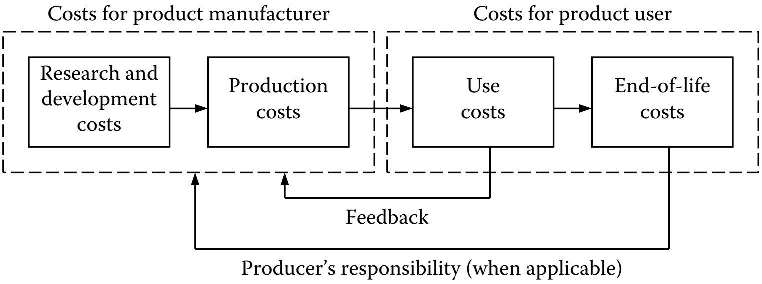


FIGURE 6.16 Life cycle costing distinguishing production and consumption costs of a product over its life cycle. (Adapted from Rebitzer, G. et al., 2003. *Environmental Progress*, 22, 241–249. With permission.)

as the electricity to run a product and any disposal costs. In some cases, legislation requires that disposal costs are included in the purchase.

There could be a feedback from savings in consumer costs back to the producer (Figure 6.16). If the producer offers products that enable the consumer to lower costs during use, they could, in theory, increase the sale price of the product without costing the consumer more over the life of the product. For example, for a €18,000 car driving 15,000 km/year, the fixed costs (cost of car, insurance, etc.) amount to €4000/year and the variable costs (oil, gasoline, services, tires, etc.) amount to €2700 (TCS 2003), of which gasoline consumption amounts to approximately €1000/year. Thus, a car that can save the consumer 20% on fuel consumption represents a saving of €200/year, which corresponds to an initial cost saving of €1500 (for an interest rate of 2% and a term of 7 years). It would, therefore, be theoretically possible for the manufacturer to sell the car for €1500 more without costing the consumer more over the life cycle of the car. In principle, the manufacturer could use this increased income to offset any additional costs involved in developing a more environmentally friendly car. The automotive sector, however, generally follows a different logic, whereby the most expensive cars are often the least fuel-efficient, in which case the user of a car with a big engine is losing doubly in economic terms. In this case, it is different performances or secondary functions that are sought, such as prestige, comfort, aesthetics, and a sense of security. The increase in popularity of low-energy-consumption light bulbs represents a successful example of increasing the initial price of a product while decreasing total life cycle costs.

In general, an LCC includes all costs associated with physical processes, materials and energy flows, labor costs, costs of knowledge use (patents), and transaction costs. We can use the inventory data obtained during an LCA to determine most of these costs. The flows of materials and energy captured in an LCA inventory can be multiplied by the unit prices paid by the company or by the market prices. The costs that cannot be derived from the LCA inventory are those associated with labor and research and development (R&D) of the product or service, so these must be determined separately.

An important and influential choice is the selection of a discount rate to compare costs that occur at different points in time over the life cycle. As discussed by

Asselin-Balençon and Jolliet (2014), we suggest using the effective cost of capital as a default discount rate which is linked to the long-term financing rate. For a producer, this may be equal to the average of the interest rate and the rate associated with the cost of capital as retributed to its shareholders. For a consumer, the long-term interest rate may be more appropriate, but the inflation rate may also be considered from a societal perspective. The internal rate of return (IRR)—the rate that makes the difference in life cycle cost between two scenarios equal to zero—is also a useful metric and well suited for decision-making purposes. The higher the IRR of a scenario, the more profitable it is. The advantage of the IRR is that it is independent of both the chosen discount rate and the interest rate. The IRR can be then compared to the long-term financing rate (e.g., 6.5%) that represents a breakeven situation.

6.8.1.2 Example: Sewage Sludge Treatment and Transport

We present an example of the transport of sewage sludge (Rebitzer 2003) to illustrate an LCC. This study finds the contribution of each stage of the sludge treatment to the total environmental impacts and total cost. More details on the treatment of sewage sludges and their environmental impacts from a broader perspective are presented in Chapter 8.

Figure 6.17 shows the climate change impacts of each stage in the treatment and transport of sewage sludge, where the drying and transport of sludge contribute most

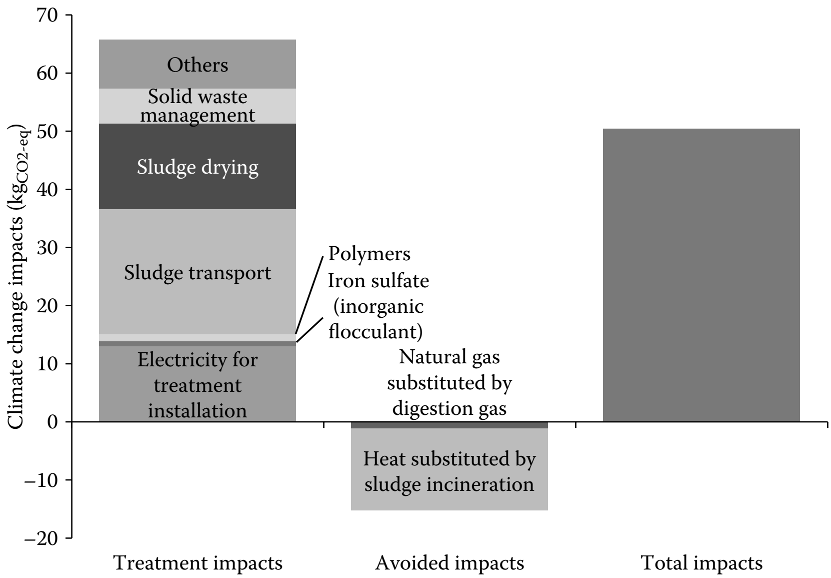


FIGURE 6.17 Climate change impacts (in equivalent kilograms of CO_2) from different elements of sewage sludge treatment and transport, assuming a transport distance of 40 km and a dry solid content under 30% at the exit of the station purification. The third column represents the sum of the impacts of the first two columns (emissions from consumption minus any emissions avoided through substitution). (Adapted from Rebitzer, G. et al., 2003. *Environmental Progress*, 22, 241–249. With permission.)

to climate change impacts. These two stages also contribute most to the life cycle costs (Figure 6.18). Acting on these stages is therefore likely to be both profitable and environmentally beneficial, such as increasing the dry solid content of the sludge at the outlet of the treatment plant, which would decrease the energy needed for both sludge drying and transportation.

A sensitivity analysis on the sludge dry solid content (Figure 6.19) shows that when a flocculating agent is applied to increase sludge dry solid content, there are substantial reductions in the costs of drying and transport. One can save up to 65% when dry solid content is increased from 15% to 45%. Because this increased dry solid content also decreases the transported loads and the energy needed to dry the sludge, climate change impacts are similarly reduced.

6.8.2 Cost Internalization

When the life cycle costs and environmental impacts are known, how can the two be compared? One approach is to express environmental impacts in financial terms, so they can be combined with economic costs. This is commonly referred to as internalization of external costs.

To internalize environmental costs, a cost must be estimated for each environmental impact, including damages to human health and ecosystems. These cost estimates can be based on a variety of approaches, such as costs reimbursed by insurance or someone's

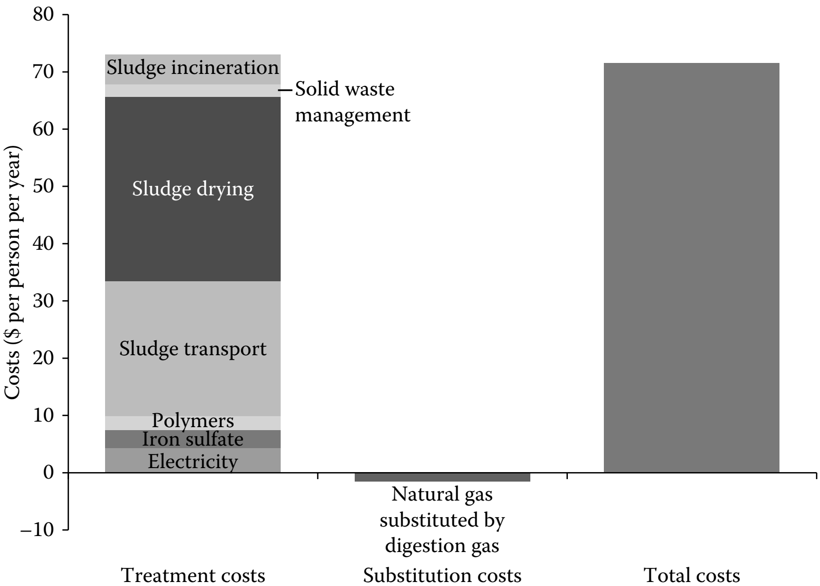


FIGURE 6.18 Financial costs (in U.S. dollars per person per year) from different elements of sewage sludge treatment and transport, assuming a transport distance of 40 km and a dry solid content of the sludge of 30% at the exit of the station purification. The third column represents the sum of the costs of the first two columns. (Adapted from Rebitzer, G. et al., 2003. *Environmental Progress*, 22, 241–249. With permission.)

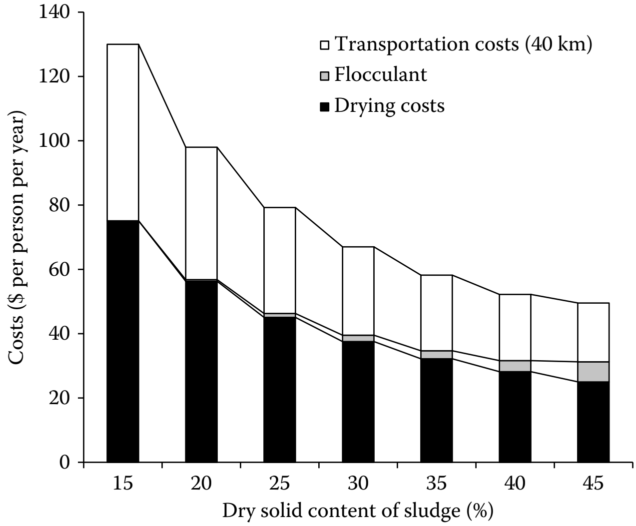


FIGURE 6.19 Dominant costs of municipal sewage treatment and their variation with the dry solid content of sludge. (Adapted from Rebitzer, G. et al., 2003. *Environmental Progress*, 22, 241–249. With permission.)

willingness to pay to avoid this damage. Estimates can account for the costs necessary to deal with an impact or the extra cost generated by the damage (such as through necessary health care), but this risks confounding the measurement of the impact severity with its ease of remedy. In any case, these estimates involve value judgments.

It is possible, for example, to calculate the equivalent cost of pollution from Swiss traffic. One way is to calculate the cost per kilometer driven C (€/km) as in Equation 6.6:

$$C = S_{\text{Human Health}} \times C_{\text{DALY}} \tag{6.6}$$

where $S_{\text{Human Health}}$ is the human health damage score due to traffic, calculated in units of disability-adjusted life years (years lost due to ill health, disability, or early death) per kilometer driven (DALY/km). C_{DALY} is the cost assigned to one year lost (€/DALY).

In 1997, Switzerland had a fleet of 3.3 million gasoline cars and 110,000 diesel cars, with each vehicle traveling an average of 13,800 km per year. The exhaust emissions from such a fleet result in approximately 3600 DALY of human health damage every year (Tauxe 2002). Assuming that society is willing to pay €100,000 to avoid the equivalent of one year of life lost (ExternE 1998), this would correspond to an external cost of €360 million. As a side note from LCC, such a health analysis can be used to compare environmental impacts to more typical impacts, such as accidental traffic deaths. Once we account for the particles emitted from road abrasion and tires, environmental emissions are predicted to result in 12,000 DALY per year, which is the same order of magnitude as traffic accidents (600 deaths and 23,800 DALY).

TABLE 6.8
Comparison of Fuel Cost to Cost Associated with Human Health Impacts

	Fuel Cost	Human Health Cost
Gasoline	8.1	0.7
Diesel	4.5	2.9

Note: € per 100 km of driving.

On this basis, Tauxe (2002) show that, for diesel, the external costs associated with human health damage are of the same order of magnitude as the cost of the fuel itself (Table 6.8).

Some impact assessment methods, such as EPS (Section 5.5.2), directly give the results of the characterization and weighting in monetary values, values that could potentially be summed to the direct costs. However, care must be taken when summing and comparing different types of costs to ensure that adding them is legitimate.

As another example, Fantke et al. (2012) determined the external costs associated with the overall use of pesticides in Europe. They quantified health impacts and related damage costs from exposure to 133 pesticides applied in 24 European countries in 2003, adding up to almost 50% of the total pesticide mass applied in that year. They found that only 13 substances applied to three crop classes (grapes/vines, fruit trees, and vegetables) contributed to 90% of the overall health impacts of about 2000 DALY in Europe per year. Considering the high uncertainties along the full impact pathway, mainly attributable to noncarcinogenic dose–response relationships and residues in treated crops, they obtained an average burden of lifetime lost per person of 2.6 hours. Accounting for the high uncertainties along the full impact pathway, which were mainly attributable to noncancer dose–response relationships and residues in treated crops, they calculated a 95% confidence interval between 22 s and 45.3 d. The costs per person over their lifetime amounted to €12, with a 95% confidence interval between €0.03 and €5142.

6.8.3 Cost-Environmental Performance Representation

Where should you invest $1 to achieve the greatest reduction of environmental impact? Accounting for economic and environmental aspects in parallel can help you choose the scenario that optimizes investment. Figure 6.20 suggests how to depict different scenarios that increase or decrease the environmental impacts and economic costs, by representing the change in cost compared to a reference scenario on the x-axis and the change in environmental performance on the y-axis.

Compared to a reference scenario, four types of alternative scenarios can be defined:

1. The first case corresponds to a "win-win" situation, in which the alternative scenario reduces both the environmental burdens and the costs.

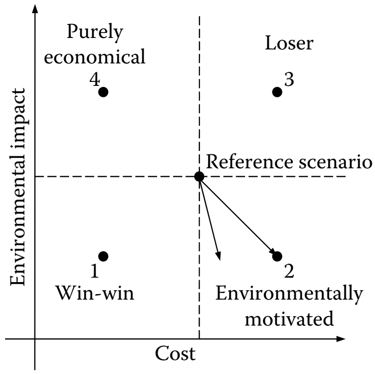


FIGURE 6.20 Representation of different scenarios with respect to a reference scenario, based on the change in environmental impacts as a function of the change in economic costs.

2. The second case corresponds to a scenario with decreased environmental impacts, but increased costs. The vector from the reference scenario to this second scenario would ideally have a steep negative slope. We can perform an analysis to determine which investment provides the greatest environmental improvement per unit investment. To this end, the ecological efficiency η is (Equation 6.7):

$$\eta = \frac{\Delta S}{\Delta C} \tag{6.7}$$

   where:
   Δ S is the reduction in environmental impact score (e.g., $kg_{CO2\text{-}eq}$ or DALY)
   Δ C is the cost increase (in e.g., €)

   This ratio can be calculated to discriminate among different scenarios, but also to determine, within a given scenario, in which life cycle stage to invest in as a priority.
3. The third case corresponds to losing both environmentally and financially, since it leads to greater environmental and economic burdens.
4. Finally, the fourth case corresponds to a financially beneficial but environmentally unfavorable scenario. This corresponds to a purely short-term economic view, as illustrated by the following example comparing different car bumpers (Rebitzer et al. 2001).

This study includes an environmental life cycle assessment and an LCC comparing three types of car bumpers (Figure 6.21).

Compared to the polyamide bumper reinforced with glass fiber, the hemp-fiber-reinforced polypropylene scenario has the lowest environmental impact and lowest cost and is thus a win-win situation. The magnesium bumper is environmentally and economically unfavorable.

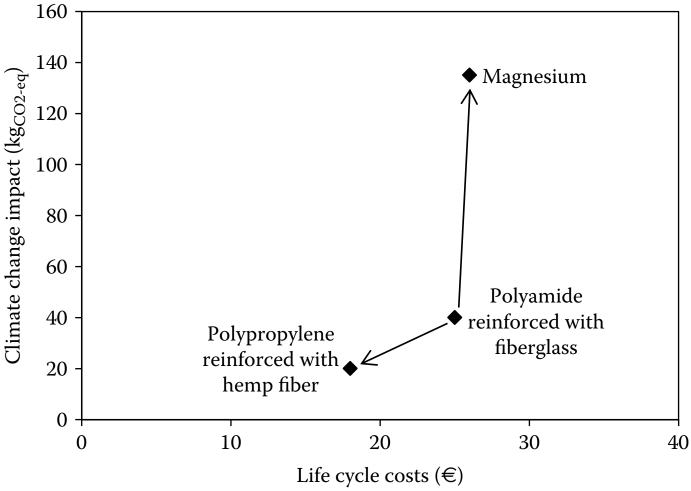


FIGURE 6.21 Relationship between cost and climate change impact for three car bumper scenarios. (From Rebitzer, G. et al., 2001. *Systematische Auswahlkriterien für die Entwicklung von Verbundwerkstoffen unter Beachtung ökologischer Erfordernisse—Abschlussphase (euroMat 2001), Forschungsbericht*, BMBF-Förderprogramm Sicherung des Industriestandorts Deutschland, Projektträger DLR, 637–688. With permission.)

The win-win case (1) seems ideal. In practice, however, many decision-makers and consumers will reinvest this saved money in an equally or even more polluting activity (e.g., a vacation in the Bahamas), making this scenario ultimately result in equal or higher environmental impacts. It is thus important to consider such rebound effects, as described in the following section.

6.8.4 Rebound Effect

As mentioned above, win-win situations only lead to environmental improvements when the financial gain is reinvested in less polluting activities, or rather, used to further reduce environment impacts. Another risk is that because of cost reduction, people will consume more of the goods and therefore limit the environmental gain. For example, as light bulbs become more energy efficient, the light level has increased in parallel in households and streetlamps, thus reducing the environmental benefits (Herring 1999). Dahmus (2014) analyzed the historical efficiencies of the following 10 activities: pig iron production, aluminum production, nitrogen fertilizer production, electricity generation from coal, oil, and natural gas, freight rail travel, passenger air travel, motor vehicle travel, and refrigeration. Over long time periods, improvements in efficiency have not succeeded in outpacing increases in the quantity of goods and services provided, resulting in a sizable increase in resource consumption across all 10 sectors.

In such situations, an LCA can be misleading, since the impacts per FU are reduced, but the ultimate environmental burden could increase. Such effects can be addressed by using a life cycle perspective to analyze the overall impact of a consumer, company, or country, which would then account for all types of consumption, including the reinvestment of saved money (Kaenzig and Jolliet 2007).

Policies can be implemented to help avoid such rebound effects, in which standards or incentives are given to ensure that financial benefits from win-win solutions are reinvested to further reduce environmental loads.

6.8.5 Accounting for Social Aspects

Using the input–output method described in Section 4.4, it is possible to quantify the creation of jobs throughout the life cycle of a product or for an investment in a given industry (Frei 2000). This method replaces the emissions per euro from each sector by the number of jobs created per euro; thus, the number of jobs created per FU is calculated as in Equation 6.8:

$$t = \mathrm{P}(\mathrm{I} - \tilde{\mathrm{A}})^{-1}\tilde{y} \tag{6.8}$$

where:

- t is the vector of created jobs
- P is the employment matrix per monetary unit invested in a sector
- $\tilde{\mathrm{A}}$ is the economic matrix expressing the expenses of each sector in every other sector
- $\tilde{y}$ is the demand expressing the amount spent in each sector per FU

This method was applied in a study conducted by Corbière-Nicollier and Jolliet (2003) to assess the environmental, economic, and social sustainability of various regional actions. This study determined the CO_2 emissions, employment, and added value generated by an increase in investment of SF1 million in one of various economic sectors of the canton of Vaud in Switzerland. Figure 6.22 presents the resulting impacts for three sectors: food and lodging, communication, and construction.

Investing in the food and lodging sector would result in the greatest CO_2 emissions (66 tons/year), but would also create the most jobs (15 persons/year). The added value does not vary much among sectors.

In 2009, a working group published guidelines for the social life cycle assessment (SLCA) of products (UNEP 2009). This was done through the UNEP-SETAC Life Cycle Initiative, and in partnership with the International Reference Centre for the Life Cycle of Products, Processes and Services (CIRAIG), the Québec Action Fund for Sustainable Development (FAQDD), and the Belgian Federal Public Planning Service. The objective of a social LCA is to analyze and improve social and environmental performances of human activities to contribute to greater profitability and greater well-being in the long term. The guidelines describe the history, key concepts, and scope of application of various tools, focusing on the concepts of sustainable production and consumption, as well as corporate social responsibility. The SLCA includes the main phases of the classic LCA according to ISO 14040 and 14044, modified to integrate social and socioeconomic considerations.

Building on this work, Table 6.9 presents the impact categories in a manner similar to the LCIA framework of the Life Cycle Initiative (Jolliet et al. 2004, table 1),

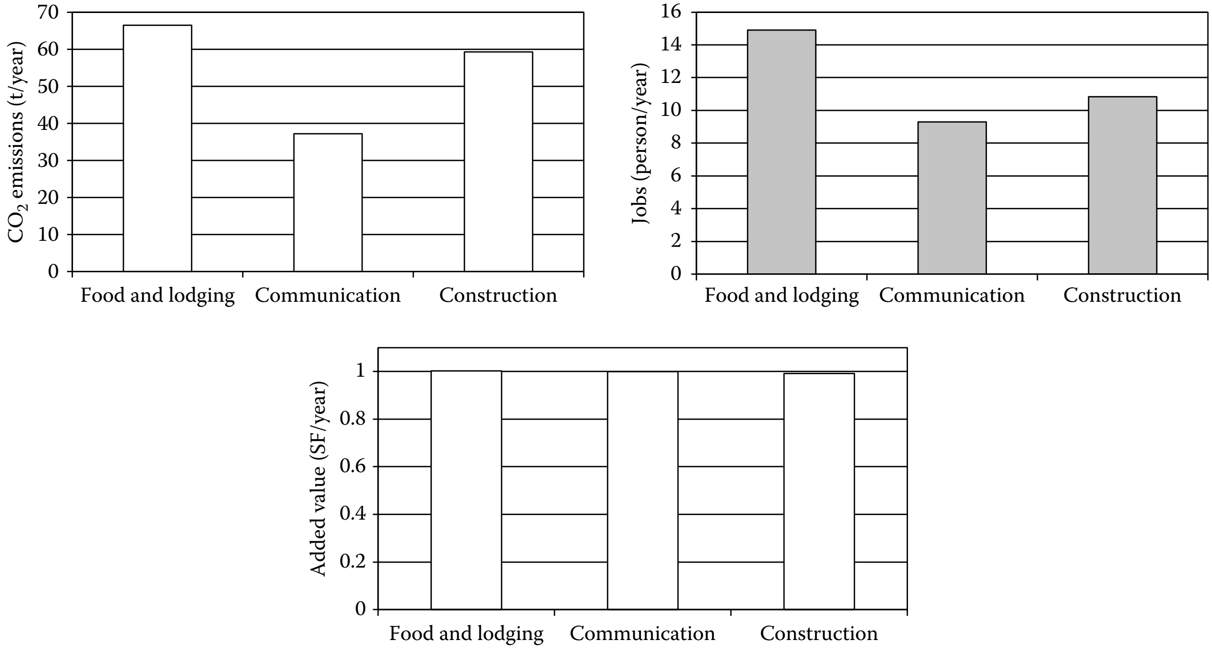


FIGURE 6.22 Impact on CO_2 emissions, employment, and added value of the investment of SF1 million in the canton of Vaud (Switzerland) in three economic sectors (food and lodging, communication, and construction).

TABLE 6.9
Table to Indicate the Types of Impact (Row) and Damage (Columns) Categories Considered in Social LCA

		Damages Related to Intrinsic Values		Damages Related to Functional Values	
		Human Health, Safety + Psychological Health	Human Rights + Equality: Society + Workers	Governance	Socioeconomic Repercussions
Human health and environmental impacts	Human toxicity, societal and workers	⊗			
	Human toxicity, worker	⊗			
	Casualties, societal	⊗			
	Casualties, worker	⊗			
	Freshwater depletion	⊗			
Labor and production	Local employment				⊗
	Fair salary		⊗	**x**	x
	Working hours		⊗		
	Forced labor		⊗		
	Child labor		⊗		
	Gender equity		⊗		
	Labor and company governance		⊗	⊗	
Local community and social life	Access to resources	x			(x)
	Education				
	Community engagement				(x)
	Interpersonal relations	x			
	Indigenous rights		⊗	x	
	Societal private + public governance			x	x

Note: "⊗" indicates links that could be quantitatively modeled, "x" indicates other links, and "()" indicates links that are only relevant if the corresponding damage category were included in the LCA defining impact categories grouped by stakeholder along the life cycle in rows and damage categories or area of protection in the columns. Further work is needed in this area, but the attributional LCA approach proposed by Norris (2006) and its adaptation to social LCA by Benoit-Norris et al. (2012; also see the Social Hotspots Database listed in Appendix I) represent an interesting initial operationalization toward a more complete social LCA.

EXERCISES

Exercise 6.1: Comparing Hand-Drying Scenarios Using LCA Software

You will now use LCA software (e.g., SimaPro) to calculate results that can be compared to your previous hand calculations from Exercises 4.2 and 5.2.

- Step 1: Enter data for the two scenarios in the software (see guidelines for performing calculations in SimaPro on the CRC Press website, Exercise 6.1) and calculate the emissions inventory and impacts.
- Step 2: Compare the energy use, CO_2 emissions, and other inventory flows of the two scenarios by using an impact assessment method (e.g., IMPACT 2002+).

Answer the following questions:

1. Compare your software results for energy use and CO_2 emissions to the hand calculations from the previous exercise, and explain any discrepancies. Fill in a copy of Table 6.10 for each scenario.
2. Based on the software results, which scenario is better from an energy use perspective? What about from a global warming perspective?
3. Use the software to determine which scenario results in more lead emissions.

The scenarios can be built based on Figure 6.23. Note: To facilitate later work (sensitivity analyses, etc.), we recommend you create a process based on the measuring unit considered (one towel, 1 MJ, one apparatus) and then adjust the reference flow of each process to apply to the considered FU. For detailed instructions on how to build these scenarios in SimaPro, see the CRC Press website, Exercise 6.1.

Exercise 6.2: Determining Uncertainty with Pedigree

For the $PM_{2.5}$ emissions associated with 1 kg liquid aluminum, the emission factor amounts to 0.0026 kg; and the uncertainty pedigree characterizing this emission flow is (1, 1, 2, 1, 1, 3) corresponding to, respectively, (U_R, U_R, U_G, U_T, U_L, U_S), for a lognormal distribution.

1. Obtain the base uncertainty U_B for $PM_{2.5}$ from Table 6.6.
2. Determine the default uncertainty factors U_R, U_R, U_G, U_T, U_L, U_S scores using Table 6.5.

TABLE 6.10
Table of Values to Calculate and Compare

Life Cycle Stage	Hand Calculations		Software Calculations		Differences		
Process (unit)	**Energy (MJ/unit)**	**Energy (MJ/FU)**	**Energy (MJ/FU)**	**CO_2 (kg/FU)**	**Energy (MJ/unit)**	**Energy (MJ/FU)**	**Comment**
Materials							
Cast iron (kg)							
Steel (kg)							
Transport							
Transport (ton-km)							
Use							
Electricity (kWh)							
Waste							
Landfill							
Total							

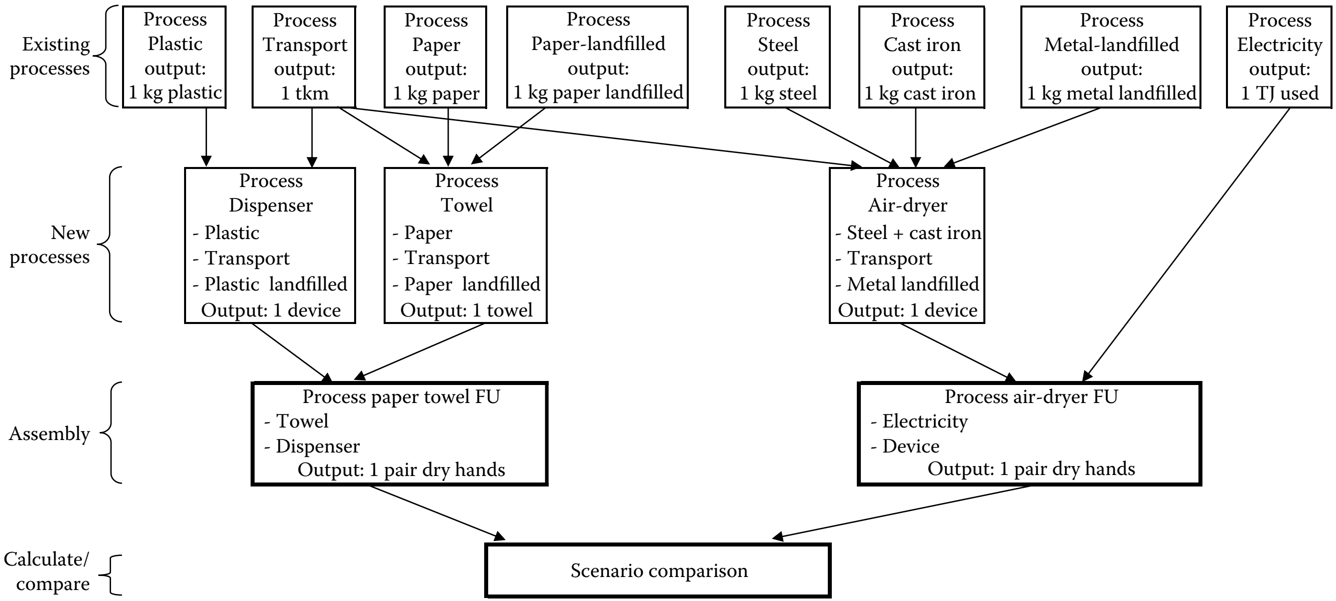


FIGURE 6.23 Diagram of useful processes for the comparison of hand-drying scenarios in LCA software.

3. Calculate the GSD_0^2 for $PM_{2.5}$ using Equation 6.2.
4. Interpret the obtained GSD_0^2, providing the 95% confidence interval on the $PM_{2.5}$ emission factors of 0.0026 kg $PM_{2.5}$ per kilogram of liquid aluminum.

Exercise 6.3: Interpretation of Steel Front-End Panel

You have been hired by Tesla Motors to work in collaboration with the material science department to develop new front-end panels. A front-end panel is a structural component that bears other elements and equipment such as the headlights and the radiator grill. Use the results presented throughout the book on the front-end panel scenarios to make recommendations for future designs.

First, you would like to determine which substances contribute most to human health impacts in the *steel front-end panel scenario.*

1. Use Figure 5.6 in Chapter 5 to determine the two midpoint categories that contribute most to the human health damage considering both short- and long-term damage.
2. Use Table 6.11 to calculate the following quantities:
   - Midpoint impact for each substance
   - Total impact for each midpoint impact category
   - Total damage for human health
   - Normalized damage for human health
3. Assume carcinogenic impacts generally have a factor 100 uncertainty (meaning that the actual value ranges between 100 times higher and 100 times smaller) and respiratory inorganic impacts have a factor 10 uncertainty. Which substances should be treated as having significant and equivalent damage in this case, and which can be considered as having a negligible contribution?
4. In Chapter 4, the CO_2 emissions of the steel scenario are calculated as follows:

   1.28 $kgCO_2$/kg steel × 10.0 kg steel/FU + 0.45 $kgCO_2$/kWh electricity × 19.7 kWh electricity + 3.67 $kgCO_2$/kg oil × 2.3 kg oil + 2.80 $kgCO_2$/L gas × 80.0 L gas + 0.01 $kgCO_2$/kg steel landfilled × 10.0 kg steel landfilled = 253.9 kg CO_2/FU

   Use this calculation to determine the priority of each of the following ecodesign actions: recycling, extending lifetime, increasing motor efficiency, reducing component weight.
5. Using Figure 4.3, discuss how these priorities can change depending on the distance traveled during the lifetime of the vehicle.

Exercise 6.4: Comparison of Two Front-End Panels

You are comparing two front-end panels for a car—one made out of aluminum and one made out of steel.

TABLE 6.11
Midpoint Impacts for Pollutants Emitted to Air and Damage Factors

		Midpoint Impacts	
Pollutant	**Emission (kg)**	**Carcinogenic Characterization Factor ($kg_{C2H3Cl-eq}$/kg)**	**Respiratory Inorganics Characterization Factor ($kg_{PM2.5-eq}$/kg)**
NO_x	0.22	0	0.127
$PM_{2.5}$	0.038	0	1
SO_2	0.44	0	0.078
Aromatic hydrocarbons	0.0041	3537	0
Dioxin (2,3,7,8 TCDD)	77.4×10^{-12}	1.72×10^{9}	0
	Normalization Factor	**Damage Factors by Category**	
	(DALY/pt)	**(DALY/$kg_{C2H3Cl-eq}$)**	**(DALY/$kg_{PM2.5-eq}$)**
Human health	0.0071	2.8×10^{-6}	7.0×10^{-4}

1. Figure 6.24 presents the probability distribution function of total primary energy use for the aluminum front-end panel. The median primary energy use is 2780 MJ/FU, and the GSD^2 of this distribution is 1.6. What are the upper and lower limits of the 95% confidence interval?
2. Figure 6.25 presents the frequency distribution of total primary energy use for the steel front-end panel. The median primary energy use is 4400 MJ/FU, and the upper and lower limits of the 95% confidence interval are 5060 and 3830 MJ. Which scenario is better according to the information of Parts a and b—the steel or aluminum (notice the difference in the *x*-axis range)?
3. Figure 6.26 shows the distribution of energy differences between the two scenarios (steel minus aluminum). The median energy difference between the two scenarios is 1620 MJ (i.e., steel uses 1620 MJ more than aluminum). Given the high uncertainty over the energy used for the two scenarios, one would expect the difference in energy between the scenarios to also be widely distributed. Why is the distribution of energy difference narrower than the distribution for each individual scenario? Using the information in the graph, describe how you could approximately calculate the probability that one scenario is better than the other (in two short sentences).

Exercise 6.5: Monte Carlo Analysis and Taylor Series Expansion

Instead of using a Monte Carlo analysis, you can use the Taylor series expansion approach (Equation 6.3) to analytically calculate the uncertainty distributions of the preceding exercises.

The steel component requires 15.4 kg steel/FU for its manufacturing, with an emission of 2.39 kg CO_2/kg steel. Assume a GSD^2 of 1.1 for these factors.

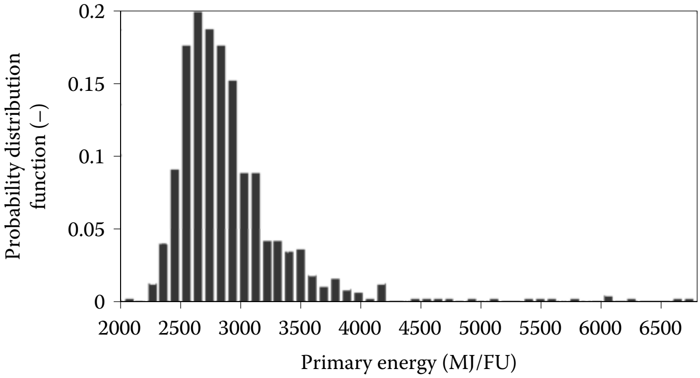


FIGURE 6.24 Probability distribution function of primary energy use for aluminum front-end panel.

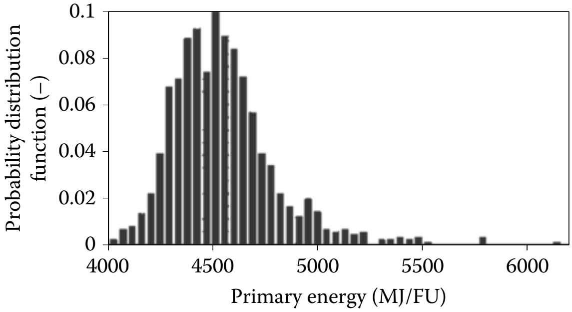


FIGURE 6.25 Probability distribution function of primary energy use for steel front-end panel.

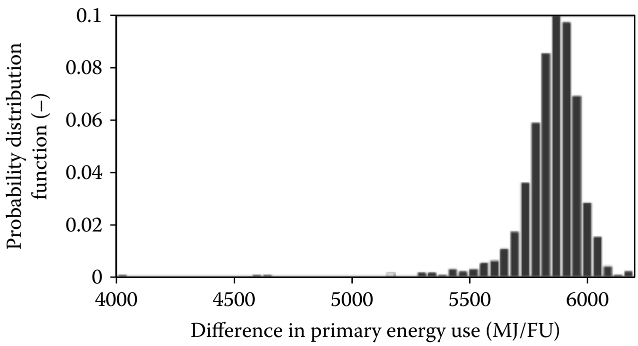


FIGURE 6.26 Probability distribution function of the difference in primary energy use for the two scenarios.

The fuel consumption per marginal weight change is further assumed to be equal to 0.00004 l/kg-km over 200,000 km; thus, an 80 L gasoline consumption/FU (assume GSD^2 of 1.1), with an emission of 2.8 kgCO_2/L gasoline, which is assumed to have a higher uncertainty with a GSD^2 of 1.77.

1. Calculate sensitivities: Assume steel manufacturing and gasoline consumption are the only contributions to CO_2 emissions over the life cycle of the front-end panel. Calculate the percentage contributions of manufacturing and gasoline consumption to the total front-end panel CO_2 emissions. These percentages are equivalent to the sensitivities of the output results to the reference flow and emission factor input parameters for manufacturing and gasoline consumption.
2. Use the four sensitivities calculated in Part a and the GSD^2s provided above to calculate the GSD^2 of the output, and compare this value to the result of Exercise 6.3a.

7 Conclusions and Key Points

7.1 KEY POINTS OF A LIFE CYCLE ASSESSMENT

Life cycle assessment (LCA) requires an iterative approach through various phases, in which one first performs a preliminary evaluation of the system to determine the order of magnitude of the different contributions, and then performs a more detailed study of the important points previously highlighted. Each phase of an LCA has major objectives and pitfalls to be avoided. The following paragraphs summarize the key points of each phase (Figure 7.1) and highlight critical issues on which to focus attention. Moreover, Appendix IV summarizes the forms to fill out for each LCA phase, reminding the reader of the key points in each phase, as well as the information that needs to be gathered and analyzed.

7.1.1 Goal and System Definition

Before performing or analyzing an LCA, it is important to consider the function of the product or system. A poor definition of the function of the system can lead to the comparison of alternatives that are not actually achieving the same function and thus jeopardize the relevance of the study. To test the validity of the function definition, one should ask the following questions:

1. Does the primary function differ between scenarios? If so, the comparison between scenarios does not make sense.
2. Do the secondary functions differ between scenarios? If so, this is important to account for in the interpretation.

Based on the product or system function, the comparative unit, or *functional unit* (FU), must be carefully defined and be the same for all scenarios. This is the variable that quantifies the system function of the system and to which all inventory flows are related. The *reference flows* are the amounts of products and energy that need to be directly purchased to achieve this FU; in other words, the first tier of required processes and materials. The ratio of each reference flow to the FU can be used to identify *key environmental parameters* that are critical to optimizing the system, such as the *lifetime* and *number of uses*.

Note that the FU is not a ratio. Moreover, the FU of a component of a complex system (e.g., car door) generally refers to the function of the entire system (e.g., passenger protection over 1 person-mile). It is important to avoid confusing the FU

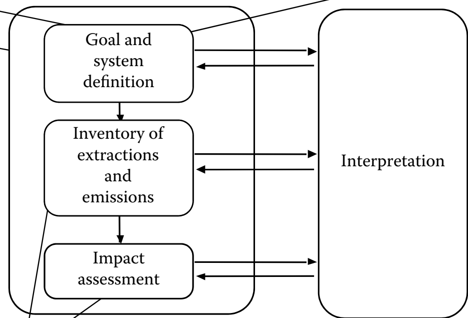
General Principle (Ch. 2)
It is recommended to carry out the LCA in two steps:
- A first screening step covers the whole LCA and assesses the order of magnitude of emissions, resource use, and impacts.
- A detailed analysis improves the assessment for the most important impacts.

1. Goal and system definition (Ch. 3)
1a. System function
The LCA assesses the environmental impacts associated with a product function. Products or systems can only be compared on the basis of a similar function.

1b. Functional unit and reference flows
- The *functional unit* is the common unit in all scenarios, quantifying the function of the system. Emissions and extractions are calculated per functional unit.
- The *reference flows* differ for each scenario, representing the amounts of goods or services purchased to generate one functional unit.
- Lifetime, number of uses, use phase efficiency, and other parameters linking reference flows to functional unit can all be *key parameters* for environmental optimization.

1c. System boundaries
System boundaries include all the necessary processes to assure the desired function.
- Rule 1: System boundaries must cover the same functional reality in all scenarios.
- Rule 2: A cut-off should be defined, where processes should be excluded if they contribute less than the cut-off percentage (e.g., 1%).
- Rule 3: Processes that are identical in the different scenarios can only be excluded if the reference and intermediary flows affected by these processes are strictly equal (i.e., total output of each system must also be identical).

2. Inventory (Ch. 4)
2a. Definition
The inventory is the quantified elementary flows of matter, energy, and substances that cross the system boundary, including pollutant emissions and extracted resources.
2b. Allocation
- Avoid allocation through *process separation* (if it is possible to dissociate the different product systems) or through *system expansion* (if it is possible to extend the system boundaries to include a proven substituted product and avoided impacts).
- Use *physical allocation* based on mass or energy content only if there is a direct causality, or use *marginal allocation* when it is possible to marginally vary the proportion between coproduct and product.
- Use *financial allocation* based on market value as a measure of the incentive for production.
- Use *arbitrary assumption* without causality only if necessary.
2c. Input–Output
The *input–output approach* deconstructs the functional unit into industrial spending to create the inventory.



4. Interpretation (Ch. 6)
Interpretation should be carried out at all levels, looking at:
- The contribution of each life cycle stage and process, to prioritize improvements that generate the largest impact at the lowest cost
- The contribution of each substance
- Each phase of the life cycle assessment
- Uncertainty to identify if scenarios are significantly different

3. Impact assessment (Ch. 5)
The *impact assessment* quantifies the impact of each emission and extraction to enable comparison of scenarios and the contributions of different life cycle stages and pollutants.
- Impact assessment methods must at least incorporate the fate (transport, dilution, and degradation) and effects (e.g., intrinsic toxicity) of emissions and extractions.
- More sophisticated methods accounting for spatial differentiation should be used when it substantially reduces variability.
- Several impact assessment methods should be used in parallel to test result robustness.

FIGURE 7.1 Summary of key concepts in life cycle assessment.

(the service) with the reference flow (what to buy). The following questions help avoid this problem:

3. Does the chosen FU represent the service offered (rather than an impact or consumption)?
4. How are the material quantities, densities, or lifetimes taken into account to determine the reference flows?
5. How are differences in the secondary function accounted for among scenarios?

Continuing to stay centered on the system function, the system boundaries must be defined based on function rather than geography. (Thus, a fast-food restaurant cannot simply be compared to a family restaurant without accounting for the food preparation of the fast food before it reaches the restaurant). Remember the three rules of system delimitation defined in Section 3.5: (1) Do not forget about any upstream processes needed to achieve the function; (2) use cut-off criteria and inclusion of processes in a methodical and systematic manner; and (3) do not exclude processes common to the different scenarios if their reference flows differ. Help yourself follow these three rules by asking yourself the following questions:

6. Do the system boundaries cover the same reality in all scenarios?
7. What is the cut-off threshold percentage to exclude minor processes, and based on which inventory or intermediary flow (mass, inventory flow, impact)?
8. Are certain production stages common to all scenarios? Do the quantities of the manufactured product or the reference or intermediary flows of common steps vary among scenarios? (If so, it is not possible to exclude the common steps.)
9. Do any of the production steps or components indirectly affect the quantity of product used or lost? (For example, the type of packaging can affect amount of product losses. If so, the product itself must also been considered when assessing its packaging options.)

7.1.2 Inventory

Based on the processes and references flows included in the system boundary, the next phase calculates the inventory of raw material extractions and substance emissions associated with each FU. This is often based on existing databases of processes, supplemented by first-hand product-specific data.

When the product studied is associated with other coproducts, it is necessary to *allocate* which emissions and extractions should be assigned to each of these coproducts. Allocation should be avoided whenever possible, either by separating out only the processes associated with the considered product or by expanding the system boundaries to count the avoided emissions due to substituting an existing product with the coproduct. A poor choice of the substituted product, however, can give totally false results; thus, it is important to consider the following questions:

10. Is the substituted product actually replaced by the coproduct? (If not, it is not legitimate to apply this system expansion.)
11. Can the coproduct be used to replace other substituted products? Which substituted product is the most unfavorable for the environment?

When allocation cannot be avoided, *physical*, *marginal*, or *financial allocations* are all methods of distributing the emissions among coproducts, listed in order of preference. Each method should only be applied if a causal relationship can be established for using that allocation method. For example, an allocation by mass (a type of physical allocation) has no meaning if there is no physical reason for the emissions to scale with mass. You thus want to ask yourself:

12. Is there a causal underlying physical relationship between the physical parameter (e.g., mass or volume) and emissions and amounts of raw materials used?

Use marginal allocation when it is possible to marginally vary the proportion between the coproduct and the product. The financial allocation applies in all other cases and can always be tested as part of a sensitivity analysis.

Whether or not allocation is used, it is important to characterize the quality of available data to ensure the quality of the results provided by the study. For this, the following quality indicators can be used: data reliability; data representativity; and temporal, geographic, and technological correlations. You want to ask yourself:

13. What is the quality of the data used in the inventory?
14. To what extent are the data measured in situ?
15. Do the unit processes correspond to the technology used?
16. Are some of the data old or from other geographic areas? How does this affect the results?

7.1.3 Impact Assessment

Using the inventory of raw material extractions and substance emissions, the *impact assessment* phase calculates the resulting impacts on humans and ecosystems. The chosen impact assessment method should include information on the fate and effect (e.g., transport, degradation, and toxicity). Because of the broad array of substances and locations, current methods may only accurately predict impacts to within an order of magnitude (which is still very useful in prioritizing actions, because impacts can vary among substances by 12 orders of magnitude). It is thus important to *apply several methods of impact assessment* to test the robustness of the conclusions. So, be critical of the results and consider:

17. Do the main conclusions change according to the impact assessment method used?
18. What impact categories may be critical for the considered problem and how well do the selected impact methods address them?
19. Are certain methods better suited to this study? And why?

7.1.4 Interpretation

Throughout the LCA phases, the results should be regularly interpreted, focusing on the dominant pollutants of each life cycle stage that generate the greatest impacts and on those stages that have the greatest potential for impact reduction. Similarly, the uncertainties of the results must be considered in the context of the analysis, along with the conditions of the study and any assumptions made. To avoid common pitfalls in the interpretation phase, the following questions should be asked:

20. Have results been interpreted at each phase of the LCA (raw inventory, impact at midpoint level, and impact at damage level)?
21. Which life cycle stage(s) and which pollutant(s) contribute most to the environmental impacts?
22. Are the differences among scenarios significant? What is the significance criteria based on? What is the level of certainty of the major conclusions?
23. What are the "key" parameters from an environmental perspective (based on sensitivity studies)?

7.2 LIMITATIONS OF AN LCA

There are many practical and conceptual limitations when conducting an LCA. First, depending on the level of detail required and the availability of data, an LCA can take time, especially during data collection. Moreover, especially if not applying the rules of good practice, the results can be influenced by many subjective factors, including the people conducting the LCA, and the choice of assumptions or impact assessment method. This possible subjectivity could cast doubt on the reliability and quality of the obtained information and thus call the results into question. It is therefore necessary to adopt a critical approach and to carefully check the assumptions and their consequences on the results of the study. Particular care should be paid to the interpretation and communication of the results. Finally, for certain substances in some impact categories, such as toxicity, the results only indicate orders of magnitude; these uncertainties should be accounted for during interpretation by, for example, looking at and presenting results on a logarithmic scale.

An LCA consists of key underlying assumptions that limit its applicability. First of all, the analysis generally only addresses adverse environmental effects, without addressing the social and economic effects, which must be evaluated using additional tools such as life cycle costing or social LCA. The spatial and temporal dimensions are generally not differentiated, and results are often aggregated spatially and temporally. The environment in which the system lies is assumed to be uniform, with only the studied system being adapted. Moreover, the impacts are considered to be additive, which means that the high scores of one category can be compensated by lower scores in other categories depending on the weight assigned to each category.

To summarize, an LCA may not be appropriate in the following cases:

- When only the social or economic dimension needs to be analyzed
- When impacts are highly spatially variable or associated with differences in local or regional sensitivities
- When major changes occur that deeply modify the studied system and involve nonlinear responses in the system
- When systems other than the studied system change as well (such as changes in the economic structure or consumer behavior)
- When aiming at checking whether toxicity limits have been exceeded

Once the LCA suggests certain advantageous improvements or adjustments, what are the real consequences of these choices? There are limitations in the ability to predict real-life consequences, because implementing an action designed to reduce one kind of impact can cause other "collateral" impacts that can be positive or negative. The LCA does not account for many potential consequences, such as the risks to human health and the natural environment due to changes in behaviors induced by the alternative; the risks to health caused by changes in available income; and the risks to health and environment due to structural changes or innovations.

The example of fluorescent light bulbs, as discussed in Section 3.3.3, illustrates this concept of unintended consequences. Fluorescent light bulbs pollute less and are cheaper than incandescent light bulbs when compared over the entire life cycle. What are the consequences of these financial savings to consumers? They may decide to install more bulbs in less well-lit areas or they may be less attentive in turning off the light when leaving a room. They can use the money saved to purchase other consumer goods. In such a case, the actual environmental improvement would be lower than that foreseen by the LCA. Two scenarios that do not generate the same costs can therefore lead to different indirect impacts, which are not accounted for in an LCA. It is thus important to identify these potential indirect effects and, if relevant, complement functional-based recommendations with qualitative consumption-based recommendations on overall effects when communicating LCA results (as discussed in Section 6.8.4).

7.3 POTENTIAL APPLICATIONS OF AN LCA

7.3.1 Overview of LCA Publications

LCAs have been applied for multiple purposes and to multiple products. Figure 7.2 depicts how LCAs have been applied as a word cloud, in which the size of each word scales with how frequently it appeared in the title of any publication containing the words “life cycle assessment” in its title.

Figure 7.2 indicates how and why LCAs are being applied and studied, at least for publication purposes. LCAs are often used to “compare” the “impacts,” “energy,” and “emissions” associated with “production” “systems,” “integrating” over the entire lifetime of the product.

LCA publications still have a strong methodological emphasis, frequently mentioning “methodology,” “sustainability,” “evaluation,” and “management.” Many address the “carbon” “footprint” and “greenhouse” “gas” (“GHG”) emissions as part of the most common mention of “impacts”.

The impacts in LCA are considered from a “production” or product-oriented perspective in contrast to, for example, risk assessment, which is receptor-oriented. The “design” stage of a product is thus also a prominent application type.

The most prominent application domain is clearly “energy,” with additional related terms such as “electricity,” “fuels,” “power,” and “solar.” A second application domain is associated with “buildings” and “materials,” including “construction.” Other important domains are “technology,” “waste” and “recycling,” and “water.” Agriculture and forestry applications are also represented by multiple words such as “food,” “plants,” “biomass,” “crops,” “wood,” “biodiesel,” “biofuel,” and “bioethanol.” “Transportation” and “vehicle” are present but not in a prominent way.

FIGURE 7.2 Most common words in the title of the 7920 papers in the Scopus database with “LCA” and “life cycle assessment” in their title (search conducted March 20, 2015). The title of each paper was pasted into Wordle, which then created this word cloud based on frequency of occurrence of each word. The words “LCA,” “life cycle assessment,” “environmental,” “case study,” and other commonly used English words were excluded; synonyms were grouped (e.g., “impacts”/“impact,” “compare”/“comparing”).

Finally, the most frequently mentioned global region is China (166 citations), followed by Europe (96) and Brazil (68), which may indicate an interesting shift from the traditional use of LCA in developed countries toward emerging countries.

We now discuss some of these application domains in more detail.

7.3.2 Application to Ecodesign

LCA has many potential applications due to the large amount of information obtained. These applications can be in the context of company management or in the public domain.

LCA provides valuable information for the evaluation and improvement of a product, identifying critical points where the environmental performance of a product can be improved. LCA can thus be used to help define certain principles of ecodesign (design of environmentally friendly products). The first principle is that it is necessary to dematerialize, and work in terms of services rather than products. For example, a store where you can print and make copies does not sell photocopiers, but only rents use of the copier to make the copies. To be successful, the designer must therefore build devices that are more reliable.

The second ecodesign principle stemming from life cycle thinking is the importance of reducing manufacturing impacts by minimizing the amount of material used for a given product function, removing toxic substances used in manufacturing, and minimizing production waste and the diversity of materials used to facilitate recycling.

The third principle focuses on the use stage, aiming to reduce energy consumption and the product weight for all processes where transport is important, and to minimize the waste during the use stage.

Finally, the last set of ecodesign principles addresses the end of life, including the importance of using recycled materials and an easily dismantled product to make the product recyclable, especially when the production of toxic materials and waste is important. We must also try to extend the life of the product, especially if the impact of the production stage is greater than that of the use stage. However, if the studied product is continually developing to reduce impacts over an otherwise dominant use stage, a longer lifetime is not recommended. For example, driving an old car without a catalytic converter produces more emissions than producing and driving a new car.

One problem in using LCA for ecodesign is that it may come too late in the design process of a product, thus limiting the flexibility. We recommend applying the tool as soon as possible in the development of a new product, even if only partially, or utilizing results obtained and lessons learned from a previous similar product.

7.3.3 Application to Product Comparisons

LCA enables the comparison of existing products and services within a company or between competitors, providing information that can be used to select a particular product or service. A comparative study can also provide ideas for improving production and design processes of products. This is currently one of the most common uses of LCA.

7.3.4 Application to Long-Term Decision-Making

LCA also applies in the context of strategic planning, and more globally at the environmental management and decision-making levels, whether in a company or in a government or nongovernmental agency. Companies define and utilize long-term environmental strategies for various reasons. First, a company may realize that environmental improvements following "end-of-pipe" approaches are significantly more expensive, since, for example, it is often more costly to treat waste and solve toxicity problems at the end of a pipe rather than simply avoid or minimize this waste in the production and use of the product. It is then advantageous to make radical changes in the life cycle of products, and LCA effectively helps to identify bottlenecks in production, possible alternative production processes, and best possible design opportunities.

Secondly, an organization that anticipates future legislation or changes in public perception can use LCA results to plan better and more flexible production strategies and technologies. This reduces the long-term risk of losing customers or business based on environmentally sensitive consumers or legislation. LCA results allow a company or administration to account for the environmental performance of their processes and products when choosing its partners (suppliers, subcontractors, etc.). Indeed, suppliers used to generally be selected based only on the cost and performance of their product. Now, customers are increasingly demanding that companies comply with quality standards and environmental criteria, which implies taking into account environmental aspects when choosing suppliers. For governments, LCA can be used to define priorities regarding environment and energy development, and to generally help guide environmental policies.

Finally, LCA can contribute to the development of standards. In fact, LCA is the mandatory basis for the environmental declaration of products and the definition of ecolabel criteria. LCA also has potential in legislation, public information, advertising, and marketing.

7.3.5 Application to Different Types of Products

LCAs are applied to a variety of systems and products (Figure 7.2, Table 7.1), with some of the main domains in packaging, motor vehicles/transport, construction, synthetic materials, and energy (Frankl and Rubik 2000). Note that the emphasis on packaging and transport found by Frankl and Rubik in 2000 does not show up in the 2015 literature survey (Figure 7.2), showing a shift toward energy, building, and waste (see Section 7.3.1).

7.3.6 LCA: A Tool for Life Cycle Thinking

Life cycle assessment is part of the larger framework of life cycle thinking, which generalizes LCA results with the goal of linking technology and sustainable development. It includes other concepts, such as industrial ecology (where wastes from one process are used as raw materials for another), risk analysis, ecolabels (environmental labels), environmental management systems, ecodesign (creation of a more

TABLE 7.1
Life Cycle Assessment by Type of Product

	Absolute Number	% of Total
Motor vehicles/transport	95	15
Synthetic materials	61	10
Packaging	103	17
Hygiene/cleaning	28	5
Construction/building materials	78	13
Food	26	4
Energy	52	8
Paper/printing	21	3
Electronic equipment	31	5
Waste	33	5
Metals	11	2
Furniture	8	1
Textiles	6	1
Other	39	6
Unknown	29	5
Total	621	100

Source: Frankl, P. and Rubik, F., *Life Cycle Assessment in Industry and Business: Adoption Patterns, Applications and Implications*, Springer, Germany, 2000. With permission.

Note: Study based on all LCAs performed until 1996 in Switzerland, Italy, and Sweden.

environmentally friendly product), and life cycle management. Currently, life cycle thinking is being developed within the methodological framework of the Life Cycle Initiative (Section 1.3).

7.3.7 Example of Life Cycle Management of a Company

Life cycle management, one aspect of life cycle thinking, aims to integrate environmental aspects into industrial processes by considering the impacts and costs of the supply chain. It seeks to increase the competitiveness of new and existing products, by examining the strengths and weaknesses of a product within its market, combined with the environmental and social aspects throughout its life cycle. Thus, life cycle management can be seen as a way to improve the environmental performance of a company within its temporal and financial constraints.

An example of a life cycle management tool is Quantis Suite 2.0 (see website listed in Appendix I), which applies the principles of a product LCA to the whole company, taking into account the supply and user chain. This tool is developed by Quantis, an environmental consulting spin-off from the Swiss Federal Institute of Technology, Lausanne (EPFL).

The Quantis software quantifies the carbon footprint and environmental performance of a company, accounting for both direct and indirect impacts as required by the international ISO 14000 standard. The impacts of the production site are evaluated and compared with those of the supply chain, use stage, and disposal. The software compares this environmental analysis to a cost analysis to identify improvement opportunities that offer the highest benefits at the lowest costs. It includes an analysis of legal compliance to quickly position the company within its legal environment without getting lost in increasingly complex legislation.

The user enters basic company data related to its products and activities. Data can be entered geographically by production site, production unit, or type of product. Results are presented as tables summarizing the primary energy consumption, equivalent CO_2 emissions, and emissions of important pollutants at a variety of levels for the product or factory, including the supply chain, company or production site, and waste disposal chain (Figure 7.3). Inventory results, which can also be provided in detail for each step, are then used in a second module to calculate impact. Finally, the costs are considered and combined with the impacts calculated in the first two

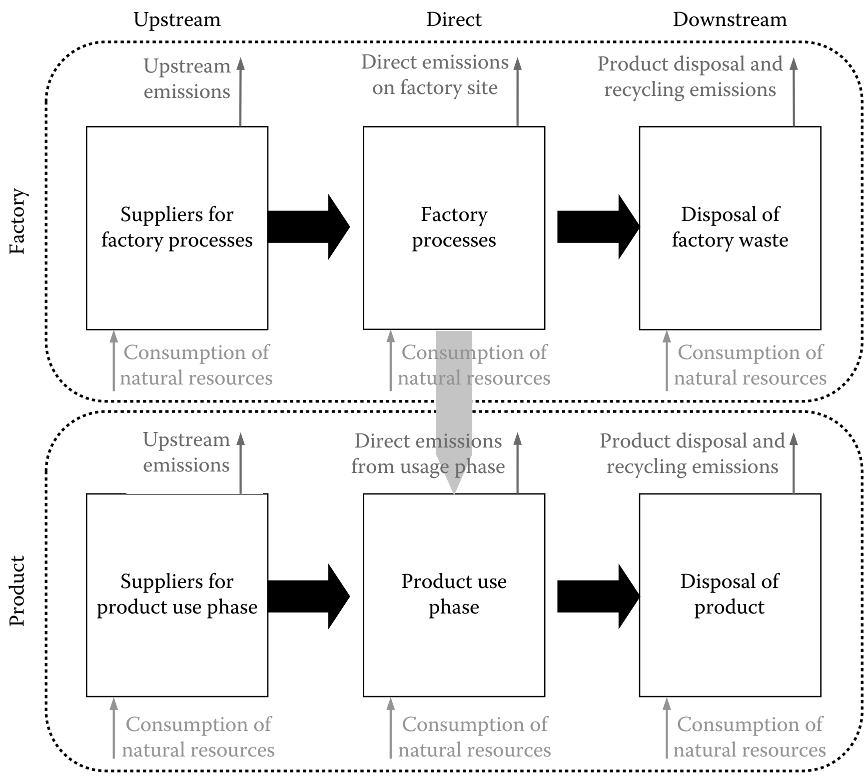


FIGURE 7.3 Life cycle stages considered in Quantis software for life cycle management. (From Della Croce, F. et al., Company-LCA: an innovative analytical tool for the quantification of companies environmental performances, internal report EPFL-GECOS, 2005. With permission.)

modules to calculate the economic benefit derived from a marginal reduction of impact for each step of the life cycle and each process of the company. This helps to identify action priorities that account for both financial and environmental aspects. One of the advantages of this approach is to offer to a broad array of companies the opportunity to evaluate their performance over the course of the life cycle and account for their specific structures without having to hire internal LCA specialists.

EXERCISES

Exercise 7.1: Critical Analysis of an LCA Case Study

Select an LCA case study paper and analyze it from a critical point of view, using Figure 7.1 and the key questions boxed in this chapter to look for biased, missing, and well-studied elements in the goal definition, inventory, impact assessment, and interpretation.

Search by keyword "LCA case study" in a literature search engine or use the article by Humbert et al. (2009), presenting the comparison of two packaging materials.

8 LCA through Example from A to Z

Treating Urban Sewage Sludge

Gregory Houillon, Olivier Jolliet, Shanna Shaked, and Myriam Saadé-Sbeih

This chapter provides a concrete and complete example of a life cycle assessment (LCA) application. It compares alternative treatments of sewage sludge from municipal wastewater, as addressed in a detailed study (Houillon and Jolliet 2005) carried out in collaboration with a group of French and Swiss industries. Following a short introduction, this example illustrates the four phases of LCA: goal and scope definition, inventory of resource extractions and emissions, impact assessment, and interpretation. For each of these phases, the calculations will be detailed for each considered scenario. The analysis of environmental impact is calculated using two methods: IMPACT 2002+ (Jolliet et al. 2003) and Eco-indicator 99 (Goedkoop et al. 1998).

This study is based on the Ecosludge project carried out at the Swiss Federal Institute of Technology Lausanne (EPFL), in partnership with Bonnard & Gardel (BG) Consulting Engineers, Ondéo Degrémont, Omnium de Traitement et Valorisation (OTV), Stéreau, and the Interdepartmental Syndicate for the Sanitation of Greater Paris (SIAAP). The authors gratefully acknowledge the inputs of the steering committee from this Ecosludge project.

8.1 INTRODUCTION

8.1.1 Overview of Case Study Application: Urban Wastewater and Sewage Sludge Treatment

Before describing the study objectives and the system studied, we first provide a basic understanding of sewage sludge treatment.

8.1.1.1 Urban Wastewater Treatment

Due to domestic usage and runoff, wastewater becomes loaded with organic and inorganic substances, either dissolved or suspended. Key substances found in wastewater include organic carbon, nitrogen, and phosphorus compounds, as well as traces of heavy metals and organic compounds. Wastewater is then transported to the wastewater treatment plant, where it is treated with physicochemical and biological processes. The most basic plants eliminate organic pollution (carbon, nitrogen, and phosphorus compounds), while the more sophisticated ones also capture micropollutants. The plant capacity is defined in terms of *equivalent habitants* (eq-hab), where 1 eq-hab represents the daily wastewater produced by a person.

8.1.1.2 Urban Sewage Sludge Treatment

The process of treating wastewater results in *sewage sludge* that can be one of two types.

1. Primary sludge is obtained by simply decanting wastewater. The solids that are thus separated from the water are generally rich in minerals (such as microsands and dirt) and also contain volatile organic materials.
2. Biological or secondary sludge results from biologically treating wastewater and is made up of bacterial bodies and their secretions.

Since wastewater treatment results in a substantial amount of sewage sludge, the sludge must undergo various treatment steps. Such treatment first reduces the sludge volume (e.g., by gravity thickening, in which the sediments are deposited), and at times a secondary volume reduction is needed (such as dehydration or centrifugation). Depending on the sector, the sludge may then undergo stabilization (such as through direct drying, liming, or anaerobic digestion), followed by case-specific treatments. Section 8.2.3 describes the different sludge treatments accounted for in this study.

Urban sewage sludges differ from *industrial sewage sludges*, which come from the treatment of *industrial wastewater* and have different characteristics. This study focuses on the treatment of urban sewage sludge for a wastewater treatment plant of an equivalent capacity of 300,000 inhabitants. These results may thus have to be adapted for different sizes or types of wastewater treatment plants.

8.1.2 Review of Environmental Assessment of Wastewater Sludge: Treatments and Key Challenges

Treatment of urban wastewater sludge is an environmentally sensitive problem due to the energy, costs, and pollutants involved. Laws concerning agricultural spreading of sludge (Spinosa 2001) and thermal oxidation processes are becoming increasingly restrictive. Controversy over the use of sludge is ongoing, and scientific arguments, as well as stakeholder values, need to be considered (Bengtsson and Tillman 2004). New treatment processes, such as pyrolysis and wet oxidation, have been introduced on to the market, but it is not yet clear which alternatives effectively reduce overall environmental impacts. The present study compares the life cycle environmental impacts of 12 wastewater sludge treatments, focusing on alternative processes and identifying key parameters to reduce environmental impacts.

A number of studies have been published comparing various sludge treatment processes, all of which add to the state of knowledge while leaving unanswered the question of which treatment processes reduce overall environmental impacts. Bridle and Skrypski-Mantele (2000) accounted for some additional processes (gasification) and demonstrated the benefit of phosphorus recovery, but only considered input–output methods, without including metal transfers. Hwang and Hanaki (2000) presented energy and CO_2 balances of sludge incineration processes using input–output methods, but did not consider other substance emissions. AERM (1999) included social aspects in their analysis of sludge treatment, but excluded alternative processes such as wet oxidation and pyrolysis and did not account for micropollutants, dioxins, and related human toxicity. Moreover, energy and coproduct recovery are not clearly described, though they can be very important for a comparative LCA (Lundin et al. 2000). Müller et al. (1999) presented a site analysis in Switzerland that cannot be used to analyze treatment processes on a general basis. Benz et al. (1995) carried out an LCA on sludge treatment processes, but using an outdated LCA methodology. Neumayr (1999) only considered digested sludges. Suh (1999) and Suh and Rousseaux (2002) performed a detailed analysis demonstrating the advantages of anaerobic digestion and the importance of heavy metals, but only compared agricultural spreading with fluidized bed incineration and landfilling. Huybrechts and Dijkmans (2001) accounted for many processes, but did not include the full life cycle approach. Lundin et al. (2004) discussed the potential advantages and drawbacks of recycling phosphorus in sludges compared with incineration scenarios, but performed no comparison with other thermal processes. Houillon and Jolliet (2005) presented preliminary results of this chapter's case study, but focused only on energy and greenhouse gas impacts of digested and undigested sludges.

Several scientific challenges must therefore be addressed to provide a clearer understanding of environmental impact pathways and key parameters:

1. How well do alternative technologies for sludge treatment, such as pyrolysis or wet oxidation, perform from an environmental point of view? Are they significantly better than high performance incineration?
2. What is the importance of avoiding energy use and emissions by using coproducts? How should the substitutions of fertilizers, natural gas, coal, or methanol be consistently compared in different treatments?
3. What are the environmental advantages of digesting sludges before landspreading or thermal treatment?
4. How important are the human health and ecosystem impacts of indirect emissions due to transport and to the production of auxiliary inputs relative to the impacts of direct emissions of respiratory inorganics and heavy metals during treatment?

This chapter addresses these questions by quantifying environmental impacts of alternative processes applied to wastewater sludge treatment (wet oxidation, pyrolysis, incineration in cement kilns) compared with processes typically used in Europe (agricultural spreading, fluidized bed incineration, landfilling). This work improves on previous studies in the following ways: Most data are provided from existing industrial plants; and substitutions, sensitivity analysis, and micropollutant transfers have been taken into account, which constitutes an important step forward for sound decision-making in this very sensitive domain. Going through each phase of an LCA, we first define the functional unit, system boundaries, and considered scenarios. Results of the inventory analysis and of the impact assessment are then presented, comparing the 12 scenarios. In the interpretation and discussion section, each reference scenario is discussed in detail, and a sensitivity analysis finally identifies key parameters leading to policy recommendations.

8.2 GOAL AND SCOPE DEFINITION

8.2.1 Objectives

The goal of this study is to evaluate several systems to treat wastewater sludge from a treatment plant for 300,000 equivalent inhabitants and to determine the key parameters influencing the environmental performance. This study quantifies the environmental impacts of six treatment processes applied both to undigested and digested mixed sludge, where each process is described in more detail in Section 8.2.3:

1. Agricultural landspreading of limed pasty sludge (AGRI)
2. Incineration in fluidized beds of pasty sludge (INCI)
3. Wet oxidation of liquid sludge (WETOX)
4. Pyrolysis of dried sludge (PYRO)
5. Incineration in cement kilns of dried sludge (CEME)
6. Landfilling of limed pasty sludge (LANDF)

Processes applied to digested sludge are noted by adding a “d” to each acronym.

8.2.2 Functional Unit

The function considered in all scenarios is the treatment of urban wastewater sludge at the output of the wastewater treatment plant (WWTP) before sludge thickening has occurred. The chosen functional unit used as the basis for comparison is the sludge resulting from 1 t of disposed dry matter (tDM), so all emissions, materials, and energy consumption are expressed relative to this functional unit (ISO 2006). The considered sludge is composed of 60% primary sludge and 40% biological sludge. The WWTP water is treated for organics, nitrogen, and phosphate compounds.

8.2.3 System Definition

8.2.3.1 Description of Studied Scenarios

This study considers 12 scenarios of sewage sludge treatment, where the first 6 treatments are without digestion and defined as follows.

1. Agricultural landspreading of limed pasty sludge (AGRI): Sludges are thickened, dehydrated, and stabilized (limed) in the WWTP, thus becoming limed pasty sludge (30% dry solid content). They are then stored an average of eight months in a controlled and deodorized storage area, since spreading over agricultural land can only take place at given periods in the year. The sludge is transported by 40 t trucks to the field, where it is then stored for an average of one month at the edge of the agricultural land, before being distributed over the land using a tractor. This spreading takes advantage of the fertilizing elements in the sludge, but micropollutants in the sludge are also spread on to the agricultural land.
2. Incineration in fluidized bed of pasty sludge (INCI): Incineration is a thermal process that leads to the destruction of the sludge organic matter. The process considered here is incineration at the WWTP, in a fluidized bed furnace at 850°C. The pasty sludge (25% dry solid content after dehydration) is burned and releases combustion gases that experience dry gas treatment before their emission into the atmosphere. The fly ash and other residues are then sent to a final waste storage center, where they may be stabilized in cement. The incineration process destroys pathogens and organic matter, so that transportation is limited to residues.
3. Wet oxidation of liquid sludge (WETOX): This WWTP process consists of using oxygen for aqueous oxidation of thickened sludge (6.8% dry solid content), at a temperature of 235°C and a pressure of 40 bar (Luck 1999). Operating at a higher pressure than fluidized bed incineration, this process generates a mineral residue that is then transported to a waste storage center. Wet oxidation breaks down the organic matter into carbon dioxide and water vapor that are emitted to air, along with other organic materials (aqueous effluent) that are then reprocessed by the WWTP. The carbon load in the effluent can be used in place of methanol for the WWTP, but this is not a standard substitution and has thus only been considered in a sensitivity analysis.
4. Pyrolysis of dried sludge (PYRO): Sludge is thickened, dehydrated to 95% dry solid content and thermolyzed at 500°C. Pyrolysis (or thermolysis) is thermochemical decomposition of organic matter without oxygen. This reaction produces a thermolysis gas, which can be reused for other applications (such as drying and heating), and a carbonaceous residue that is transported to a waste storage center. This process reduces flue gas treatment. The carbonaceous residue can be burned on or off the WWTP site. The thermolysis gas produced can be transported over longer distances than the heat recovered from the incineration process described in (2), thus decoupling production from recovery in both space and time.
5. Incineration of dried sludge in cement kilns (CEME): In this process, sludge is dried at the WWTP to 95% dry solid content and transported to the cement factory. Then, sludge is burned in an oven at 1400°C, with other fuels, to destroy organic matter and produce clinker (a type of cement). After use of the cement produced with sludge ash, it is deposited into an inert waste storage center. The mineral component of the sludge thus becomes valuable as building material. Some micropollutants are emitted into the air during sludge combustion.

6. Landfilling of limed pasty sludge (LANDF): After liming (described in 1.), sludge is stored in a deodorized building for 15 days at the WWTP. Then, the limed pasty sludge (30% dry solid content) is transported to an organic-waste storage center. Landfills generally only require that sludges must be at least 30% dry solid content. But, French and Swiss legislation trends are restricting the amounts of such landfills (those with organic matter, since this is not considered a final residue). 60% of the landfill methane is collected and flared off to reduce methane emissions.

The incineration (INCI), wet oxidation (WETOX), pyrolysis (PYRO), and cement kiln (CEME) scenarios all use thermal oxidation processes, the impacts of which are discussed in Section 8.4.

The next six scenarios use the same treatments as the first six, with the addition of anaerobic digestion prior to the treatment process. Through this process, the organic materials present in the sewage sludge are degraded by bacteria in the absence of oxygen, thus producing biogas. The scenarios using digested sewage sludge are represented by a lowercase letter d:

7. AGRI d: Spreading of digested, limed paste-like sewage sludge.
8. INCI d: Fluidized bed incineration of digested paste-like sewage sludge.
9. WETOX d: Oxidization of digested, thickened sewage sludge.
10. PYRO d: Pyrolysis of digested, dried sewage sludge.
11. CEME d: Incineration in cement kiln of digested, dried sewage sludge.
12. LANDF d: Landfilling of digested, limed paste-like sewage sludge.

Details of each scenario, the sludge composition, and the main characteristics of the WWTP are given in Houillon and Jolliet (2005).

8.2.3.2 System Boundaries and Flow Chart

Along with the extraction, transport, and use of materials and energy, the system boundaries include: site infrastructure, sludge thickening, main sludge treatment processes, matter losses, sludge and residue transport, and solid waste treatment. The sludge treatment systems consist of different substeps for each type of treatment, ordered by their position in the treatment chain (Figure 8.1), which enables identification of common modules among the different scenarios. These modules can be structured into seven treatment steps common to most scenarios, which are indicated by the left integer in each module number of Figure 8.1, corresponding to each row. The right integers indicate different ways of performing this treatment step. The input quantities and specific aspects of each module differ for each scenario:

1. Primary volume reduction (thickening)
2. Secondary volume reduction (dehydration, centrifugation, etc.)
3. Stabilization (direct drying, liming, anaerobic digestion)
4. Storage (dried sludge, paste-like sludge, etc.)
5. Sludge and residue transportation (40 t truck)
6. Main sludge treatment (spreading, incineration, etc.)
7. Treatment of by-products (e.g., storage)

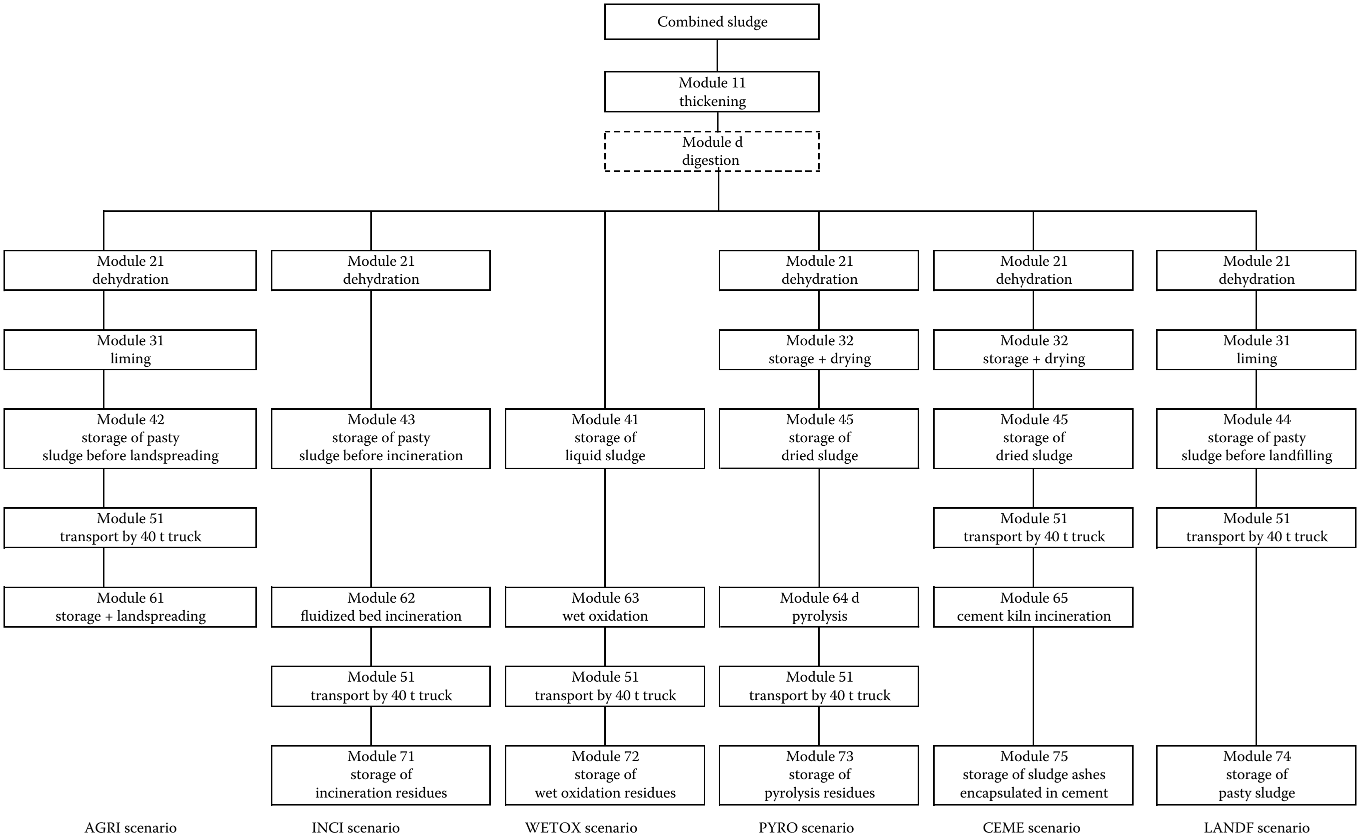


FIGURE 8.1 Sludge treatment scenarios expressed as series of common types of modules. In the digestion scenarios, digestion occurs after thickening.

The treatment processes of the digested sludge, represented by *d*, take into account the digestion module, indicated in a dashed box.

8.2.4 System Modeling: Reference Flows, Direct Emissions, Substitutions, and Data Quality

Table 8.1 summarizes the main reference flows for each scenario, showing that electricity and fuel are used in all disposal routes. Drying before pyrolysis and

TABLE 8.1
Main Reference and Intermediary Flows for the Treatment of the Undigested and Digested (in Italics) Sludges

	Unit	AGRI	INCI	WETOX	PYRO	CEME	LANDF
		AGRI d	INCI d	WETOX d	PYRO d	CEME d	LANDF d
Acid (hypochloric, nitrous, sulfuric)	kg/tDM	1.8	—	—	5.4	5.4	1.2
	kg/tDMd	1.8	—	—	5.4	5.4	1.2
Active coal	kg/tDM	—	2	—	—	—	—
	kg/tDMd	—	2	—	—	—	—
Copper sulfate	kg/tDM	—	—	14	—	—	—
	kg/tDMd	—	—	14	—	—	—
Electricity	kWh/tDM	233.7	400.4	796.8	944.3	336.8	159.8
	kWh/tDMd	236.6	284	490.9	488.2	305.9	186.4
Fuel	kg/tDM	11.1	14.4	1.8	37.4	38	14.3
	kg/tDMd	6.5	10.3	1.8	37.4	37.7	8.8
Lime	kg/tDM	400	30	—	30	—	400
	kg/tDMd	185.6	20.3	—	20.3	—	185.6
Natural gas	Nm3/tDM	—	65	—	314.5	314.5	—
	Nm3/tDMd	—	131.6	—	212.3	212.3	—
Nitrogen	kg/tDM	—	—	—	0.4	—	—
	kg/tDMd	—	—	—	0.2	—	—
Polymer	kg/tDM	7.1	7.1	0.1	7.1	7.1	7.1
	kg/tDMd	4.9	4.9	0.1	4.9	4.9	4.9
Oxygen	Nm3/tDM	—	—	810	—	—	—
	Nm3/tDMd	—	—	270	—	—	—
Weld	kg/tDM	—	—	35.5	6	—	—
	kg/tDMd	—	—	23.9	1	—	—

Note: tDM(d) represents tons of dry matter (digested) and Nm^3 represents normal cubic meters. Dashes indicate that data is not relevant to that scenario.

incineration in cement kilns requires substantial amounts of natural gas (315 Nm^3/tDM, where Nm^3 is a normal cubic meter) and electricity (166 kWh/tDM). Wet oxidation demands most electricity for oxygen production and the actual wet oxidation process. For agricultural spreading and landfilling, lime is needed for stabilization (400 kgCaO/tDM).

Resource extractions and pollutant emissions associated with the supply chains of each reference flow are then calculated by multiplying each flow by emission factors from standard LCA databases, as described in Section 8.2.4.3.

8.2.4.1 Direct Emissions and Micropollutant Transfers

In addition to the indirect emissions associated with intermediary flows, there are also direct emissions of substances contained in the sludge. For carbon-based emissions, CO_2 is emitted during combustion, and methane is emitted by landfilling. Human and ecosystem toxicity is impacted by emissions due to dry matter losses, heavy metal transfers, and wastewater treatment generated by sludge treatment processes. Table 8.2 characterizes the amount of micropollutants in the sludge, which are then multiplied by transfer factors based on measurements and literature data (Carpi and Lindberg 1997). These transfer factors describe, for each scenario, the fraction of each micropollutant transferred from the sludge to air, water, and soil, summed over all treatment steps (Figure 8.2). We consider here 6 organic and 16 metallic micropollutants.

8.2.4.2 Substitutions

To account for the coproducts of the various sludge treatments, we first identify for each scenario the type and amount of product substituted in the technosphere (Table 8.3). Emissions and resource use associated with these substitutions are avoided and thus subtracted from the inventory of each scenario, following the system expansion approach described in Section 4.5.3. Sludge treatment can lead to scenario-specific substitutions of both matter and energy.

For matter substitution, agricultural spreading (AGRI) decreases the need for N-, P-, and K-fertilizers due to the presence of nitrogen, phosphorus, potassium, and

TABLE 8.2
Micropollutants Contained in Sludge (kg/tDM)

Metallic	kg/tDM	Metallic	kg/tDM	Organic	kg/tDM
Ag	0.01	Mn	0.3	Benzo[*b*]fluoranthene	0.0006
Al	0.01	Mo	0.005	Fluoranthene	0.0015
As	0.005	Ni	0.05	Benzo[*a*]pyrene	0.0006
Cd	0.01	Pb	0.3	Dioxins	0.00005
Co	0.005	Sb	0.01	Furans	0.000005
Cr total	0.1	Se	0.003	PCBs	0.0005
Cu	0.5	Sn	0.01		
Hg	0.005	Zn	1.4		

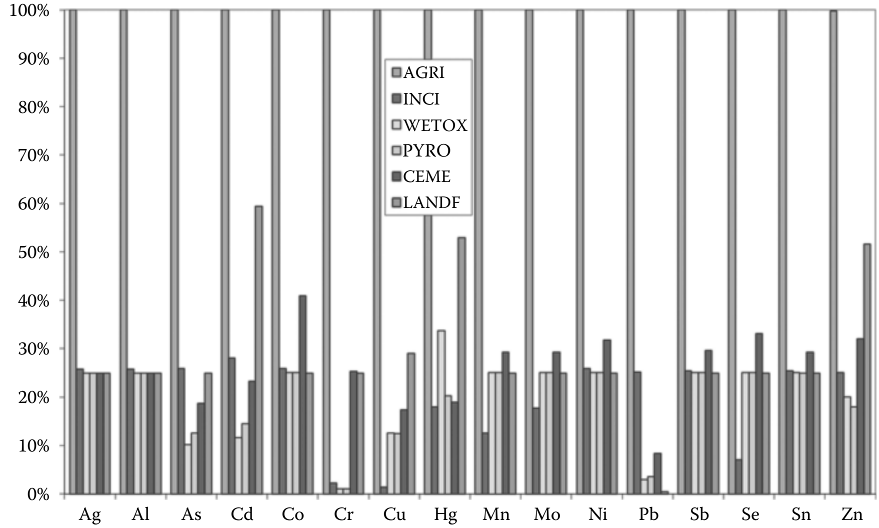


FIGURE 8.2 Fraction of each type of sludge micropollutant that is transferred to the environment (air, soil, or water) for the different scenarios. All values are normalized relative to the agriculture scenario.

limestone in sludge. Wet oxidation (WETOX), on the other hand, produces an easily degradable effluent, which can substitute methanol in the process of denitrification in the WWTP. This methanol substitution is addressed in the sensitivity analysis of the energy use in each scenario (Section 8.4.1). Finally, sludge incineration in cement kilns (CEME) substitutes the use of limestone by providing mineral matter for clinker production (Obrist and Lang 1986).

Fluidized bed sludge incineration (INCI) leads to energy substitution by recovering heat from the flue gas, and therefore substituting natural gas used for heating. For wet oxidation (WETOX), heat is recovered from the processed wastewater, also reducing natural gas needs. For pyrolysis (PYRO) and cement kilns (CEME), the pyrolysis gas and heat recovered from the direct sludge drying system both replace natural gas use. The cement kiln sludge treatment also replaces fuel and coal by producing heat for the cement production process.

Finally, adding the process of sludge anaerobic digestion prior to any treatment allows for the substitution of natural gas and reduction in organic matter. Digested sludges, however, reduce substitution capabilities of subsequent thermal oxidation processes due to the lower calorific value.

8.2.4.3 Data Sources and Quality

Thanks to the direct involvement of industries in the study, most reference and intermediary flows and direct emission factors come from measurements on industrial plants or plant design models (Houillon and Jolliet 2001), reducing uncertainties by making them as close as possible to real conditions. Additional emission factors

TABLE 8.3
Amounts of Materials and Energy Carriers Substituted for the Scenarios without/with Digestion

Treatment Step	Substitutions	Unit	AGRI/ AGRI d	INCI/ INCI d	WETOX/ WETOX d	PYRO/ PYRO d	CEME/ CEME d	LANDF/ LANDF d
Anaerobic digestion	Natural gas	Nm^3/tDM	0/152	0/152	0/152	0/152	0/152	0/152
Stabilization	Natural gas	Nm^3/tDM	—	—	—	80/54	80/54	—
Main treatment	Natural gas	Nm^3/tDM	—	180.5/123	192/96	450/175	—	—
	Ammonium nitrate	kg N	20.8/17.8	—	—	—	—	—
	Triple superphosphate	kg P	28.7/33	—	—	—	—	—
	Potassium chloride	kg K	2.4/2.4	—	—	—	—	—
	Lime	kg CaO	200/137.5	—	—	—	—	—
	Fuel	kg	—	—	—	—	104/63	—
	Coal	kg	—	—	—	—	170/103	—
	Marl and limestone	kg	—	—	—	—	350/350	—

TABLE 8.4
Nonrenewable Primary Energy Use and CO_2 and CH_4 Emissions per Unit of the Main Inputs in Sludge Treatment, Based on Ecoinvent 1.0 Data

Products	Nonrenewable Primary Energy	Unit	CO_2	CH_4	Unit
Active coal	70.5	MJ/kg	5.2	1.2×10^{-2}	kg/kg
Ammonium nitrate	49.4	MJ/kg	1	9.2×10^{-3}	kg/kgN
Chemical organic	41.8	MJ/kg	1.8	3.0×10^{-3}	kg/kg
Coal	33.85	MJ/kg	122.7	0.4	kg/GJ
Copper sulfate crystallized (36% wet)	89	MJ/kg	0.8	1.8×10^{-3}	kg/kg
Electricity	13.6	MJ/kWh	150.4	0.3	kg/GJ
Fuel	46.7	MJ/kg	93.5	0.1	kg/GJ
Hydrochloric acid	22.5	MJ/kg	0.8	1.6×10^{-3}	kg/kg
Lime	2.8	MJ/kgCaO	1.4	1.3×10^{-3}	kg/kgCaO
Limestone	0.11	MJ/kg	6.0E−3	1.0×10^{-5}	kg/kg
Marl	0.11	MJ/kg	6.1E−3	1.0×10^{-5}	kg/kg
Methanol	35	MJ/kg	0.9	Not available	kg/kg
Natural gas	39.25	MJ/Nm^3	64.4	0.2	kg/GJ
Nitric acid	11.8	MJ/kg	0.56	1.6×10^{-3}	kg/kg
Nitrogen	6.8	MJ/kg	0.3	5.2×10^{-4}	kg/kg
Oxygen	10.5	MJ/kg	0.3	5.2×10^{-4}	kg/kg
Polymer	41.8	MJ/kg	1.9	Not available	kg/kg
Potassium	11.8	MJ/kgK$_2$O	0.63	1.7×10^{-3}	kg/kgK$_2$O
Sulfuric acid	5.22	MJ/kg	0.2	5.1×10^{-4}	kg/kg
Superphosphate triples	46	MJ/kgP	2.46	3.8×10^{-3}	kg/kgP
Transport by 40 t trucks	2.6	MJ/tkm	0.1	2.6×10^{-4}	kg/tkm
Weld	22	MJ/kg	0.8	1.6×10^{-3}	kg/kg

Note: For future studies, we recommend instead using the latest ecoinvent data for these factors.

come from OFEFP (1998) and Suter et al. (1996a,b). Table 8.4 presents a sample of emission factors for CO_2 and CH_4 emissions associated with each input in sludge treatment.

Sensitivity studies comparing these original data with those from the ecoinvent database (Frischknecht et al. 2005) do not show major differences for the considered processes and did not justify a full update in the final study stage. For a selection of the same flows, ecoinvent 2.2 data are available in Appendix 3.

To check that emissions are reasonable, we calculate and check the mass balances for carbon, nitrogen, and phosphorus. Figure 8.3 shows the carbon mass flows in the agricultural landspreading scenario, with the carbon that starts in the sludge eventually all being accounted for. A ton of dry sludge matter contains 338.7 kg carbon, which leaves the system as suspended matter in wastewater and as gases (CH_4, CO,

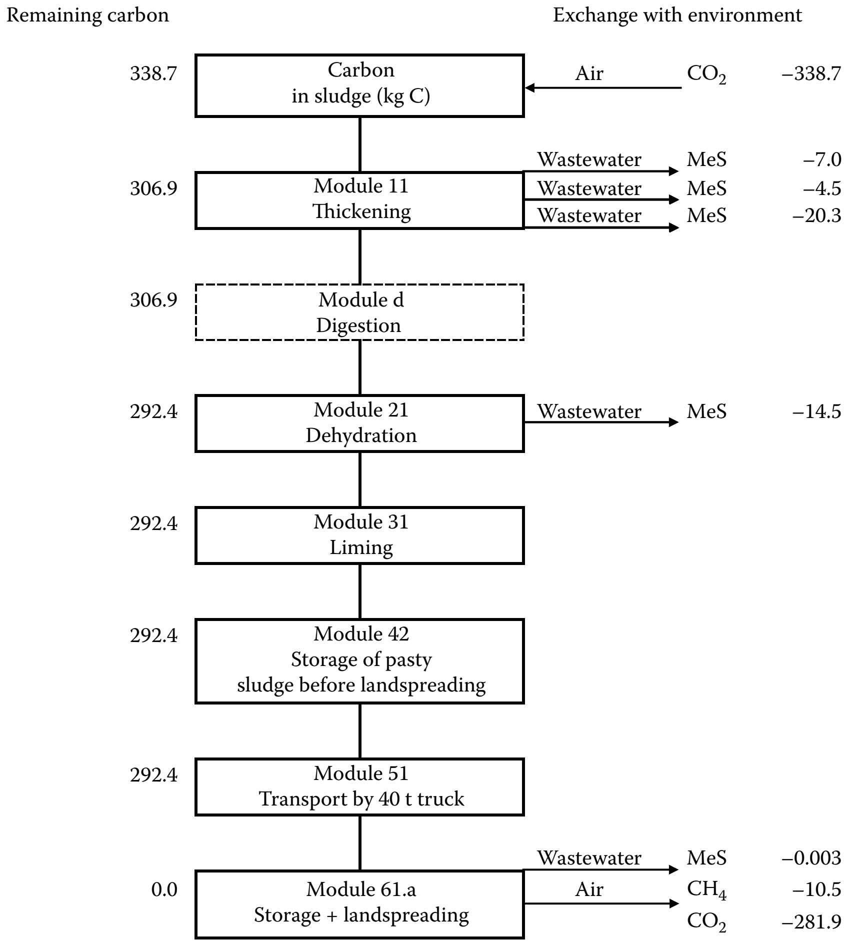


FIGURE 8.3 Carbon balance of sludge for agricultural landspreading of limed pasty sludge scenario (expressed in kg carbon per ton dry matter).

CO_2, and volatile organic compounds) during sludge digestion and combustion. The amount of carbon remaining in the system after the last process is zero and thus closes the balance. Carbon dioxide emissions linked to fossil fuel consumption are added in a second step.

Uncertainties arise for various reasons, including data quality, as well as approximations in the inventory data, transfer coefficients, and substituted flows. Moreover, since the net inventory flows are the differences between the large flows associated with treatment emission/consumption and the large substituted flows, the relative uncertainties over these net flows are high. This LCA methodology is also limited in the types of impacts it considers; pathogenic and sanitary risks are not taken into account, and neither are noise, odors, or visual impacts.

8.3 INVENTORY RESULTS

8.3.1 Intermediary Flows and Detailed Calculation of the Energy Consumption of the INCI d Scenario

We first illustrate the inventory calculation by presenting a detailed determination of the energy consumption for the incineration scenario that includes digestion. The INCI d scenario contains seven unit processes (Figure 8.1): thickening, digestion, dehydration, storage, and sludge recovery, followed by fluidized bed incineration, transportation of residue by 40 t truck, and incineration residue storage. The intermediary flows for each unit process are presented in Figure 8.4. They correspond to the flows of matter and energy required per functional unit, namely per ton of treated dry matter sludge. For example, to digest 1 t of dry matter sludge, 65 kWh of electricity and 38 Nm^3 of biogas are necessary, whereas 59 Nm^3 of excess gas are produced.

These intermediary flows are determined from various sources, with preference for values obtained or calculated from data supplied by wastewater treatment plant

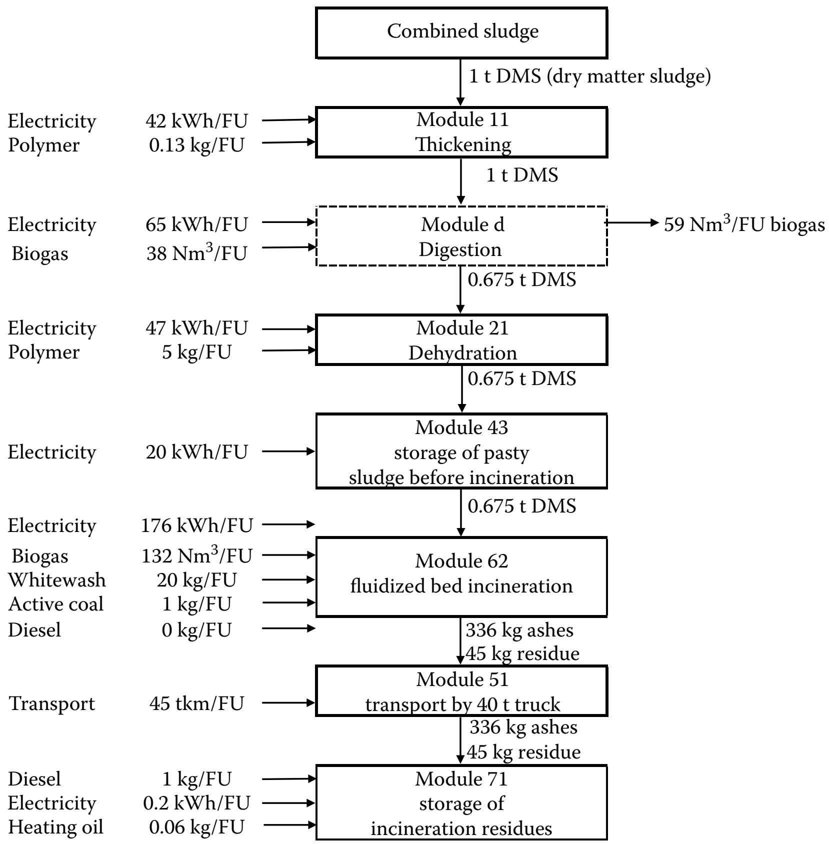


FIGURE 8.4 Intemediary flows of the INCI d scenario.

operators and by project industrial partners. Other data has been taken from previous studies.

It is possible to calculate the nonnewable primary energy consumed per ton of dry sludge matter in the INCI d scenario (Table 8.5) by multiplying each intermediary flow in Figure 8.4 by the required energy consumed to make each unit of these flows available. For the thickening module (11), 41.6 kWh of electricity and 0.13 kg of polymers are required for thickening 1 t of dry sludge; 1 kWh requires the consumption of 13.6 MJ of nonrenewable primary energy, and 1 kg of polymer consumes 41.8 MJ of nonrenewable primary energy. The energy consumption of the module is thus given by: 41.6 kWh/FU × 13.6 MJ/kWh + 0.13 kg/FU × 41.8 MJ/kg = 566 + 5 = 571 MJ of nonrenewable primary energy.

The *substituted energy* part of Table 8.5 corresponds to the various processes that can be used to replace other energy sources. The biogas production during digestion can be substituted for natural gas used to generate hot water. Moreover, heat generated by the incineration fumes can also be recovered to heat water. These two types of substitutions generate energy savings of 9227 MJ nonrenewable primary energy per ton of dry sludge.

In this scenario, the nonrenewable primary energy consumed (10,768 MJ) exceeds the 9,227 MJ saved. The largest energy consumption occurs during incineration (8000 MJ), but the incineration module also allows for the substitution of 3257 MJ for heating hot water. The net consumption would thus be higher if the various substitutions were not actually realized.

8.3.2 Overall Inventory Results

The process described to inventory energy use in Section 8.3.1 is applied to inventory-pollutant emissions across all scenarios, and presented in Table 8.6. The analysis of the different transfer rates of heavy metals (Figure 8.2) shows that results are similar for the different thermal oxidation processes (incineration, wet oxidation, pyrolysis, and cementary incineration), representing 25% of the significantly higher transfers in case of agricultural application. For cadmium, mercury, and zinc, the landfill scenario transfers amount to approximately double the thermal oxidation transfers.

8.4 IMPACT ASSESSMENT

Two methods were used for impact assessment: Eco-indicator 99 (Goedkoop et al. 1998; Goedkoop and Spriensma 1999) and Impact 2002+ (Jolliet et al. 2003). Since Eco-indicator 99 did not have factors available for some sludge micropollutants, complementary factors were developed for these substances. The present analysis focuses on four impact categories: nonrenewable primary energy, global warming, human toxicity, and ecotoxicity. Impacts in other impact categories, such as acidification and resource use, are highly correlated with nonrenewable primary energy (Huijbregts et al. 2010) and are therefore not reported separately here.

TABLE 8.5
INCI d Nonrenewable Primary Energy Assessment Calculations

Energy Consumption			Intermediary Flows per FU	Energy per Unit Flow[a]	Energy per FU
11	Thickening				
		Electricity	41.6 kWh/FU	13.6 MJ/kWh	566 MJ/FU
		Polymer	0.13 kg/FU	41.8 MJ/kg	5 MJ/FU
33	Digestion				
		Electricity	65 kWh/FU	13.6 MJ/kWh	884 MJ/FU
21	Dehydration				
		Electricity	46.6 kWh/FU	13.6 MJ/kWh	633 MJ/FU
		Polymer	4.7 kg/FU	41.8 MJ/kg	198 MJ/FU
44	Storage and pasty sludge recovery prior to incineration				
		Electricity	19.8 kWh/FU	13.6 MJ/kWh	269 MJ/FU
62	Fluidized bed incineration				
		Electricity	175.5 kWh/FU	13.6 MJ/kWh	2,387 MJ/FU
		Natural gas	131.6 Nm^3/FU	39.25 MJ/Nm^3	5,166 MJ/FU
		Whitewash	20.3 kg/FU	2.8 MJ/kgCaO	57 MJ/FU
		Active coal	1.4 kg/FU	70.5 MJ/kg	99 MJ/FU
		Diesel	7.2 kg/FU	46.7 MJ/kg	336 MJ/FU
51	Residue transport by 40 t truck				
		Transport	45.3 tkm/FU	2.6 MJ/tkm	118 MJ/FU
71	Incineration residue storage				
		Diesel	1 kg/FU	46.7 MJ/kg	44 MJ/FU
		Electricity	0.2 kWh/FU	13.6 MJ/kWh	3 MJ/FU
		Heating oil	0.06 kg/FU	44.4 MJ/kg	3 MJ/FU
Total					10,768 MJ/FU
	Substituted Energy		**Substituted Fuel**	**Energy per Unit**	**Energy per FU**
33	Digestion biogas production				
		Produced biogas	234 Nm^3/FU		
		Substituted natural gas[b]	152.1 Nm^3/FU	39.25 MJ/Nm^3	5,970 MJ/FU
62	Fluidized bed incineration: Heat recovery				
		Energy savings	2 390 MJ/FU		
		Substituted natural gas	83.0 Nm^3/FU	39.25 MJ/Nm^3	3,257 MJ/FU
Total					9,227 MJ/FU
Balance					1,776 MJ/FU

[a] Nonrenewable primary energy.

[b] The substituted energy is calculated by multiplying the volume of biogas produced by the lower heat value of 23.4 MJ/Nm^3 for biogas. The volume of natural gas substituted is then calculated by dividing this energy by the heat production efficiency of 0.8, then by assuming a natural gas substitution (lower heat value of 36 MJ/Nm^3).

TABLE 8.6
Detailed Pollutant Emissions Inventory for the Different Sludge Treatment Scenarios

Substance	Category	Unit	AGRI	AGRI d	INCI	INCI d	WETOX	WETOX d	PYRO	PYRO d	CEME	CEME d	LANDF	LANDF d
Ag	Soil	g	9.6	9.6	1.23	1.23	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25
Ag	Air	mg	—	—	125	125	0	0	0	0	0	0	0	0
Ag	Water	g	0.401	0.4	1.23	1.23	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25
Al	Soil	g	11.6	9.27	0.662	1.17	0.842	–0.85	3.86	3.67	2.97	1.37	4.09	1.57
Al	Water	g	111	92.8	102	84.6	303	224	233	153	–389	–209	162	134
Al	Air	g	8.42	6.71	4.98	4.01	12.5	9.2	10.3	6.62	–51.6	–30.1	13.5	10.2
Aldehydes	Air	g	–0.359	–0.416	0.00231	0.00187	0.00956	0.00677	0.0057	0.00367	0.00228	0.00197	6	5.7
Alkanes	Air	g	–4.06	–3.98	0.997	0.836	2.88	2.15	4.81	3.3	–4.74	–2.43	2.53	2.05
Alkenes	Air	g	0.0625	0.0127	0.417	0.339	1.15	0.856	1.06	0.686	–2.43	–1.35	0.859	0.685
AOX	Water	mg	1.92	1.36	0.952	0.848	2.46	1.75	3.9	2.79	–2.16	–1.04	3.84	3.06
As	Soil	g	4.68	4.68	0.617	0.617	0.5	0.499	0.522	0.522	0.476	0.475	0.626	0.625
As	Water	mg	144	52.4	819	781	607	449	565	407	–699	–330	958	888
As	Air	mg	30.5	23.4	78.7	75.8	49.2	37.8	37.9	26.2	301	367	45.5	34.1
B	Air	g	3.3	3.07	3.88	3.23	10.5	8.16	9.05	5.79	–1.79	–0.0647	3.3	3.08
Ba	Water	g	10.3	6.94	9.78	8.15	28.4	21.1	26.7	17.6	–39	–21	17.9	13.8
Benzene	Water	mg	181	112	77.8	70.5	213	136	425	306	–345	–189	385	293
Benzo[*a*] pyrene	Soil	mg	576	576	—	—	—	—	—	—	—	—	—	—
Benzo[*a*] pyrene	Water	mg	24	24	—	—	—	—	—	—	—	—	12	12
Benzo[*a*] pyrene	Air	μg	951	641	148	123	514	361	591	467	431	308	922	685
Benzo[*b*] fluoranthene	Soil	mg	576	576	—	—	–	—	—	—	—	—	150	150
Benzo[*b*] fluoranthene	Water	mg	24	24	—	—	—	—	—	—	—	—	12	12

(*Continued*)

TABLE 8.6 (Continued)
Detailed Pollutant Emissions Inventory for the Different Sludge Treatment Scenarios

Substance	Category	Unit	AGRI	AGRI d	INCI	INCI d	WETOX	WETOX d	PYRO	PYRO d	CEME	CEME d	LANDF	LANDF d
BOD	Water	g	1.15	0.741	0.191	0.206	0.524	0.349	0.834	0.669	−1.01	−0.558	1.24	0.902
Cd	Soil	g	6.36	5.87	1.4	1.4	1.08	1.08	1.15	1.15	1.02	1.02	1.98	1.98
Cd	Water	g	0.147	0.0939	1.41	1.41	0.0574	0.0523	0.294	0.288	0.834	0.844	2.03	2.02
Cd	Air	g	−0.00354	−0.00916	0.0198	0.0187	0.067	0.0627	0.0499	0.0451	0.459	0.461	1.99	1.99
Chloride	Water	kg	1.27	0.711	1.1	0.946	2.97	2.18	3.36	2.33	−3.26	−1.68	6.15	5.31
CO	Air	kg	0.355	0.169	0.491	0.318	1.39	0.759	1.54	1.06	0.982	0.673	0.505	0.309
CO_2	Air	kg	370	−35.2	129	39.5	133	16.7	324	58.1	−75.1	−133	569	93.1
Cobalt	Soil	g	4.36	4.42	0.617	0.617	0.624	0.624	0.625	0.625	0.492	0.492	0.625	0.625
Cobalt	Water	g	0.429	0.381	0.818	0.78	1.22	1.06	1.09	0.927	0.229	0.596	0.955	0.886
Cobalt	Air	mg	23.4	20.3	88	83.6	87.8	65.6	60.9	40.4	386	448	23.3	20.4
COD	Water	kg	1.68	1.68	1.5	1.5	36.4	36.4	1.51	1.5	1.5	1.5	1.82	1.82
Cr	Soil	g	79.4	77	0.883	0.886	0.873	0.864	0.891	0.89	0.937	0.928	12.5	12.5
Cr	Air	g	0.0401	0.0313	0.537	0.533	0.172	0.153	0.0595	0.0397	13	13	0.0479	0.0369
Cr (III)	Water	g	3.78	3.27	1.89	1.71	3.03	2.23	2.46	1.65	7.41	9.2	14.2	13.8
Cr (VI)	Water	mg	0.48	0.353	0.219	0.176	0.603	0.446	0.478	0.306	−1.93	−1.11	0.541	0.41
Cu	Soil	g	477	476	4.43	4.43	31.2	31.2	31.2	31.2	29.5	29.5	72.6	72.6
Cu	Water	g	19.1	18.8	0.54	0.452	32.7	32.3	32.4	32	55.4	56.3	73.4	73.3
Cu	Air	g	0.11	0.0869	2.75	2.74	0.474	0.419	0.309	0.258	−0.198	−0.107	0.159	0.123
Cyanide	Water	mg	36.5	22.5	10.5	9.71	30.8	22.8	37.1	28	14.7	12.5	44.8	34.8
Dioxin	Soil	mg	48	48	—	—	—	—	—	—	—	—	12.5	12.5
Dioxin	Water	mg	2	2	—	—	—	—	—	—	—	—	1	1
Dioxin	Air	μg	—	—	541	362	0	0	0	0	438	438	0	0
F	Water	kg	−4.79	−5.51	0.000373	0.000374	0.00118	0.00078	0.00148	0.00115	0.000464	0.000339	0.00257	0.00195
Fe	Soil	g	4.06	−0.661	−1.08	−0.146	−0.966	−4.34	4.97	4.73	3.38	0.195	5.74	0.608

Fe	Water	kg	−279	−320	0.193	0.16	0.555	0.419	0.456	0.288	0.0429	0.0774	0.141	0.137
Fluoranthene	Soil	g	1.44	1.44	—	—	—	—	—	—	—	—	0.375	0.375
Fluoranthene	Water	mg	60	60	—	—	—	—	—	—	—	—	30	30
Formaldehyde	Air	g	0.647	0.276	0.182	−0.072	1.74	0.963	1.39	0.981	−0.745	−0.479	0.842	0.451
Furan	Soil	mg	4.8	4.8	—	—	—	—	—	—	—	—	1.25	1.25
Furan	Water	μg	200	200	—	—	—	—	—	—	—	—	100	100
Furan	Air	μg	—	—	541	362	0	0	0	0	438	438	0	0
H_2S	Air	g	0.321	−1.26	−0.981	−0.533	−1.76	−2.69	−0.807	−0.101	5.06	2.13	1.02	−0.636
HCl	Air	g	61	50.3	58.1	48	157	113	169	99.4	−219	−120	134	115
Heat losses	Soil	MJ	2.42	−2.81	−6.61	−8.92	10.6	1.29	−0.73	0.932	7.87	1.87	10.7	4.73
Heat losses	Water	kWh	−12.2	−7.52	12.4	−17.3	589	427	1.05×10^7	7.28×10^6	1.05×10^7	7.28×10^6	2.04	3.55
Heat losses	Air	kWh	1.30×10^3	93.5	340	−659	2.69×10^3	1.31×10^3	2.42×10^3	1.92×10^3	2.34×10^3	1.11×10^3	2.49×10^3	1.15×10^3
HF	Air	g	5.38	4.76	7	5.77	18.8	14.6	19	11.6	−2.01	0.572	10.3	9.46
Hg	Soil	g	3.65	3.65	0.0372	0.0372	0.37	0.37	0.324	0.324	0.329	0.329	0.882	0.882
Hg	Air	g	1.16	1.16	0.837	0.835	0.0893	0.0774	0.723	0.711	0.221	0.224	0.896	0.894
Hg	Water	mg	−72.1	−113	37.2	37.2	2.74	2.41	2	2	401	401	883	882
Hydrocarbons	Water	mg	14.8	11.5	19.5	15.9	77.9	55.4	57.7	38	8.13	10.6	27	24.2
Land use II–IV	Nonmat.	m^2y	12.1	8.31	2.68	2.88	3.61	3.62	3.39	3.41	2.95	2.33	17.4	12.2
Methane	Air	kg	14.4	6.78	0.177	0.136	0.894	0.307	0.711	0.606	0.0958	−0.145	112	47.8
Mn	Soil	g	288	288	18.7	18.7	37.5	37.4	37.6	37.6	35.5	35.4	37.6	37.5
Mn	Air	g	0.435	0.301	0.439	0.436	0.419	0.371	0.374	0.313	13.4	13.4	0.311	0.222
Mn	Water	g	14.9	14.4	21	20.5	44.5	42.6	42.8	40.9	31.1	34.8	41.1	40.4
Mo	Air	mg	10.6	8.96	16.1	14.3	33.1	25.2	25	16.8	171	194	12.5	10.3
Mo	Soil	g	4.8	4.8	0.441	0.441	0.625	0.625	0.625	0.625	0.589	0.589	0.625	0.625
Mo	Water	g	0.642	0.567	0.771	0.715	1.66	1.39	1.4	1.13	−0.538	0.0167	1.15	1.05
N	Water	g	170	170	32.4	32.4	417	417	61.7	61.7	61.7	61.7	41.4	41.4
N_2O	Air	g	−159	−133	8.14	7.47	102	96.1	20.5	18.5	8.67	10.7	27.6	23.9
NH_3	Air	kg	1.3	1.32	0.00533	0.00477	0.268	0.158	0.00874	0.00769	0.0067	0.00666	0.855	0.849
NH_3	Water	kg	6.46	6.46	2.58	2.58	38.7	38.7	4.94	4.94	4.94	4.94	10.9	10.6

(*Continued*)

TABLE 8.6 (Continued)
Detailed Pollutant Emissions Inventory for the Different Sludge Treatment Scenarios

Substance	Category	Unit	AGRI	AGRI d	INCI	INCI d	WETOX	WETOX d	PYRO	PYRO d	CEME	CEME d	LANDF	LANDF d
Ni	Soil	g	44.2	44	6.17	6.17	6.24	6.24	6.25	6.25	5.68	5.68	6.25	6.25
Ni	Water	g	1.42	1.12	6.68	6.59	7.7	7.35	7.42	7	5.54	6.45	7.09	6.92
Ni	Air	g	−0.279	−0.429	0.817	0.784	0.644	0.486	0.474	0.328	0.464	1.4	0.266	0.211
Nitrate	Water	kg	0.00509	0.00383	14	14	203	203	39.8	39.8	42.6	42.6	48.7	47.3
Nitrite	Water	kg	0	0	0.225	0.225	3.14	3.14	3.52	3.52	0.43	0.43	0.304	0.304
NMHC	Air	kg	0.162	0.0056	0.118	0.108	0.258	0.147	0.894	0.641	−0.012	0.0319	5.14	2.3
NO_x	Air	kg	1.09	0.748	0.808	0.579	1.55	1.29	2.89	1.97	1.99	1.2	2.03	1.57
P	Air	g	−5.56	−6.43	0.0633	0.0514	0.177	0.134	0.142	0.0911	−0.473	−0.271	11.1	10.6
Particles	Air	kg	3.73	2.53	0.758	0.545	0.562	0.42	1.42	0.944	0.363	0.247	7.75	5.39
Pb	Soil	g	288	288	37.4	37.4	8.62	8.62	8.72	8.72	8.38	8.38	0.755	0.755
Pb	Water	g	11.5	11.2	38	37.9	1.83	1.38	3.11	2.64	4.01	4.87	1.73	1.53
Pb	Air	g	0.182	0.111	0.835	0.824	0.347	0.288	0.476	0.431	9.99	10.4	0.302	0.217
PCBs	Soil	mg	480	480	—	—	—	—	—	—	—	—	125	125
PCBs	Water	mg	20	20	—	—	—	—	—	—	—	—	10	10
Phenol	Water	mg	76.5	−24.5	89.5	82.7	248	160	432	315	−289	−151	409	312
Phosphate	Water	kg	−3.72	−4.31	0.242	0.241	0.0177	0.0134	0.25	0.245	−0.0235	−0.0127	13.4	12.7
Sb	Soil	g	9.6	9.6	1.24	1.24	1.25	1.25	1.25	1.25	1.17	1.17	1.25	1.25

Sb	Air	mg	4.43	3.98	70.5	69.7	19.3	16.2	15.8	11.8	−6.7	−2.83	4.94	4.34
Sb	Water	g	0.403	0.402	1.24	1.24	1.25	1.25	1.25	1.25	1.79	1.79	1.25	1.25
Se	Soil	g	2.88	2.88	0.0905	0.0905	0.375	0.375	0.375	0.375	0.335	0.335	0.375	0.375
Se	Water	g	0.692	0.585	0.603	0.511	1.94	1.49	1.54	1.13	−1.65	−0.734	1.2	1.04
Se	Air	mg	66.5	54.2	78.3	70.9	147	107	114	75.7	212	269	64.4	53.1
Sn	Soil	g	9.6	9.6	1.24	1.24	1.25	1.25	1.25	1.25	1.18	1.18	1.25	1.25
Sn	Water	g	0.401	0.401	1.24	1.24	1.25	1.25	1.25	1.25	1.3	1.3	1.25	1.25
Sn	Air	mg	1.38	1.21	67.2	66.9	10.4	9.37	4.46	3.3	443	444	1.57	1.35
SO_x (as SO_2)	Air	kg	0.236	−0.225	1.51	1.26	4.49	3.3	4.34	2.81	−0.842	−0.289	6.89	6.33
Sulfide	Water	kg	1.88	1.88	1.88	1.88	6.07×10^{-5}	4.31×10^{-5}	1.88	1.88	1.88	1.88	1.93	1.93
Suspended substances	Water	g	11.7	11.7	—	—	—	—	—	—	—	—	—	—
Tributyl tin	Water	mg	−9.94	−13.1	2.44	2.06	6.92	5.01	7.14	4.87	−11.6	−6.38	6.78	5.3
V	Air	g	0.683	0.544	0.601	0.486	1.97	1.42	1.48	0.983	−4.54	−2.6	0.779	0.617
Water	Raw	t	9.37	8.28	11.1	8.93	53.5	23.7	69.1	46.4	54.1	39.1	8.57	7.75
Zn	Soil	kg	1.32	1.32	0.175	0.175	0.17	0.17	0.153	0.153	0.172	0.172	0.361	0.361
Zn	Water	g	57.5	57.2	176	176	113	112	102	101	273	275	363	362
Zn	Air	g	1.26	0.873	2.18	2.16	1.4	1.3	0.767	0.65	−0.268	−0.106	1.8	1.26

8.4.1 Energy Consumption

As illustrated in Section 8.3.2 for the INCI scenario, the nonrenewable primary energy consumption in each scenario (Figure 8.4) is obtained by multiplying the intermediary flows of Tables 8.1 and 8.3 by the nonrenewable energy per unit input of Table 8.4. Energy consumption includes the feedstock energy in the product, along with the energy needed to extract, transform, and transport it. Avoided energy use due to substitutions is then subtracted, yielding the net primary energy consumption for each scenario (Figure 8.5).

Main contributions to the energy consumption include the wet oxidation sludge treatment process (20.7 GJ/tDM), drying (16.4 GJ/tDM) used in pyrolysis and cement kilns, and the pyrolysis sludge treatment process (8.9 GJ/tDM). Since the sludges are dehydrated in these scenarios, transport energy consumption is relatively low, as observed by Neumayr (1999). Energy for pure industrial water treatment, site infrastructure, dry matter losses, and wastewater treatment from sludge treatment processes are negligible. Substitutions are important in the nonrenewable energy category, since they can compensate for more than 50% of the energy consumption, especially for the wet oxidation, pyrolysis, and cement scenarios. The introduction of anaerobic digestion substantially reduces energy consumption for each process (by 6 GJ/tDM), but its total influence varies with each scenario, because it also reduces the lower calorific heating value of the sludge and therefore also reduces the substituted energy (see Figure 8.6 for the ultimate energy balance of all scenarios, including ones that account for digestion).

As might be expected, agriculture spreading with digestion (AGRI d) has the most favorable energy balance, as it provides more energy than it consumes. If digestion is not included, however, the agriculture scenario (AGRI) requires more energy than high-quality incineration with heat recovery (INCI) (Figure 8.6). Considering all 12 scenarios, AGRI d, LANDF d, and CEME d have the lowest net energy consumptions as long as the sludge is used to effectively replace fossil fuels in cement kilns; if sludge is used to replace other types of waste, cement kiln incineration becomes the worst scenario. Wet oxidation has the highest energy consumption (14 GJ/tDM), with a net consumption of 8 GJ/tDM even after substitution. For digested sludges, thermal oxidation scenarios (incineration [INCI d], pyrolysis [PYRO d], wet oxidation [WETOX d], and cement [CEME d]) have very similar and relatively limited net energy consumptions.

To put into context the amounts of energy involved for sludge treatment in these scenarios, we can compare them with the primary energy required for pumping drinking water. According to Crettaz et al. (1999), 0.43 kWh of Swiss average electricity are needed per cubic meter pumped freshwater, corresponding to 3.7 MJ of nonrenewable primary energy per cubic meter. About 0.27 kg DM of sludge is produced per cubic meter of treated water. Since the various scenarios range in net energy balance from −4 to +14 MJ/kg DM, this corresponds to between −1.1 and +3.8 MJ/m^3 treated water and is thus of the same order of magnitude as the energy needed to pump the corresponding amount of freshwater.

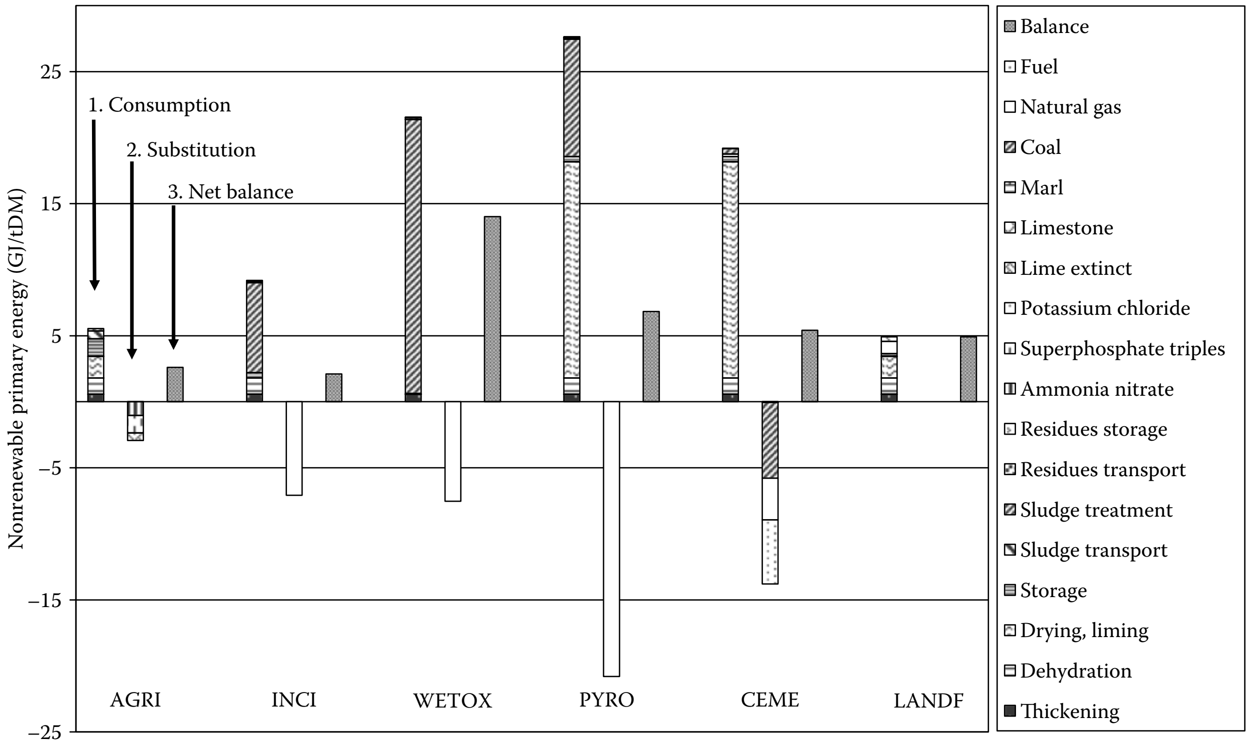


FIGURE 8.5 Nonrenewable primary energy used in the undigested sludge treatment scenarios. Each scenario has three columns: 1. Energy used for the treatment of 1 tDM; 2. Energy saved by substitutions; 3. Net primary energy consumption.

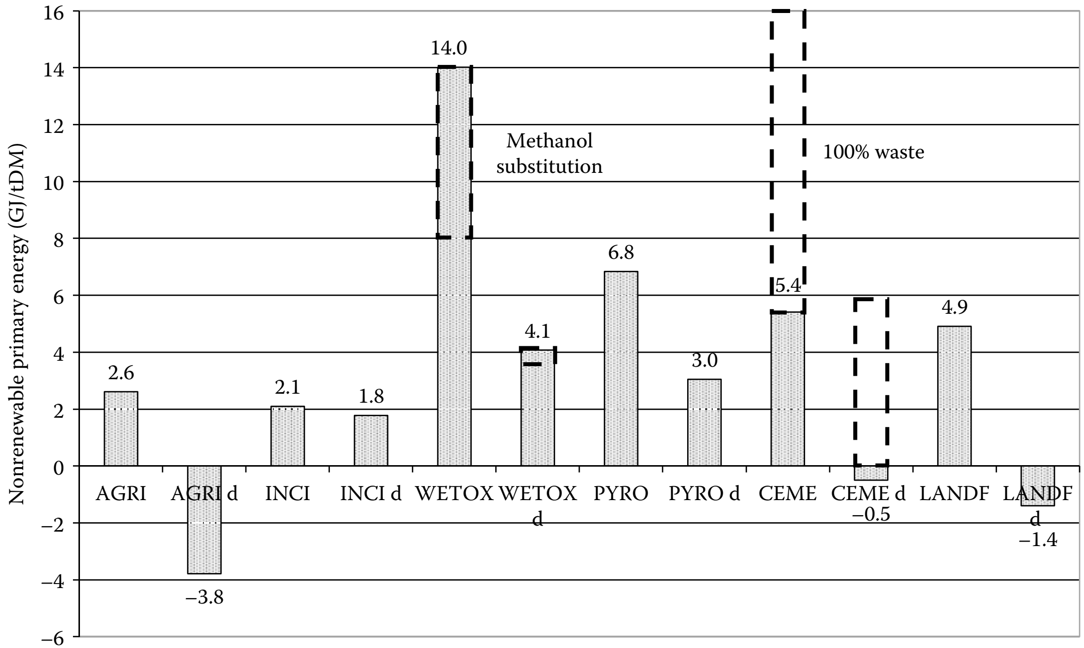


FIGURE 8.6 Net nonrenewable primary energy consumption of the 12 sludge treatment scenarios without and with digestion (d). The dashed lines present results in the case where sludges substitute other waste for cement kiln incineration and in the case of methanol recovery for wet oxidation.

8.4.2 Global Warming

Global warming scores are calculated by multiplying the intermediary flows of Table 8.1 by the emission factors of carbon dioxide and methane per unit input of Table 8.4. The direct methane emissions from organic matter degradation are then added to yield the total emissions for each greenhouse gas, which are then multiplied by the IPCC global warming potentials for a 500-year time horizon (to account for long-term effects). Nitrous oxide (N_2O) emissions due to input processes are negligible, and direct N_2O emissions in the sludge treatment processes were not included due to lack of available data. The avoided emissions associated with substitutions (such as production and spreading of fertilizers) are subtracted. The same amount of 1241 kg_{CO2}/tDM biogenic CO_2 is subtracted from the treatment emissions in all scenarios, to account for the CO_2 fixed prior to wastewater treatment (e.g., during the agricultural production of food consumed by humans). Figure 8.7 presents the contributions of each scenario to the greenhouse effect.

Methane emissions are substantial in the landfill scenarios for both digested and undigested sludge (56% and 76%, respectively), leading these scenarios to have the highest global warming scores. The undigested agricultural spreading scenario (AGRI) also has a high climate change impact due to carbon dioxide emissions in lime production (72%) and methane emissions (21%). In contrast, the agriculture spread of digested sludge (AGRI d) results in quite low greenhouse gas impacts, since the produced biogas increases the substituted energy, and the amount of sludge—and therefore lime—is reduced. In the case of the thermal oxidation scenarios, digested sludges have close to equivalent greenhouse gas impacts, where incineration in cement kilns is slightly better than other scenarios if fossil fuels are effectively substituted.

Other impact categories such as acidification and respiratory inorganics were not analyzed by Houillon and Jolliet, but the emissions of associated substances, such as NO_x, SO_2, and particulate matter, are generally correlated with greenhouse gas emissions, except for the landfill scenarios.

8.4.3 Human Toxicity and Ecotoxicity

Human and ecosystem toxicity scores were calculated based on pollutant emissions, the transfer rates of organic and metallic micropollutants, and the corresponding characterization factors for Eco-indicator 99 and IMPACT 2002+. Due the high uncertainties over these characterization factors, results must be considered as comparative rather than absolute values, as presented in Table 8.7.

LANDF and AGRI lead to higher human toxicity scores than the thermal oxidation scenarios. For agriculture use, this is due to the micropollutant emissions on agricultural lands, leading to their partial transfer into the food chain. This food chain exposure is important, as Bennett et al. (2002) demonstrated that intake fractions of persistent pollutants can be significantly higher through food ingestion than through water ingestion or inhalation. The landfill scenario also results in exposure to micropollutants, but a fraction is stored in the ground, resulting in lower effects on human toxicity. Both the IMPACT 2002+ and Eco-indicator 99 methods show

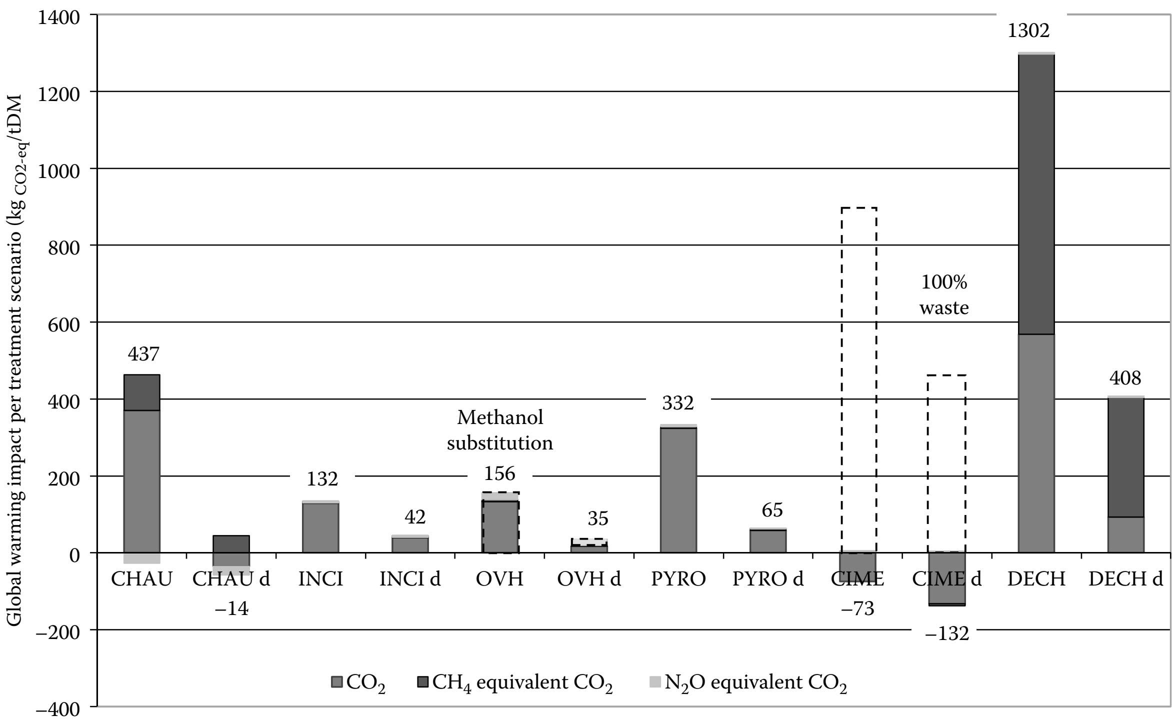


FIGURE 8.7 Net global warming scores for the 12 sludge treatment scenarios. The dashed lines present results in the case where sludge substitutes other waste for cement kiln incineration and in the case of methanol recovery for wet oxidation.

TABLE 8.7
Relative Impact Assessment Scores for Human and Ecosystem Toxicity

Scenario	Thermal Oxidation (INCI, WETOX, PYRO, CEME)	AGRI	LANDF
Average percentage of micropollutants transferred	25%	100%	30%
Human toxicity (relative to thermal oxidation)	1	8	4
Ecotoxicity (points) (relative to thermal oxidation)	1	8	2

that all scenarios involving thermal oxidation lead to similar and better results. The ranking of scenarios for ecotoxicological impacts is similar to human toxicity, with all scenarios involving thermal oxidation also scoring best.

When looking at overall impact on human health and ecosystems, accounting for all other impact categories of the IMPACT 2002+ method, the scenario ranking remains similar to the results expressed for human toxicity and ecotoxicity in Table 8.7.

8.5 INTERPRETATION AND RECOMMENDATIONS

Energy consumption and carbon emission results for each reference scenario are discussed in more detail and compared with other literature values in Section 8.5.1, and sensitivity studies are carried out in Section 8.5.2, leading to the final recommendations in Section 8.5.3.

8.5.1 REFERENCE SCENARIOS

8.5.1.1 Agricultural Landspreading of Limed Pasty Sludge

The energy consumption of potassium chloride in this scenario is negligible compared with that of ammonium nitrate (1 GJ/tDM) and superphosphate triples (1.3 GJ/tDM). Liming and storage require substantial amounts of energy (Figure 8.4), which is confirmed by Kobayashi and Sago (2000). The minor energy consumption of dehydrated sludge transport is confirmed by Neumayr (1999), whereas transport can play an important role for sludges with higher water content and greater transport distances (Rebitzer et al. 2003). Digesting sludge reduces the amount of dry matter and thus also limits the impacts of liming and transportation. Compared with undigested sludge, the energy saved from fertilizer substitution of digested sludge is similar, while digestion enables an additional substitution of natural gas. The total energy balance of agricultural landspreading of digested sludge (−3.8 GJ/tDM in Figure 8.5) is close to the value of −3.5 GJ/tDM obtained by Remelle (1995).

In terms of greenhouse gas emissions, lime production emits a significant amount of CO_2 (583 kg_{CO2}/tDM), whereas lime substitution avoids the emission of 270 kg_{CO2}/tDM. Methane is emitted due to anaerobic digestion of organic matter during storage and spreading. Similar to energy consumption, the transport contributions to CO_2 emissions are low (32.6 kg tDM), as also observed by Müller et al. (1999).

8.5.1.2 Incineration in Fluidized Bed of Pasty Sludge

Fluidized bed incineration energy consumption is mainly due to electricity consumption and the burning of natural gas, as also observed by Ministerium für Umwelt und Naturschutz (2001; 2.65 GJ/tDM). Heat recovery reduces the primary energy requirements by a factor of three and avoids the emission of 335 kg_{CO2}/tDM. The net balance of 42 kg_{CO2}/tDM for the incineration of digested sludge is close to the results obtained by Suh (1999), Suh and Rousseaux (2002), with 50 kg_{CO2}/tDM. For this sludge treatment process, there is little difference between the digested and undigested scenarios (Figure 8.5). This is due to the need for additional natural gas to burn digested sludges, thus reducing the benefit of the recuperated gas during the digestion process. In France, according to Prouve (1994), most fluidized beds treat undigested sludges.

8.5.1.3 Wet Oxidation of Liquid Sludge

The main energy consumption and carbon emissions during wet oxidation arise from the use of electricity (405 kg_{CO2}/tDM) and the energy-intensive production of oxygen (281 kg_{CO2}/tDM). The avoided energy and emissions linked to heat recovery (−7.5 GJ/tDM and −356 kg_{CO2}/tDM) are similar to those of fluidized bed incineration (−7.1 GJ/tDM). Digestion allows for a large reduction of energy consumption, due to greatly reduced organic matter and thus electricity and oxygen consumption.

Methanol recovery is considered a viable option for only the undigested sludge (175 kg methanol/tDM), as it is rather low for digested sludge (20 kg methanol/tDM). Such a methanol recovery substantially reduces the energy consumption from 14 down to 7.9 GJ/tDM and enables the greenhouse gas balance to become close to zero by avoiding the emission of 154 kg_{CO2}/tDM (dashed lines in the WETOX scenarios of Figures 8.5 and 8.6).

8.5.1.4 Pyrolysis of Dried Sludge

The energy demand and greenhouse gas emissions of pyrolysis are dominated by electricity and natural gas consumption in the drying process (6.8 GJ/tDM and 997 kg_{CO2}/tDM). Pyrolysis gas substitution is essential to this treatment, in that it replaces 75% of the process consumption (−17.7 GJ/tDM and −835 kg_{CO2}/tDM), making it important to have it fully recovered. Heat recovery during the drying process results in another smaller reduction in CO_2 emissions (−148 kg_{CO2}/tDM). For digested sludge, drying still requires a substantial amount of energy.

8.5.1.5 Incineration of Dried Sludge in Cement Kilns

Although the direct drying system of incineration requires substantial energy consumption, dried sludge enables fuel and coal substitution in the cement factory. This leads to a better energy and greenhouse gas balance than for pyrolysis, as also observed by Chassot and Candinas (1997). Applied to digested sludge, even more energy is substituted, leading to a net avoidance of energy (−0.5 GJ/tDM) and CO_2 emissions (negative balance of −132 kg_{CO2}/tDM), as also found by Sasse et al. (1999).

Substitutions compensate for about 72% of the energy consumed in this disposal option, making it essential to know the types of fuel that are substituted. This scenario assumes a substitution of coal and fuel in the cement factory, but the cement industry also uses waste (such as tires and plastics) as substitution fuels. If sludges are not substituting other waste rather than fossil fuels, no bonus should be credited in the LCA methodology. This would increase energy consumption to 16 GJ/tDM for undigested sludge and 6 GJ/tDM for digested sludge, making it the worst scenario in terms of greenhouse gas emissions and energy consumption. Benz et al. (1995) also noticed this problem, indicating that cement kiln incineration is better than fluidized bed incineration only if coal is chosen as a substitution. Using coal has such a high impact that some may propose simply eliminating the use of coal rather than crediting the sludge substitution.

8.5.1.6 Landfilling of Limed Pasty Sludge

The energy demand for landfilling is substantial (4.9 GJ/tDM), being about double that for incineration in fluidized beds or agricultural spreading. This is due to dehydration, liming, and transport, and to the lack of substitutions for undigested sludges. Both energy consumption and greenhouse gas emissions are strongly reduced by sludge digestion, due to its reduction in liming, storage, and transport. Another drawback to landfilling is that biogas burned in the landfill is generally not recovered, in contrast to biogas produced at the WWTP. Figure 8.6 shows how important it is for landfill disposal to burn methane emitted into the air by the organic fermentation to convert it into carbon dioxide. This operation is considered in the study by assuming that 60% of the methane is burned in a flare (Suh 1999). The remaining 40% of the methane still provides a substantial contribution of about 50% of the landfill global warming balance, leading to the worst score of all disposal routes. Landfilling undigested sludge cannot therefore be recommended.

8.5.2 Sensitivity Analyses

8.5.2.1 Transport Distances

Changing transport distances affects each scenario differently. For the agricultural spreading of sludge, multiplying transport distances by a factor of two increases energy consumption by about 0.6 GJ/tDM and CO_2 emissions by 32 kg_{CO2}/tDM. So, although AGRI and INCI have similar energy uses in their reference scenarios, doubling the transport distance leads to a 50% higher energy consumption for the agricultural spreading of undigested sludge compared with undigested sludge incinerated in a fluidized bed (Figure 8.8). The energy consumption of the landfill scenario also increases by 0.9 GJ/tDM when transport distances are doubled. Otherwise, the transport distance does not greatly influence the energy balance, because the sludge has become dried or pasty before being transported (decreasing its weight). This conclusion may be different with liquid sludge, as observed by Dennison et al. (1997) and Rebitzer et al. (2003). The influence of transportation is even lower for digested sludge, due to the decrease in organic matter.

8.5.2.2 Residue Stabilization

The residue of incineration, wet oxidation, and pyrolysis can be stabilized instead of being directly landfilled, so we consider the influence of cement stabilization

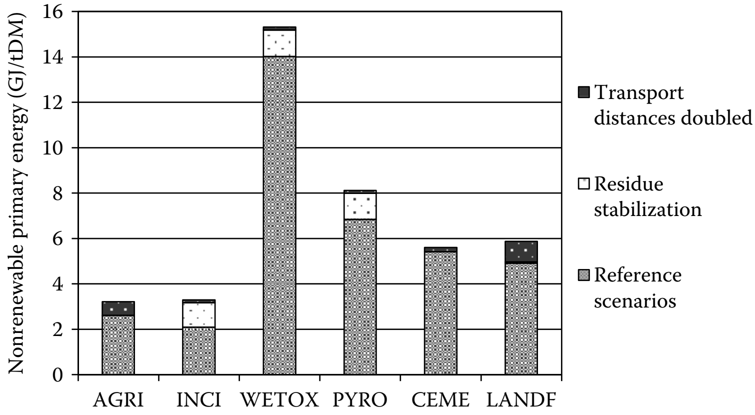


FIGURE 8.8 Sensitivity analyses of the nonrenewable energy balance of the six sludge treatment scenarios without digestion, including residue stabilization and doubling of transport distance.

here. Cement stabilization only moderately increases the energy balances by about 1.2 GJ/t for the three scenarios (INCI, WETOX, and PYRO) to which it is applied (Figure 8.8). Residue stabilization has a greater influence on greenhouse gas emissions, increasing these emissions by 185 kg_{CO2}/tDM for the incineration and wet oxidation scenarios. Consequently, stabilization can only be justified by a substantial reduction in toxicity impacts associated with leachate, which need further studies.

8.5.3 Recommendations and Outlook

As a result of the preceding interpretation and sensitivity studies, we have identified several key processes and areas of improvement:

1. Substitutions (e.g., heat recovery for digestion and thermal oxidation processes and fertilizer replacement) play an important role in all 12 treatments, emphasizing the importance of these flows being recovered and carefully considered in any environmental assessment.
2. Anaerobic sludge digestion should likely be integrated in all processes except incineration, as it simultaneously recovers natural gas while reducing sludge mass, thereby decreasing energy consumption and greenhouse gas emissions during treatment. Digestion also limits the increase in energy consumption if heat is not fully recovered in subsequent processes.
3. Energy and greenhouse gas balances for agricultural spreading are only favorable compared with thermal oxidation processes if sludges are digested. Sludges for agricultural spreading must be stabilized using the minimum amount of lime to limit related carbon dioxide emissions. Micropollutant emissions and their impacts on human and ecosystem toxicity remain substantially higher for spreading than for all other treatments. The removal of heavy metals from sludge could be one solution to improve this scenario,

as well as measures to reduce heavy metal emissions into wastewater at source.

4. Sludge landfilling is unfavorable for most impacts, especially for undigested sludges. If carried out, it is strongly recommended to treat biogas by burning it in a flare, as this dramatically decreases the greenhouse gas emissions by avoiding additional methane releases, thus reducing greenhouse gas emissions from 2381–1302 $kg_{CO2\text{-}eq}$/tDM.
5. The environmental impacts of the different thermal processes (INCI, WETOX, PYRO, and CEME) are approximately equivalent, but high-quality incineration in a fluidized bed remains one of the most promising scenarios. Wet oxidation and pyrolysis do not provide significant impact reductions compared with incineration and require specific conditions to remain competitive; wet oxidation requires the recovery of effluent as methanol and pyrolysis requires the recovery and use of pyrolysis gas. For cement kiln incineration, a drying system at the cement factory site could improve the energy savings and greenhouse gas balance. But, in practice, fresh sludge transport is difficult because of odor, transport, and handling problems. The results obtained by cement kiln incineration are only valid if sludges are used to substitute coal and fuel. Otherwise, this scenario would be the worst of all thermal options; moreover, the less stringent legal requirements for cement kiln emissions compared with incineration plants could also lead to higher human health effects. There are other potential substitutions of by-products resulting from thermal processes, but these require improved residue quality. Roads can be constructed using incineration ashes and wet oxidation mineral residue, and coal can be substituted by the carbonaceous residue from pyrolysis. For future studies, further methodological developments are needed to increase the assessment reliability, especially to account for micropollutant speciation as a function of soil characteristics. As this LCA focuses only on environmental issues, these also have to be balanced against economic costs and social criteria.

9 Metacomparison of the Life Cycle Environmental Impacts of Bio-Based Products

Gregory Houillon, Josef Kaenzig, Jinglan Hong, Andrew Henderson, and Olivier Jolliet

One can produce a large variety of products from biomass, including energy (heat and power), biofuels, chemicals, lubricants, surfactants, solvents, and biopolymers. Often, expected environmental benefits can induce a designer, manufacturer, or industry to switch from a conventional, petrochemical-based product to one derived from biomass. While use of bioproducts often leads to important environmental benefits, there is the potential for negative repercussions, and this balance needs to be better understood. This chapter illustrates the systematic use of life cycle assessment (LCA) for meta-analysis, reviewing the LCA state of the art for a wide range of bio-based products. Such metastudies are usually commissioned by agencies or sponsors that are primarily interested in obtaining a broad overview of the field, rather than detailed results for a single product.

9.1 INTRODUCTION

Bio-based raw materials (e.g., agricultural and forest resources) and products derived from them (e.g., biofuels) have received attention of late as potentially environmentally friendly substances, due to their renewable nature and their ability to substitute for fossil fuels in various applications (Perez-Garcia et al. 2005; Malça and Freire 2006; Gabrielle and Gagnaire 2008; Kim and Dale 2008; Schmehl et al. 2008). Indeed, unlike the case of their fossil-based counterparts, the materials and renewable energies produced from biomass (agriculture, silviculture) can potentially reduce energy consumption and greenhouse gas emissions, and lessen deleterious impacts on air, water, and soil (e.g., Perez-Garcia et al. 2005; Kim and Dale 2008). It is commonly hoped that their use would preserve fossil resources while promoting the adoption of sustainable agricultural practices (Clift 2007; Goldemberg 2007). However, bio-based products can also generate additional environmental impacts that can vary widely from one supply chain to another. Therefore, environmental assessments that quantify these impacts must be conducted to identify the most promising alternatives.

Quantitative analytical tools such as LCA contribute to such assessments, helping to replace preconceived ideas with data-driven findings. One of the challenges and limitations of LCA is to create meaningful comparisons across products and studies, because of variability of assessments, even with the International Organization for Standardization (ISO) framework.

The environmental impact of some classes of bio-based products has been extensively studied using LCA methods, showing important variation depending on the product and the study. Though individual analyses may show positive results—for example, for polylactic acid-based products (Vink et al. 2003, 2004)—broader reviews have not proved a systematic advantage with respect to reduced emissions and energy consumption (Meyer-Aurich et al. 2008). Patel et al. (2005) reviewed bio-based polymers and natural fibers, evaluating the available LCA studies using comparisons based on weight and functional unit (FU). Quirin et al. (2004) and Von Blottnitz and Curran (2007) reviewed the environmental impacts of bioethanol, which is made from varying feedstocks for use as a transportation fuel, in comparison with conventional fuels. However, these reviews usually focus on a single application of biomass and do not allow for comparison across different uses of biomass. Since bio-based material resources are also limited by available land areas, it is of great interest to compare application, identify tendencies, and provide recommendations that may lead to efficient use of biomass. The present chapter therefore addresses the following challenges, aiming at

- The qualitative and quantitative analysis of the environmental impact of a wide range of bio-based products, based on a meta-analysis of LCA studies
- The development of a method to compare nonrenewable energy consumption and greenhouse gas emissions, as well as eutrophication and acidification across various biomass supply chains
- The application of the method to 10 nonfood plant supply chains, helping identify general tendencies in environmental performance
- The identification of key parameters, strengths, and limitations of the LCA approach applied to this field

To address these needs, a meta-analysis was carried out to improve the evaluation of possible environmental gains resulting from switching to plant resource supply chains. This work was adapted from a study initially carried out for the French National Agency for Environmental and Energy Management (Houillon et al. 2004a, 2004b) that reviewed the state of the art in LCA of bio-based products, updating it and extending it to a second stage to additional studies on bioethanol and biodiesel. Rather than presenting data on each supply chain, a comprehensive, quantitative, qualitative, and critical inventory of all available data was created and used to select the most appropriate studies for detailed analysis. Second, we present a unique approach to compare both the absolute and relative gains offered by 10 categories of bioproduct applications (agrimaterials, bioethanol, biodiesel, and agricultural biomass for energy production, biopolymers, surfactants, lubricants and hydraulic fluids, solvents and intermediate chemical products) compared with their fossil counterpart. The metastudy specifically compares the nonrenewable primary energy use and environmental impacts (greenhouse gas emissions, acidification, and eutrophication) of various uses of biomass. Finally, we give recommendations for the development of biomass resource supply chains and suggest avenues for improving LCA knowledge.

9.2 METHODS: META-ANALYSIS OF LCA STUDIES

9.2.1 Overview of LCA Studies

The following approach was developed to compare the results of available LCA studies on bio-based products. Based on criteria including completeness, system boundaries, data consistency, year of publication, and ISO 14044 compliance, a number of LCA studies on 10 nonfood plant supply chains (agrimaterials, bioethanol, biodiesel, and agricultural biomass for energy production, biopolymers, surfactants, lubricants and hydraulic fluids, solvents and intermediate chemical products) were selected for detailed analysis for the different product applications. The meta-analysis was carried out as a stepwise procedure (Table 9.1). Based on a large number of more than 900 LCA references, a first subset of LCA studies was collected. A critical analysis was carried out on these collected studies and only the most relevant in each sector were selected, leading to a limited number of studies of high interest that were analyzed in further detail (see Table 9.1 and CRC Press website, Meta comparison). The supply chains were divided into two groups (Group 1 = more commonly studied areas; Group 2 = less-studied areas) based on the number of LCA studies. The number of available LCA studies varied strongly from one field of application to another. In general, there are many LCA studies on biofuels, energy crops, and timber that have been performed according the ISO 14044 norms. However, fewer

TABLE 9.1
Summary of Biomass Supply Chains and LCA Studies Analyzed

Group	Bio-Based Product	LCA References Collected	LCA Studies Collected	LCA Studies Analyzed
1	Agrimaterials (fiber/wood)	36/132	17/82	5/7
1	Ether alcohols (bioethanol)	216	148	12
1	Ester oils (biodiesel)	203	127	10
1	Forest biomass	114	75	8
2	Agricultural biomass	76	55	5
2	Biopolymers	40	27	9
2	Surfactants	26	13	6
2	Hydraulic oils and lubricants	27	11	4
2	Solvents	9	6	3
2	Chemicals and other intermediate products	11	7	2

studies have been carried out on chemical bioproducts and biomaterials following the ISO standards.

9.2.2 Analysis of Quality and Selection of Studies Analyzed in Detail

These selected studies were then systematically evaluated according to a set of specific criteria: range of scenarios, reliability of studies, technological sensitivity, geographic sensitivity, consistency of results, and additional needs. A qualitative indication of the literature available for the 10 biomass supply chains is shown in Figure 9.1. These criteria are briefly discussed in the following paragraphs.

The diversity of scenarios shows a trend according to the end point for biomass; there is a wider range of scenarios represented in the literature for those applications with a solid product (e.g., fiber and wood), while those applications with a liquid end product (e.g., surfactants) have a lower diversity of scenarios represented in the literature. The available literature contains relatively few studies that are compliant with ISO standards, except in the case of agrimaterials (particularly solid wood materials), biofuels, and forest biomass (bioenergy; heat and electricity). As a result, many LCA results have not been subject to external review, particularly in the case of biopolymers, surfactants, hydraulic oils and lubricants, solvents, and chemical and other intermediates.

In terms of technological development, many biomass supply chains are still in the prototype stages; in contrast, conventional fossil fuel supply chains have been in development for decades. Therefore, the continued development and optimization of biomass supply chains will likely result in higher efficiency and improve their environmental performance, resulting in medium to high sensitivity to technological developments (Figure 9.1). Unlike the technological sensitivity, the geographic sensitivity is largely moderate for plant supply chains as a whole. Geographic effects are most notable in the agricultural production stage, due to climatic differences, and are more rarely seen in the conversion and processing stages.

The consistency of results is highly variable, depending on the supply chain and impact categories. The variations observed are due to the following: differences in LCA methodology among the various studies (boundaries of the system studied, methods of impact assessment, etc.), uncertainties (related to specific pollutant emissions data, knowledge of the agricultural production stage, and biomass conversion processes), and technological knowledge of the supply chains. The exception to the latter is in the case of surfactants, for which technological knowledge is satisfactory, but LCA knowledge is limited. Finally, additional data needs have been identified. These fall broadly into the following categories of difficulty in characterizing supply chains (e.g., changing supply chains and processes) and lack of complete LCA studies (e.g., failure to take all impact categories into account, problems with the selection of appropriate FUs, and failure to take the complete life cycle into account).

9.2.3 Quantitative Comparison of Various Supply Chains across Studies

The comparison of biomass supply chains requires a new methodology. The objective is not to provide typical, specific values for each chain (the state of current

		Diversity of scenarios	Reliability of studies	Technological sensitivity	Geographic sensitivity	Consistency of results	Additional needs
Group 1	Agrimaterials (fiber and wood)	4	2	4	3	2	4
	Ether alchohols (bioethanol)	2	3	5	3	4	4
	Ester oils (biodiesel)	2	3	4	3	4	4
	Forest biomass	4	3	4	3	4	4
Group 2	Agricultural biomass	5	1	5	4	2	5
	Biopolymers	5	3	5	2	4	3
	Surfactants	1	3	4	3	3	4
	Hydraulic oils and lubricants	3	2	5	3	2	4
	Solvents	2	2	4	3	2	4
	Chemical and other intermediates	5	1	?	?	?	5

FIGURE 9.1 Summary of criteria applied to LCA references (1 = very weak, 3 = medium, 5 = very strong). (Adapted from Houillon, G. et al., *Bilan environnemental des filières végétales pour la chimie, les matériaux et l'énergie. Etat des connaissances: Analyse cycle de vie (ACV)*, Synthèse publique, Paris, ADEME. Available from www.ademe.fr/partenaires/agrice/publications/documents_francais/ACV_Synthese.pdf. 2004a).

knowledge does not always allow for this), but to reveal certain trends based on available LCA studies. The nonrenewable primary energy consumption and environmental impacts of bio-based products may be compared with conventional, petrochemical-based products on three levels. One metric is per kilogram of product, but this is possible only if the mass per FU is the same, which is rarely the case. Secondly, products with the same functional unit can be compared in a classic LCA, that is, per functional unit. Such a comparison provides an analysis of gains or emission reduction per functional unit, identifying supply chains and products that require less energy for product manufacture.

However, in a meta-analysis, functions between compared products differ widely when comparing across the various uses of products; furthermore, energy gain or reduction in emissions per functional unit cannot be directly compared between different biomass supply chains. Therefore, a new approach is needed to bring all biomass uses to a similar comparative metric valid across different supply chains. We propose two metrics, an absolute (A) and a relative (B), which are presented in the following subsection (Houillon et al. 2004).

9.2.3.1 (A) Absolute Gain per Hectare of Cultivated Land

First, if the use of available agricultural area is considered, a comparison of absolute gains per hectare of cultivated land is carried out, while taking into account that the biomass is replacing a reference fossil product with a consistent functional unit (e.g., biofuels produced on all available farmland in France could not replace all the fossil fuels used in that country). This absolute comparison pertains to the use of biomass in terms of agricultural production, addressing the question of what kind of biomass and which products make best use of the limited area available for agriculture and lead to the highest environmental benefits, as compared with conventional products. Equation 9.1 represents this metric, namely the gain (or reduction) in impact per biocultivated hectare and year for a given study ($G^{\text{per ha biocultivated-year}}$ [in e.g., MJ/ha-year]):

$$G^{\text{per biocultivated ha-year}} = \left(S^{\text{per FU}}_{\text{conventional fossil}} - S^{\text{per FU}}_{\text{bio-based}}\right) \times N^{\text{per biocultivated ha-year}}_{\text{FU}} \tag{9.1}$$

where:

$S^{\text{per FU}}_{\text{conventional fossil}}$ (MJ/FU) indicates the total impact per FU of the considered study for the conventional fossil scenario of reference

$S^{\text{per FU}}_{\text{bio-based}}$ (MJ/FU) indicates the total impact per functional unit of the bio-based scenario

$N^{\text{per biocultivated ha-year}}_{\text{FU}}$ (FU/ha-year) is the annual production of functional units per cultivated hectare in the bio-based scenario

The studied impacts are the nonrenewable primary energy (in MJ) as well as the global warming impact (in $kg_{CO2\text{-}eq}$), the eutrophication impacts (in $kg_{PO4\text{-}eq}$), and the acidification impacts ($kg_{SO2\text{-}eq}$).

A similar metric was also developed independently and used by Dornburg et al. (2004), comparing the land requirements, energy savings, and reduction in greenhouse gas emissions of bio-based polymers and bioenergy.

9.2.3.2 (B) Gain Relative to the Substituted Part

A more classical metric is to calculate the relative gain due to the bio-based product, relative to the conventional product. However, care must be taken when comparing in a consistent way products that are entirely substituted by the bio-based product and those in which only a part is substituted. In the case where only a part of the total product is substituted, we only divide by the part of the conventional product that is substituted by the biomass, as shown in Equation 9.2.

$$G\% = \frac{\left(S^{\text{per FU}}_{\text{conventional fossil}} - S^{\text{per FU}}_{\text{bio-based}}\right)}{\left(S^{\text{subsituted part per FU}}_{\text{conventional fossil}}\right)} \times 100\% \tag{9.2}$$

In Equation 9.2, the denominator $S^{\text{subsituted part per FU}}_{\text{conventional fossil}} = S^{\text{per FU}}_{\text{conventional fossil}} - S^{\text{per FU}}_{\text{common parts}}$ (in e.g., MJ/FU) is the impact score per functional unit associated only with the part of the conventional fossil product that is substituted by the biofuels. For example, if a bio-based circuit board is to be compared with a conventional circuit board, the non-board components (chips, transistors, solder, etc.) that are common to the fossil and the biofuel scenarios must not be included in the denominator. These common parts can be kept in the numerator of both the total fossil and the total bio-based scenario, since these cancel when calculating the difference.

9.3 RESULTS AND DISCUSSION

9.3.1 Comparison of the Environmental Impacts of Bio-Based Products

9.3.1.1 Comparison across the 10 Categories of Bio-Based Products

Figures 9.2 and 9.3 show comparisons of the environmental benefits for 10 bio-based products categories using the two metrics discussed in the previous subsection: (A) Figure 9.2 per hectare-year of land cultivated and (B) Figure 9.3 for relative environmental efficiency per FU. The CRC Press website provides the detailed references and data used. Data are presented as modified boxplots. Minimum and maximum values for each category are represented by horizontal bars connected to the main box body by vertical lines. The main box body indicates the twenty-fifth and seventy-fifth percentiles of the data, and the middle line indicates the median value, to look at general tendencies. In cases of product categories with few data points, some of these statistical descriptors may be nonexistent. Since the data are not necessarily a representative sample of any of the considered product categories, and because each application leads to individually potentially valid results, it is important to also account for the full ranges of variation in the interpretation.

For energy consumption and global warming (panels a and b of Figures 9.2 and 9.3), almost all of the bioproduct LCAs show significant benefits over conventional fossil products (positive gains). However, eutrophication impacts (Figures 9.2c and 9.3c) are often increased due to emissions, mainly of phosphate or nitrogen applied as fertilizer, during the agricultural production stage of the bioproduct. As far as acidification impacts (Figures 9.2d and 9.3d) are concerned, reported data on

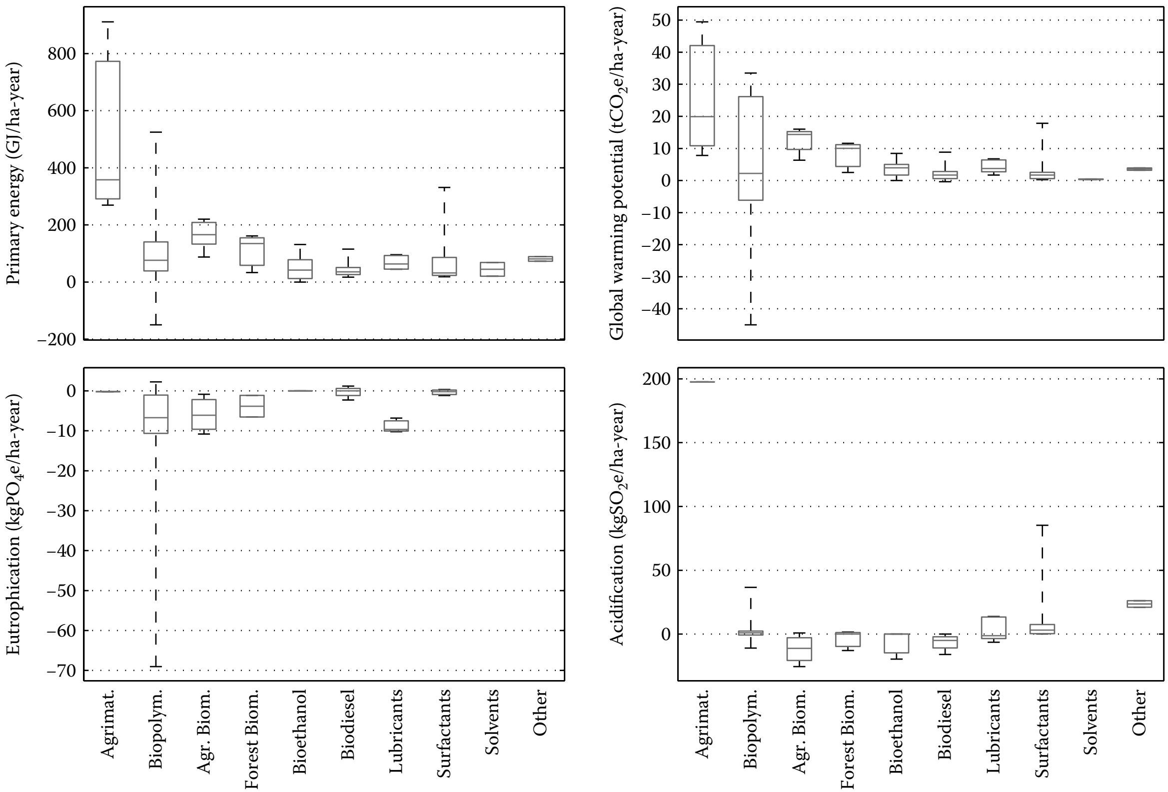


FIGURE 9.2 Advantages and limitations of comparison metrics between plant resource supply chains and their reference fossil resource counterparts. Environmental gain per hectare and year (metric A) due to the use of bio-based products compared with conventional products for (a) nonrenewable primary energy, (b) global warming, (c) eutrophication, (d) acidification impact changes.

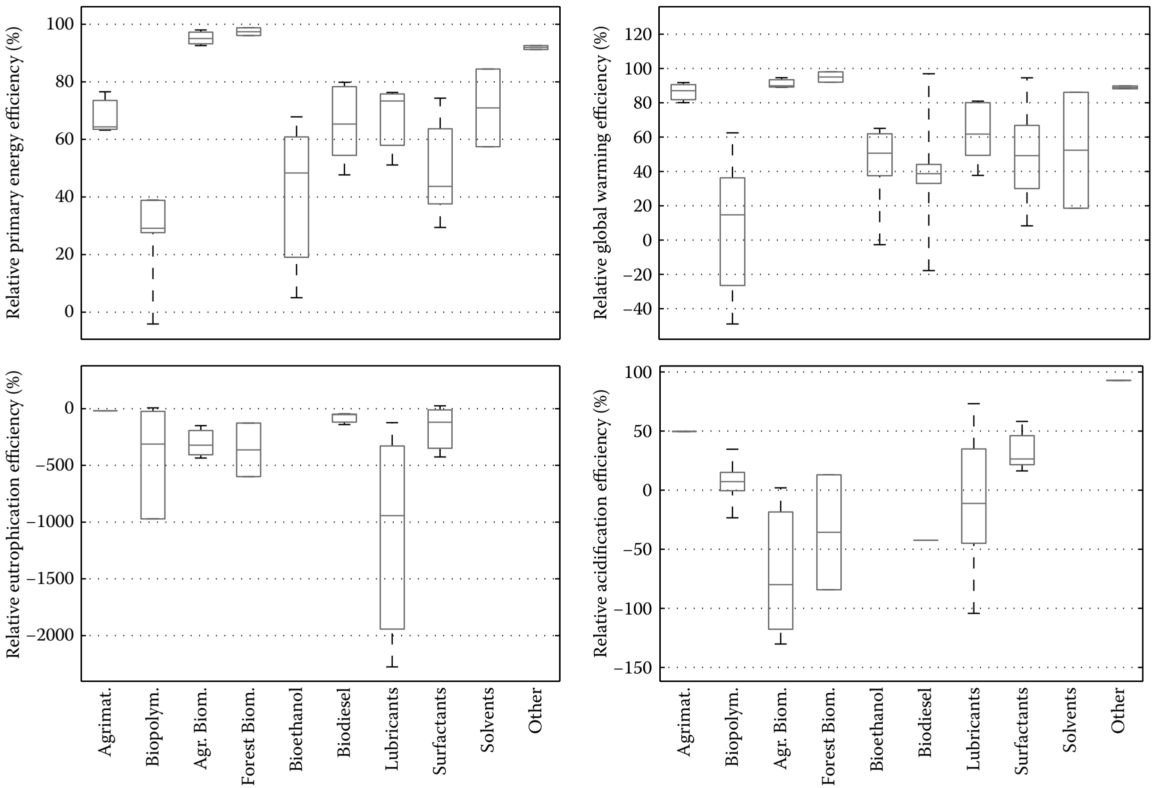


FIGURE 9.3 Relative gain (metric B) due to the bio-based product, relative to the substituted part of the conventional product (with a maximum value of 100%), based on common functional units, for (a) nonrenewable primary energy, (b) global warming, (c) eutrophication, (d) acidification impact changes.

biomaterials, lubricants, and surfactants tend to show environmental benefits, while LCAs for biofuel and energy crops tend to report a higher acidification potential than petrochemical products. For correct interpretation, it is important to note that the availability and reliability of data on primary energy consumption and global warming are significantly greater than eutrophication and acidification, for which data are limited.

In terms of the gain in nonrenewable primary energy and in global warming potential *per cultivated hectare*, the following general tendency can be observed (Figure 9.2a and b): Reported gains may be high (> 300 GJ/ha-year, > 20 $t_{CO2\text{-}eq}$/ha-year) for certain agrimaterials and highly variable—from negative to high for biopolymers. The two biomass categories (agriculture and forest) lead to moderate (100–300 GJ/ha-year, 10–20 $t_{CO2\text{-}eq}$/ha-year) gains per cultivated hectare. These gains are more limited for biofuels and other uses of biomass (< 100 GJ/ha-year, < 10 $t_{CO2\text{-}eq}$/ha-year), with a few individual exceptions. These gains are comparable with those obtained by Dornburg et al. (2004) for bio-based polymers and bioenergy.

For the relative gain in nonrenewable primary energy and in global warming potential, the following general tendency can be observed (Figure 9.3a and b): As was the case with the comparison per cultivated hectare, reported gains may be high for some agrimaterials yet highly variable—from limited to moderate for biopolymers. Agriculture and forest biomass are associated with high relative gains due to limited use of energy in their supply chain. These relative gains are moderate for the biofuels and other uses of biomass.

The reason for these differences and the limitation for these categories is further discussed in the following subsections for each specific group of bioproducts.

9.3.1.2 Agrimaterials

Transportation applications often show strong relative and absolute gains that are possible with agrimaterials, because plant-source products (e.g., pallets, car parts with natural vs. glass fibers) can be much lighter than their conventional fossil counterparts. When a bio-based replacement is used, the energy gain linked to less energy-intensive material can result in an energy gain of up to 200 GJ of primary nonrenewable energy per hectare of fiber crop. In addition, in transportation, the reduced weight of the agrimaterial allows for indirect gain and a corresponding reduction in vehicle fuel consumption during the use stage, with a gain of several hundred gigajoules per hectare of crop. This additional gain is only valid if the change in weight of the considered part is reflected in the final weight of the vehicle.

Across all applications, useful lifetime and end-of-life recovery options are important factors for the agrimaterials sector.

Although interesting results are seen in the agrimaterials supply chain, great potential exists for improvement. This is particularly true in the case of fiber materials, because this is a relatively young technology that has not yet been optimized. Uncertainty is relatively low for certain types of agrimaterials: The supply chain for wood products is fairly well understood in terms of LCA data needs, but fiber sub-supply chains have not been as extensively studied.

9.3.1.3 Biopolymers

The gains achieved from biopolymers are highly variable, because this supply chain is highly variable and depends strongly on the application and the material that is being substituted. Life cycle inventory knowledge of the biopolymer supply chain is moderate (i.e., strong diversity of scenarios, with uncertainties). In addition, end-of-life recovery of biopolymers and the choice of products replaced are important parameters for the supply chain. Biopolymer biodegradability can be an asset or a liability, depending on the end-of-life option chosen for the product. Given the rapid development of the biopolymer supply chain, future environmental gains may be more significant thanks to mass production and improved production technologies.

9.3.1.4 Agricultural and Forest Biomass

The relative gains achieved from bioenergy derived from agricultural and forest biomass are strong (Figure 9.3: from 85% to close to 100%) because their supply chains offer strong relative gains associated with the low energy requirements for product manufacture. The absolute gain, normalized by cultivated area, is moderate. This gain per cultivated area is slightly higher in the case of bioenergy from agricultural biomass because of greater crop yield. However, combustion technology must be improved to lessen the impact of these two supply chains on human health. Although not included in the analysis, particulate emissions from the combustion of biofuels can have significant human health implications and need to be mitigated.

9.3.1.5 Biofuels

Similar positive gains achieved from bioethanol and biodiesel are observed for energy and greenhouse gases in Figures 9.2 and 9.3. However, the relative gain is moderate, and the absolute gain is weak. This trend is because, in the case of biofuels, the upstream supply chain is sometimes longer and more complex than in the case of fossil resources.

Reviewed studies show that the ether alcohol sector is fairly well understood in terms of the cereal (wheat, corn) and sugar (sugar cane and sugar beet) subsupply chains. However, the lignocellulose (wood, straw, grass, etc.) subsupply chains have been far less well studied. Of note is that environmental gains in the lignocellulose subsupply chains appear promising, since this chain can make use of coproducts that are not yet well utilized (e.g., forest waste, pulp and paper, agricultural and municipal coproducts). Therefore, improvement in the environmental performance of ether alcohols may be possible through pairing with other plant supply chains (agricultural biomass, etc.). Because data are contradictory, this gain remains to be validated.

9.3.1.6 Other Uses

Variable positive gains achieved from surfactants, lubricants, solvents, and other chemicals are observed. However, knowledge of the supply chain of those products is poor, so these positive gains should be interpreted with caution. Results show the following trends: (1) the conditions of use, allocation of coproduct emissions, and quantities needed are important parameters for the lubricants supply chain; (2) the impact of fossil solvents can be reduced through the use of other solvents not derived from the plant resources supply chain (e.g., water-based paints instead of paints containing organic solvents); (3) for other chemicals, not enough information is available to permit identification of general trends or trends associated with individual subsupply chains. For the small sample of products studied, the environmental impact in the various impact categories seems to follow trends similar to those of the other plant-based supply chains. As far as the other impact categories are concerned, results are not sufficiently reliable or well documented to permit identification of trends.

9.3.2 Key Parameters and Comparative Metrics

One of the outcomes of this study is the characterization of key factors that contribute to the environmental benefits of different bio-based applications. The parameter most directly associated with bio-based products is biodegradability. However, biodegradability is only an advantage when there is a need and opportunity for it. For instance, chainsaw lubricants are completely lost during use and directly emitted into nature, so biodegradability is an important benefit. Another example is waste bags that are composted together with their contents (Heyde 1998) or biodegradable films used in agriculture that can be left on fields after use and degrade into the soil. Such products save the time and resources normally needed to remove conventional films and avoid the need for the disposal of plastic waste. In most of the LCA studies analyzed, the selected end-of-life options are crucial to the outcomes. Other factors to consider when evaluating and comparing environmental impacts of bio-based products are the type and yield of the biomass, the allocation of environmental impacts to coproducts (e.g., straw), the definition of the FU, the amount of product necessary to fulfill the FU, the technology for the production, and the lifetime of the product.

As shown in the preceding sections, metrics to compare absolute change per hectare of cultivated land (metric A) and change relative to a substituted part (metric B) allow a comparison across studies and constitute an interesting basis for analyzing different supply chains. Table 9.2 summarizes these metrics and shows their relation, as well as inherent advantages and limitations, to comparisons possible for specific products or applications. The quality of the land and the substitutability of the land is also important and deserves further attention. Energy gains per hectare are especially interesting when the considered area cannot easily be used for food production (e.g., forest), avoiding competition between food and bio-based product or bioenergy.

When land is readily available, cost often becomes the main limiting factor. In that case, combining environmental and economic analyses is key, considering the energy substituted per dollar of additional cost. The combined application of environmental LCA and life cycle costing (as discussed in Section 6.8.1) may be used for such analyses.

9.4 CONCLUSIONS

9.4.1 Comparison of Bioproducts

Results show that almost all of the bioproduct LCAs indicate significant positive benefits over conventional products with respect to energy savings and reduced global warming impacts, both per hectare of cultivated land and in terms of impact

TABLE 9.2
Advantages and Limitations of Comparison Metrics between Plant Resource Supply Chains and Their Reference Fossil Resource Counterparts

Level of Comparison	Comparison Metrics	Advantages	Limitations	Results
Meta-analysis	Absolute impacts (fossil − plant) per hectare-year of cultivated biomass	Allows a comparison for all plant and fossil supply chains. Shows optimization of available agricultural areas	Adds further uncertainties (yield, quantity of plant material necessary per functional unit, coproducts)	Absolute value
Meta-analysis	Relative gain (fossil − plant)/fossil product substituted per functional unit	Allows a comparison for all plant and fossil supply chains. Highlights products that use small amounts of nonrenewable energy for product manufacture	Does not show absolute gain	Relative value (relative to fossil-derived product replaced)
Application-specific	Impacts (fossil − plant) per functional unit	Respects system operation	Cannot compare systems that have different functional units	Absolute value
Material/energy	Impacts (plant) and impacts (fossil) per kilogram or per megajoule of useful energy	Allows for easier comparison	May distort the comparison (e.g., if there are different quantities per functional unit)	

per FU. However, acidification and especially eutrophication often show increases in impact. Results point out those products with the highest environmental benefits per hectare of cultivated land: Natural fibers may exceed 300 GJ/ha-year, agricultural and forest biomass achieve between 100 and 200 GJ/ha-year, and biofuel production less than 100 GJ/ha-year. This comparison suggests that, given the limited areas available for biomass production, applications such as bioethanol are of secondary priority when land area is the limiting factor. For all bio-based products, the different stages of development and optimization of production processes must inform the interpretation of results.

Overall, the use of biomass for bio-based materials and as energy crops offers a higher potential for energy savings and greenhouse gas reduction than biofuels or products based on vegetable oils (e.g., lubricants and surfactants). As agricultural and forest biomass require little processing when used for energy, their relative environmental efficiency is very high. Although a similar trend is observed when comparing the environmental impact per hectare of cultivated land, uncertainty increases somewhat due to additional parameters such as yield, biomass content, and data on the environmental impacts of the coproducts and their economic value. Despite these uncertainties, this land-based metric can be used to compare possible uses of limited arable land.

In addition, the comparison of plant and fossil resource supply chains illustrates several important lessons that are generalizable to all the supply chains studied. (1) No one plant resource supply chain stands out above all the others in all impact categories. (2) The replacement of fossil fuels by plant supply chains reduces impacts related to nonrenewable primary energy consumption and the global warming potential, except in the case of bacterial polymers and certain applications that involve the use of other biopolymers. (3) With regard to the eutrophication impact category, plant resource supply chains are often higher in impact than their fossil counterparts, as they may require significant amounts of fertilizer. Plant resource supply chains based on the use of coproducts may have lower eutrophication impacts. (4) Most of the studies reviewed show that plant supply chains that produce chemicals have a weaker acidification impact than fossil fuel chains. Conversely, these same studies indicate that plant supply chains that produce energy have a greater acidification impact than the reference fossil fuel supply chains. (5) The lack of data or poor data reliability, and the variety in units used, prevent a meta-analysis of the supply chains in terms of the following impact categories: destruction of the ozone layer, photochemical pollution, terrestrial and aquatic toxicity, and human health.

9.4.2 Plant Resource Supply Chains

The following recommendations are intended to optimize the benefits from plant resource supply chains, based on environmental considerations alone. However, constraints other than those of an environmental nature may create other priorities.

- Focus on supply chains where the potential energy and global warming benefits are moderate to high (agrimaterials, biopolymers, agricultural biomass for bioenergy, and forest biomass for bioenergy), while taking exceptions into account.

- Emphasize supply chains in which the intrinsic characteristics of materials or energy from plant resources are superior to those of their fossil counterparts (resistance, weight, useful life, biodegradability, quantities required to perform the same function, etc.), and look for both:
  - Direct advantages (low energy requirement for product manufacture, less than or comparable with that of fossil resources, longer useful life for materials, etc.).
  - Indirect advantages (fuel economy resulting from vehicle weight reduction giving the plant resource supply chain a greater than 100% advantage over fossil supply chain).
- Promote synergies among supply chains (through the use of coproducts, etc.).
- Support technological improvement in all supply chains. With the exception of surfactants, plant resource supply chains are much less developed than their fossil counterparts, from both a technology and market share perspective. The gains to be achieved here are manifold. For example, advances in energy conversion technology will significantly impact supply chains for agrimaterials (fibers), biopolymers, forest biomass for bioenergy, and agricultural biomass for bioenergy.

Consistent integration of other factors that currently limit the development of plant resource supply chains, including market potential and economic viability, is also desirable.

9.4.3 Improvement of LCA Knowledge

The following gaps in LCA knowledge are to be addressed in priority order to better evaluate the environmental impacts of plant resource supply chains:

- Broadening LCA knowledge in growth areas (strong potential in limited markets or moderate potential in larger markets or both) and improving the quantification of environmental gains associated with the supply chains in question: collect missing LCA data and update obsolete data. Integration of logistic chains (long-distance transportation) receives little attention in LCA studies and must also be a priority.
- Using LCA approaches in R&D to investigate and invest in the most promising plant supply chains.
- Extending LCA to other limiting factors; for example, to a link to an economic study (or life cycle cost analysis) and to a study of the potential for substitution on a market scale.

9.4.4 Methodological Outlook

Bio-based products and conventional, petrochemical products have associated environmental impacts. This study has not attempted to highlight a product or class of products as "preferable," for any such conclusions would depend on the weighting of the different impact categories within a sustainability framework. Rather, this meta-analysis used two main cross-product and cross-supply chain metrics to indicate that bio-based products, in general, present benefits in terms of nonrenewable energy consumption and global warming impacts relative to conventional products. However, when agricultural production is involved, the eutrophication potential is usually higher for bio-based products than for petrochemical products. These conclusions might not hold when non-bio-based materials are lighter or have better characteristics (e.g., for transport applications). Furthermore, technological advances may have large influences on the production—and hence impacts—of bio-based products, as many of these processes supply chains are still relatively young in comparison with comparable petrochemical systems.

In terms of methodology development, such metastudies provide interesting insights to compare different uses of a given resource or different processing or treatment alternatives (e.g., waste treatment strategies). A thorough analysis of the background hypotheses of each individual study and a selection of the best available studies is key for providing useful insights. Another crucial point is to define a common metric that puts studies on a comparable basis across different types of application, such as the change in impact per hectare of cultivated land between the biomass and fossil scenarios. The present metastudy has focused on data for the period 2000–2008. Since the number of LCAs of food and agriproducts have substantially increased in recent years, an update to this metastudy would be of high interest, whereas the general trends are expected to be robust. Metastudies do not lend themselves to updates or to following the evolution in the environmental performances of a given technology (here, additional data were collected for the biodiesel five years after the initial study). Variations of hypotheses and background data across different LCA studies make it difficult to identify gradual changes in emissions and impacts per FU, unless there are dramatic changes. For this purpose of trend analyses, scenario analysis within one consistent study is likely better suited to follow up incremental changes in the performance of a given technology.

Appendix I

WEBSITES

ADEME—French EPA (Life Cycle Assessment—ACV)	http://www.ademe.fr/expertises/consommer-autrement/passer-a-laction/dossier/lanalyse-cycle-vie/quest-lacv
American Center for Life Cycle Assessment (ACLCA)	http://lcacenter.org/
BEA (U.S. economic matrix)	http://www.bea.gov/
Brightway2	brightwaylca.org
Carnegie Mellon University (EIO-LCA model)	www.eiolca.net/
CEDA (I/O software)	http://cedainformation.net/
CIRAIG (Quebec LCI database)	http://www.ciraig.org/http://www.ciraig.org/fr/bd-icv.php
CML (characterisation factors for the Dutch handbook on LCA)	http://cml.leiden.edu/software/
$DynCO_2$ (dynamic carbon footprinter), CIRAIG	http://www.ciraig.org/en/dynco2.php
E3IOT database	http://www.cml.leiden.edu/software/data-e3iot.html
Eco-indicator (characterisation factors for Eco-indicator 1999)	http://cpmdatabase.cpm.chalmers.se/StartIA.asp
ecoinvent (database, updates …)	http://www.ecoinvent.org/
EDIP (characterisation factors for EDIP 1997)	http://cpmdatabase.cpm.chalmers.se/StartIA.asp
EORA MRIO database	http://worldmrio.com/
EPA (U.S. Environmental Protection Agency)	http://www.epa.gov/nrmrl/std/lca/lca.html
EPS (characterisation factors for EPS 2000)	http://cpmdatabase.cpm.chalmers.se/StartIA.asp
EXIOPIOL (development of European economic matrices)	www.feem-project.net/exiopol/index.php www.exiobase.eu
GTAP (compilation matrices économiques)	https://www.gtap.agecon.purdue.edu/databases/v7/
IMPACT 2002+ (characterisation factors)	http://www.quantis-intl.com/impact2002.php
IMPACT World+	www.impactworldplus.org/
Korea National Cleaner Production Center	http://www.kncpc.or.kr/en/main/main.asp
LCA Digital Commons	https://www.lcacommons.gov/discovery/
Life Cycle Initiative	http://www.lifecycleinitiative.org/
Life Cycle Strategies. Australian Inventory database. Also available for SimaPro users.	http://www.lifecycles.com.au/#!australasian-database/cbm5
Open LCA	http://www.openlca.org/web/guest;jsessionid=E054C17A4DF836D95EF2FD52078A40EC
Personal website of G. Doka; links to LCA-related sites	http://www.doka.ch/lca.htm

Plastics Europe, Association of Plastics Manufacturers	http://www.plasticseurope.org/plastics-sustainability/eco-profiles.aspx
Quantis Suite 2.0 (Quantis software)	http://www.quantis-intl.com/
ReCiPe	http://www.lcia-recipe.net
RMIT	http://www.rmit.edu.au/research/research-institutes-centres-and-groups/research-centres/centre-for-design-and-society/research-areas/life-cycle-assessment/
Social Hotspots Database	http://socialhotspot.org
SETAC, Life Cycle Assessment Global Advisory Group	http://www.setac.org/group/AGLCA
TRACI	http://www.epa.gov/nrmrl/std/traci/traci.html
United Nations Environment Programme, resource efficiency	http://www.unep.org/resourceefficiency/
University of Michigan (iMod laboratory)	https://sph.umich.edu/research-projects/group.cfm?deptID=2&groupID=7
USEtox	http://www.usetox.org
WIOD	http://www.wiod.org/new_site/data.htm
World Food LCA Database	http://www.quantis-intl.com/microsites/wfldb/

Appendix II

MAJOR LCA SOFTWARE

This appendix details the main software currently available for LCA such as SimaPro (Goedkoop et al. 2003), GaBi (GaBi 2003), Quantis Suite® (http://www.quantis-intl.com/en/offer/software-and-it-services/), CMLCA (Heijungs and Frischknecht 2005), openLCA Open-IO (Ciroth et al. 2007), Earthster (Sylvatica 2010), Umberto (IFU Hamburg GmbH 2003), and TEAM (ECOBILAN 2004) (Table A2.1).

SimaPro is appropriate for environmental design of products and for detailed environmental assessment of the contribution of system processes. It also easily allows analysis of the contribution of different pollutants in different impact categories. SimaPro 6 allows the study of the propagation of uncertainties using Monte Carlo analysis/assessment (Section 6.5.2) and to combine the life cycle assessment (LCA) process approach and input–output (I/O) approach (Chapter 4). The ecoinvent database (Section 4.3.2) is available and it is possible to access all of the unit processes of the database.

GaBi has the advantage of introducing nonlinear relationships programmed by the user. It also offers the opportunity to purchase additional data for the automotive and telecommunications sectors. The ecoinvent database is available and presented in an aggregate manner. GaBi is less flexible in terms of interpretation: The determination of the contribution of each pollutant requires a separate worksheet and the tool does not provide details on each unit process that composes the data. English and German versions are currently available, but compatibility is limited at the moment.

Quantis Suite applies the principle of product LCA to a whole enterprise, taking into account the supply and user chain. Developed by Quantis, a spin-off from the Swiss Federal Institute of Technology, Lausanne (EPFL), this tool takes into account the direct and indirect impacts as required by ISO 14000. The consideration of combined costs also allows the calculation of the economic gain derived from a decrease in impact for each life cycle stage. One of the advantages of this approach is to offer a wide circle of companies the opportunity to evaluate their performance throughout the life cycle, taking into account their specific structure.

CMLCA (Chain Management by Life Cycle Assessment) is a software supporting the technical steps of LCA. Although CMLCA does not provide a very flexible user interface, its analytical possibilities are rich (full matrix inversion, integrated methods for sensitivity analysis, and uncertainty analysis). The program also allows the creation of hybrid inventories, consisting of process data and I/O data. However, the complete I/O database is not free.

The OpenLCA project is a modular software for life cycle assessment and sustainability evaluations. The software will be available free of charge as open source. It is an LCA calculator, with a format converter and an uncertainty module. The

TABLE A2.1
Major LCA Software

LCA Software	Key Features	Language	Ecoinvent	Supplier
SimaPro	Environmental design of products Detailed environmental assessment Propagation of uncertainty (Monte Carlo) Combined approach (process and I/O)	French English German Spanish Italian Danish Dutch	Yes	PRé Consultants bv, Plotterweg 12, 3821 BB Amersfoort, the Netherlands
GaBi	Introduction by the user of nonlinear relations Additional databases in automotive and telecommunications	English German	Yes	PE Europe GmbH, Hauptstraße 111–113, 70771 Leinfelden-Echterdingen, Germany IKP University of Stuttgart, Department Life Cycle Assessment, Hauptstraße 113, 70771 Leinfelden-Echterdingen, Germany
Quantis Suite	Assessment of the activity of a company by site, product, or management unit	English French	Yes	Quantis, Parc scientifique EPFL, Bât. A, 1015 Lausanne, Switzerland http://www.quantis-intl.com
CMLCA				http://www.cmlca.eu/
openLCA				http://www.openlca.org/home
Open-IO				Applied Sustainability Center, Business Building 475, University of Arkansas, Fayetteville, AR 72701 www.open-io.org http://www.sustainabilityconsortium.org/open-io/use-the-model/
Earthster				Sylvatica, 22 Trafton Street, York, ME 03909 http://www.earthster.org/
TEAM	Additional processes but source unclear	English English	Can be imported	Ecobilan, 32, rue Guersant, 75017 Paris, France
Umberto	Wider scope: LCA is one possible use	English	Yes	Institut für Umweltinformatik Hamburg GmbH, Grosse Bergstrasse 219, 22767 Hamburg, Germany

Open-IO (www.open-io.org) has already produced an I/O database specific to the United States.

Earthster aims to provide to all businesses the means to conduct evaluations of and publications on the life cycle, in order to document and publish their environmental and social performance. This “open source” software is available on the Internet free of charge. Producers have the possibility of downloading and using the free software to quickly assess their performances and compare them with the industry averages.

The scope of Umberto is broader than the other tools, LCA representing one of the possibilities offered by the software.

TEAM offers some processes which are not available in other databases but it does not always provide a clear description of the sources and unit processes at the origin of the data.

Appendix III

LCI DATA: ECOINVENT

This appendix presents extracts from the ecoinvent 2.2 database. The comprehensive database provides more than 650 emission and extraction factors (1200 when including subcompartments such as low and high population densities) for close to 4000 processes, covering

- Energy supply, including all of the electric mixes (coal, gas, cogeneration, nuclear, wind, etc.) and energy carriers (extra light oil, fuel oil, kerosene, steam, coal, high- and low-pressure natural gas, heat pump, etc.), for a wide range of countries in Europe and worldwide
- Materials and construction processes (bricks, glass packaging, primary and secondary aluminum, lead, nickel, stainless steel, all the most common plastics [polypropylene, polystyrene, etc.], wood construction materials, etc.)
- Chemicals (oxygen, nitrogen, etc.)
- Detergents
- Papers (graphics, recycled, etc.)
- Waste treatment services (household waste, sewage sludge, plastics, solvents, etc.)
- Most common agricultural products and processes (potato, sugarbeets, etc.)
- Transportation (trucks, cargo, trans- and transoceanic freight, air, short- and long-distance passenger airplanes, trams, buses, short- and long-distance trains, etc.)
- Computers, printers, and related accessories

Table A3.1 provides the values of nonrenewable primary energy, air emissions of fossil CO_2, the ratio of CO_2 emissions/nonrenewable primary energy, the $CO_{2\text{-eq}}$ (100 and 500 years) emissions and scores for four damage categories of IMPACT 2002+ for 50 process values. The full database also provides impact scores calculated by other impact assessment methods (IMPACT 2002+, ReCiPe, TRACI, etc.).

TABLE A3.1
Extracts from the ecoinvent© 2.2 Database

Process	Location	Unit	Total Fossil + Nuclear (MJ-eq)	CO_2 (kg)	Ratio g CO_2/MJ (g CO_2/MJ)	$CO_{2\text{-eq}}$ 100 Years (kg)	$CO_{2\text{-eq}}$ 500 Years (kg)	Climate Change (Point)	Ecosystem Quality (Point)	Human Health (Point)	Resources (Point)
Agriculture											
Wheat grains integrated production, at farm	Switzerland	kg	3.3	0.18	55	0.59	0.4	3.99×10^{-5}	1.16×10^{-4}	6.03×10^{-5}	2.22×10^{-5}
Ammonium nitrate, as N, at regional storehouse	Europe	kg	60.3	2.89	48	8.65	5.91	5.93×10^{-4}	5.14×10^{-5}	4.90×10^{-4}	4.10×10^{-4}
Pesticide, unspecified, at regional storehouse	Europe	kg	216.4	7.29	34	7.73	7.47	7.49×10^{-4}	1.38×10^{-4}	7.85×10^{-4}	1.45×10^{-3}
Triple superphosphate, as P_2O_5, at regional storehouse	Europe	kg	33.19	1.98	60	2.08	2.03	2.03×10^{-4}	2.76×10^{-4}	4.89×10^{-4}	2.23×10^{-4}
Equipment											
Building, multistorey	Europe	m^3	7998.27	489	61	546	530	5.28×10^{-2}	2.32×10^{-2}	9.19×10^{-2}	5.50×10^{-2}
Tractor, production	Switzerland	kg	126.17	5.46	43	5.96	5.79	5.75×10^{-4}	1.56×10^{-4}	6.00×10^{-4}	8.47×10^{-4}
Material											
Steel, low alloyed, at plant	Europe	kg	27.48	1.63	59	1.8	1.72	1.72×10^{-4}	7.83×10^{-5}	3.26×10^{-4}	1.92×10^{-4}
Aluminium, primary, at plant	Europe	kg	160.35	9.55	60	11.8	12.48	1.25×10^{-3}	2.28×10^{-4}	1.34×10^{-3}	1.08×10^{-3}
Aluminum, secondary, from old scrap, at plant	Europe	kg	22.36	1.32	59	1.4	1.35	1.35×10^{-4}	1.21×10^{-4}	1.26×10^{-4}	1.53×10^{-4}
Aluminum, production mix, at plant	Europe	kg	112.96	6.71	59	8.26	8.71	8.75×10^{-4}	1.72×10^{-4}	9.38×10^{-4}	7.63×10^{-4}
Concrete, normal, at plant	Switzerland	m^3	1381.58	257	186	263	259	2.60×10^{-2}	1.66×10^{-3}	1.13×10^{-2}	9.14×10^{-3}
Glued laminated timber, indoor use, at plant	Europe	m^3	3949.83	192	49	209	202	2.00×10^{-2}	3.70×10^{-2}	4.48×10^{-2}	2.64×10^{-2}
Cement, unspecified, at plant	Switzerland	kg	3.38	0.75	222	0.76	0.75	7.55×10^{-5}	2.91×10^{-6}	2.62×10^{-5}	2.23×10^{-5}
Copper, at regional storage	Europe	kg	31.23	1.86	60	2.04	1.96	1.95×10^{-4}	1.42×10^{-3}	3.20×10^{-3}	3.81×10^{-4}

Acrylic dispersion, 65% in H_2O, at plant	Europe	kg	52.28	1.99	38	2.16	2.06	2.06×10^{-4}	3.36×10^{-5}	1.75×10^{-4}	3.51×10^{-4}
Tap water, at user	Europe	kg	0.00555	0.000298	54	0.000318	0.000307	3.08×10^{-8}	2.33×10^{-8}	2.94×10^{-8}	3.69×10^{-8}
Glass fiber, at plant	Europe	kg	44.74	2.41	54	2.64	2.51	2.52×10^{-4}	4.06×10^{-5}	3.72×10^{-4}	3.04×10^{-4}
Glass wool mat, at plant	Switzerland	kg	45.06	1.38	31	1.49	1.43	1.43×10^{-4}	2.96×10^{-5}	1.27×10^{-4}	3.03×10^{-4}
Paper, newsprint, at regional storage	Europe	kg	24.33	1.22	50	1.3	1.26	1.26×10^{-4}	4.50×10^{-5}	1.46×10^{-4}	1.62×10^{-4}
Kraft paper, bleached, at plant	Europe	kg	30.12	1.59	53	1.7	1.63	1.64×10^{-4}	1.12×10^{-4}	2.26×10^{-4}	2.03×10^{-4}
Paper, recycling, no deinking, at plant	Europe	kg	13.12	0.78	59	0.83	0.8	8.02×10^{-5}	1.65×10^{-5}	3.43×10^{-5}	9.01×10^{-5}
Glass-fiber-reinforced plastic, polyamide, injection molding, at plant	Europe	kg	144.66	7.66	53	8.72	8.05	8.08×10^{-4}	2.61×10^{-5}	4.75×10^{-4}	9.75×10^{-4}
Polyethylene, HDPE, granulate, at plant	Europe	kg	76.35	1.56	20	1.93	1.7	1.69×10^{-4}	1.76×10^{-6}	9.59×10^{-5}	5.12×10^{-4}
Polystyrene foam slab, at plant	Europe	kg	104.94	3.33	32	4.18	3.64	3.59×10^{-4}	1.41×10^{-5}	2.06×10^{-4}	7.04×10^{-4}
Polyvinylchloride, at regional storage	Europe	kg	59.4	1.83	31	1.97	1.89	1.89×10^{-4}	3.80×10^{-6}	8.52×10^{-4}	3.98×10^{-4}
Solvents, organic, unspecified, at plant	Global	kg	64.64	1.74	27	2.3	1.97	1.97×10^{-4}	1.02×10^{-5}	1.48×10^{-4}	4.35×10^{-4}
Packaging glass, green, at regional storage	Switzerland	kg	11.49	0.63	55	0.67	0.65	6.50×10^{-5}	9.82×10^{-6}	7.35×10^{-5}	7.79×10^{-5}
Flat glass, uncoated, at plant	Europe	kg	12.58	0.52	41	0.55	0.53	5.33×10^{-5}	1.19×10^{-5}	1.23×10^{-4}	8.51×10^{-5}
Disposal											
Disposal, steel, 0% water, to inert material landfill	Switzerland	kg	0.197	6.57×10^{-3}	33	7.13×10^{-3}	6.81×10^{-3}	6.79×10^{-7}	2.02×10^{-7}	1.32×10^{-6}	1.30×10^{-6}
Disposal, aluminum, 0% water, to sanitary landfill	Switzerland	kg	0.521	2.01×10^{-2}	39	2.15×10^{-2}	2.07×10^{-2}	2.07×10^{-6}	2.35×10^{-5}	6.50×10^{-6}	3.46×10^{-6}
Disposal, packaging paper, 13.7% water, to municipal incineration	Switzerland	kg	0.29	1.85×10^{-2}	64	2.49×10^{-2}	2.18×10^{-2}	2.17×10^{-6}	1.93×10^{-6}	4.59×10^{-5}	1.96×10^{-6}

(*Continued*)

TABLE A3.1 (Continued)
Extracts from the ecoinvent© 2.2 Database

Process	Location	Unit	Total Fossil + Nuclear (MJ-eq)	CO_2 (kg)	Ratio g CO_2/MJ (g CO_2/MJ)	$CO_{2\text{-eq}}$ 100 Years (kg)	$CO_{2\text{-eq}}$ 500 Years (kg)	Climate Change (Point)	Ecosystem Quality (Point)	Human Health (Point)	Resources (Point)
Disposal, municipal solid waste, 22.9% water, to sanitary landfill	Switzerland	kg	0.4	1.90×10^{-2}	48	5.17×10^{-1}	1.71×10^{-1}	1.72×10^{-5}	6.69×10^{-7}	3.46×10^{-6}	2.65×10^{-6}
Disposal, polypropylene, 15.9% water, to municipal incineration	Switzerland	kg	0.209	2.53×10	12060	2.54×10	2.53×10	2.55×10^{-4}	3.71×10^{-7}	4.15×10^{-5}	1.41×10^{-6}
Disposal, plastics, mixture, 15.3% water, to municipal incineration	Switzerland	kg	0.69	2.34×10	3389	2.35×10	2.34×10	2.36×10^{-4}	1.05×10^{-6}	4.97×10^{-5}	4.62×10^{-6}
Fuel Supply, without Combustion											
Diesel, low sulfur, at regional storage	Europe	kg	54.65	4.73×10^{-1}	9	5.25×10^{-1}	4.94×10^{-1}	4.91×10^{-5}	2.07×10^{-5}	7.95×10^{-5}	3.60×10^{-4}
Petrol, unleaded, at regional storage	Europe	kg	57.16	6.49×10^{-1}	11	7.09×10^{-1}	6.74×10^{-1}	6.71×10^{-5}	2.18×10^{-5}	1.00×10^{-4}	3.77×10^{-4}
Fuel Supply, Final Energy with Combustion											
Wood chips, burned in cogen 6400 kWth, emission control	Switzerland	MJ_{final}	0.04	2.00×10^{-3}	50	8.66×10^{-3}	5.52×10^{-3}	5.51×10^{-7}	2.55×10^{-6}	3.36×10^{-6}	2.48×10^{-7}
Pellets, mixed, burned in furnace 15 kW	Switzerland	MJ_{final}	0.24	1.10×10^{-2}	46	1.25×10^{-2}	1.18×10^{-2}	1.18×10^{-6}	2.66×10^{-6}	6.39×10^{-6}	1.60×10^{-6}
Natural gas, burned in boiler, condensing modulating <100 kW	Switzerland	MJ_{final}	1.24	6.56×10^{-2}	53	7.29×10^{-2}	6.80×10^{-2}	6.81×10^{-6}	1.56×10^{-7}	9.72×10^{-7}	8.58×10^{-6}
Light fuel oil, burned in boiler 100 kW, nonmodulating	Switzerland	MJ_{final}	1.3	8.62×10^{-2}	66	8.86×10^{-2}	8.72×10^{-2}	8.74×10^{-6}	4.69×10^{-7}	2.62×10^{-6}	8.60×10^{-6}

Heat: Useful Energy, Useful Energy with Combustion											
Heat, light fuel oil, at boiler 10 kW, nonmodulating	Switzerland	MJ_{useful}	1.41	9.21×10^{-2}	65	9.46×10^{-2}	9.31×10^{-2}	9.34×10^{-6}	5.49×10^{-7}	2.90×10^{-6}	9.29×10^{-6}
Heat, at cogen 160 kWe Jakobsberg, allocation exergy	Switzerland	MJ_{useful}	0.6	3.16×10^{-2}	53	3.58×10^{-2}	3.31×10^{-2}	3.32×10^{-6}	5.89×10^{-8}	4.77×10^{-7}	4.13×10^{-6}
Heat, wood pellets, at furnace 15 kW	Switzerland	MJ_{useful}	0.29	1.34×10^{-2}	46	1.53×10^{-2}	1.44×10^{-2}	1.44×10^{-6}	3.25×10^{-6}	7.79×10^{-6}	1.95×10^{-6}
Electricity											
Electricity, at cogen 160 kWe Jakobsberg, allocation exergy	Switzerland	kWh	12.31	6.52×10^{-1}	53	7.37×10^{-1}	6.81×10^{-1}	6.83×10^{-5}	1.19×10^{-6}	9.60×10^{-6}	8.52×10^{-5}
Electricity, hydropower, at reservoir power plant, nonalpine regions	Europe	kWh	0.05	4.19×10^{-3}	84	1.10×10^{-2}	6.29×10^{-3}	6.31×10^{-7}	6.79×10^{-8}	5.37×10^{-7}	3.12×10^{-7}
Electricity, production mix photovoltaic, at plant	Switzerland	kWh	1.21	6.35×10^{-2}	52	7.35×10^{-2}	7.21×10^{-2}	7.22×10^{-6}	1.97×10^{-6}	8.04×10^{-6}	8.23×10^{-6}
Electricity, production mix UCTE	Europe	kWh	10.47	4.91×10^{-1}	47	5.15×10^{-1}	4.99×10^{-1}	5.02×10^{-5}	8.30×10^{-6}	4.12×10^{-5}	6.95×10^{-5}
Electricity mix	US	kWh	12.11	7.11×10^{-1}	59	7.49×10^{-1}	7.25×10^{-1}	7.28×10^{-5}	1.33×10^{-5}	6.21×10^{-5}	8.04×10^{-5}
Electricity mix	Switzerland	kWh	7.91	1.05×10^{-1}	13	1.12×10^{-1}	1.07×10^{-1}	1.08×10^{-5}	5.31×10^{-6}	1.29×10^{-5}	5.22×10^{-5}
Transport of Goods											
Transport, lorry 16–32 t, EURO4	Europe	tkm	2.55	1.46×10^{-1}	57	1.53×10^{-1}	1.49×10^{-1}	1.49×10^{-5}	4.70×10^{-6}	1.47×10^{-5}	1.68×10^{-5}
Transport, freight, rail	Europe	tkm	0.71	3.75×10^{-2}	53	3.95×10^{-2}	3.85×10^{-2}	3.86×10^{-6}	7.54×10^{-7}	4.84×10^{-6}	4.73×10^{-6}
Transport, transoceanic tanker	Ocean	tkm	0.09	5.47×10^{-3}	61	5.64×10^{-3}	5.55×10^{-3}	5.57×10^{-7}	7.63×10^{-8}	1.49×10^{-6}	5.88×10^{-7}
Transport of Persons											
Transport, ICE (InterCity Express)	Germany	pkm	0.98	5.64×10^{-2}	58	6.01×10^{-2}	5.79×10^{-2}	5.82×10^{-6}	6.90×10^{-7}	2.47×10^{-6}	6.52×10^{-6}
Transport, passenger car	Europe	pkm	3.01	1.72×10^{-1}	57	1.83×10^{-1}	1.78×10^{-1}	1.78×10^{-5}	2.91×10^{-6}	1.34×10^{-5}	1.99×10^{-5}
Computers											
Laptop computer, at plant	Global	unit	2781.03	168.96	61	251.09	272.12	2.73×10^{-2}	7.57×10^{-3}	2.50×10^{-2}	1.88×10^{-2}

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Appendix IV

LCA FORMS

This appendix describes the LCA Tables A4.1 through A4.8 forms used to define the objectives and the system, perform the first calculations by hand and quickly check some points of the LCA.

A4.1 GOAL AND SCOPE DEFINITION

TABLE A4.1
Description of the Study

General objective (information or product development, strategy, policy, regulation)
Target audience (internal/consumer/government/nongovernmental organization, etc.)
Practitioner and stakeholders (sponsor, LCA practitioner, steering committee, peer reviewer, stakeholders, etc.)

TABLE A4.2
Product/System Function and Description of Scenarios

Products	Primary Function	Secondary Functions
Scenario 1		
Scenario 2		
Scenario 3		

TABLE A4.3
Production and Functional Unit

Product or System	Functional Unit (Service Offered)[a]	Reference Flows (What Is Purchased per Functional Unit)[b]	Key Parameters Linking Reference Flows to Functional Unit
Scenario 1			
Scenario 2			
Scenario 3			

[a] Provide the number of functional units considered (e.g., 1 person-km, 1000 persons-km).
[b] Provide the amount or fraction of each reference flow to be purchased per functional unit (e.g., 1/200,000 or 1 car per person-km, 4 L gasoline per person-km).

A4.2 INVENTORY

TABLE A4.4

Flowchart

Start from the Functional Unit and the Reference Flows and Add or Rename the Relevant Unit Processes and Intermediary Flows Until They Can Be Linked to a Database.

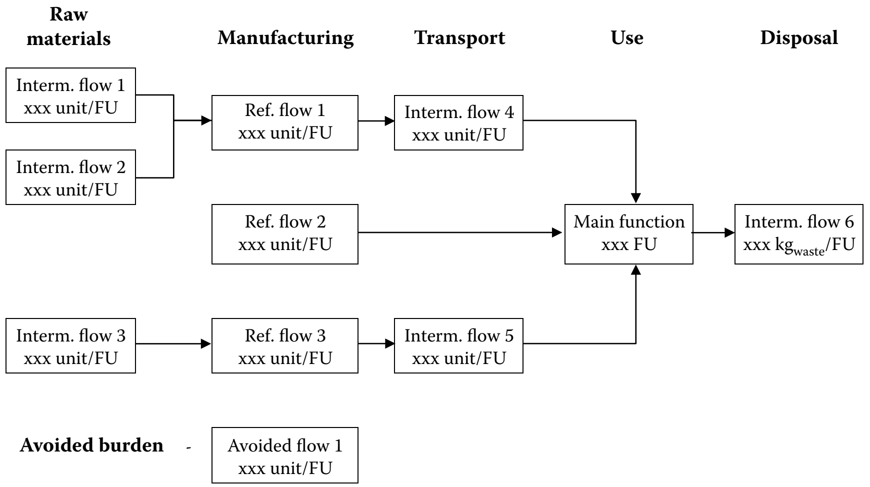

TABLE A4.5
Energy Balance (Hand Calculation)

Life Cycle Stages	Intermediary Flows (unit/FU)	Nonrenewable Primary Energy per Unit (MJ/unit)	Nonrenewable Primary Energy per FU (MJ/FU)
Raw material			
Manufacturing			
Use			
Transport			
Packaging			
Waste			
Avoided burden			
Total			

TABLE A4.6
CO_2 Balance (Hand Calculation)

Life Cycle Stages	Intermediary Flow (unit/FU)	Emitted CO_2 per Unit (g CO_2/unit)	Emitted CO_2 per FU (g CO_2/FU)	Ratio Check (g CO_2/MJ)
Raw material				
Manufacturing				
Use				
Transport				
Packaging				
Waste				
Avoided burden				
Total				

TABLE A4.7

Classification of The Different Types of Product

Position your Product in this Classification Scheme. Lifetime and Amount of Material are Key for Passive Products, Efficiency for Active Products, and Weight for Mobile Products.

	Fixed	Mobile (Transport Dominant)
Passive		
Active (use stage dominant)		

A4.3 LIFE CYCLE COSTING (LCC, OFF LCA)

TABLE A4.8
Calculation of Life Cycle Costs

Product or System	Functional Unit (Service Offered)	Reference Flows (What Is Purchased)	Costs (off LCA)[a]
Scenario 1			
Scenario 2			
Scenario 3			

[a] Remember that the costs are not included in the environmental life cycle assessment.

Glossary

allocation: Attribution of some environmental emissions and resource use among the product studied and the other coproducts; used in the case of multiproduct systems.

anaerobic digestion: Waste treatment process during which organic matter reacts to produce biogas without oxygen.

aquatic acidification: Phenomenon corresponding to an increase of the concentration of protons (H^+) in the water, which causes a decrease in pH. These additional protons come primarily from nitric acid (HNO_3) or sulfuric acid (H_2SO_4), derived from gases such as NO_x and SO_2. Freshwater acidification also results in the dissolution of some toxic metals such as aluminum.

aquatic ecotoxicity: Toxicity with respect to living aquatic organisms, excluding human beings.

aquatic eutrophication: Excessive enrichment of an aquatic environment by nutrients (e.g., nitrogen, phosphorus) and organic matter, especially strong if the water is stagnant or its circulation is reduced. This causes overabundant development of plant biomass, whose subsequent decomposition consumes, partially or entirely, dissolved oxygen in water and reduces the aquatic environment biodiversity.

bioconcentration: Phenomenon by which living beings absorb substances naturally present in their habitat, which accumulate in their bodies to sometimes higher concentrations than those at which they occur in the natural environment.

bioconcentration factor: For a given substance, the ratio of the concentration in the body to the concentration in water.

bonus: See "System expansion."

by-product: Secondary/side product generated during the production process. This by-product has an economic value, but does not match the studied function or is used outside the system being studied.

chlorofluorocarbons: Chemicals consisting of carbon, fluorine, and chlorine, known by their commercial name Freon, partly responsible for the destruction of the stratospheric ozone layer.

classification: Step of the environmental life cycle impact assessment in which a set of midpoint environmental impact categories are defined; emissions and extractions are then classified into any relevant midpoint categories.

climate change: Global phenomenon of climate equilibrium modification due to increased concentrations of greenhouse gases in the atmosphere.

cost–benefit analysis: Analysis to determine the investment that yields the largest environmental improvement, to promote the scenario where environmental improvement margin is the largest per unit of investment.

critical review: Critical study of a life cycle assessment aimed at ensuring its quality by checking that the assumptions, methods, and data are consistent with the objectives of the study and that the results are comprehensive enough to support a conclusion based on the objectives listed in the goal and scope definition phase.

damage (end point) category: Category addressing the damage to different areas of protection (such as human health or ecosystems), represented by a damage indicator.

damage (end point) characterization: Step of the environmental life cycle impact assessment, evaluating the contribution of midpoint categories to one or more damage categories corresponding to different areas of protection.

damage (end point) characterization factor: Estimates the damage to the area of protection *d* caused per unit of the midpoint reference substance of category *m*.

damage (end point) impact score: Sum of each damage category of the damages caused per unit of the midpoint reference substances multiplied by the midpoint impact scores.

dematerialization: One of the ecodesign principles, aimed at designing services rather than products.

distance to target: Method defining weighting factors in environmental impact assessment. Links the weighting factors with political, administrative, or environmental objectives.

dryness: Ratio of dry mass to total mass.

ecodesign: Integrates environmental aspects into the design or redesign of products.

ecolabeling: Identification of products which, based on their production and disposal, have a minimal impact on the natural environment.

ecosystem: Dynamic set of living organisms (plants, animals, and microorganisms) that interact with each other and with the environment (soil, climate, water, light) in which they live.

ecotoxicity: Toxicity with respect to living organisms, excluding human beings.

effect concentration 50: Concentration of a substance for which 50% of individuals of a given species are affected in terms of mobility, reproduction, or mortality. This value is obtained by ecotoxicological tests on living organisms.

effect factor: Factor characterizing the potential risks and severity of each risk.

electricity mix: Electricity mix from different sources (fossil, nuclear, hydraulic, etc.) and different technologies.

elementary flow: Flow linking a unit process to the environment; input elementary flows correspond to the use of natural resources, and elementary flows exiting a unit process are emissions to air, water, or soil.

end of pipe: Approach aimed at decreasing pollution by the implementation of technologies for treatment of waste and anything else emitted to the environment.

energy carrier: Element carrying energy, such as fuel.

environmental audit: Assessment of the environmental performance of an individual business, including follow-up suggestions.

environmental efficiency: Ratio of impact reduction to cost increase.

environmental impact assessment: Study focusing on predicting the impact of a planned installation at a precise location.

environmental system management: Tool for the management of a company to reduce and compensate for its environmental impacts.

exposure factor: Equivalent fraction of the medium *n* (air, water, soil, or food) ingested daily by the general population (inhaled or ingested orally).

fate factor: Factor characterizing the transport and diffusion of pollutants in the environment.

final energy: Energy provided and purchased by user.

financial allocation: Allocation based on economic causality, assuming that a product is primarily made for its mercantile value. Emissions and resource use are allocated among coproducts according to their respective economic values.

flowchart/flow diagram: Diagram of the processes required for a certain product or function and the relationships of these processes; depicts each unit process considered within the system and quantifies the intermediary flows linking these unit processes.

fossil carbon dioxide: Carbon dioxide emitted during combustion or degradation of petroleum products.

functional boundaries: System boundaries describing the same functional reality in the various scenarios studied.

functional unit: Quantifies the function of a system in terms of the service offered, and is the same for all scenarios compared in an LCA.

geographical correlation: Estimation of the difference between the area defined in the study and the area of which the data is representative.

global warming potential: Conversion factor characterizing the contribution of each greenhouse gas to the change in radiative forcing, thus representing the relative greenhouse contribution of each gas relative to CO_2.

goal and scope definition: First phase of the life cycle assessment, describing the study, its objectives and scope; analyzing the function of the system studied; defining the functional unit; and specifying the boundaries of the system and its limitations.

greenhouse effect: Natural phenomenon of atmospheric temperature increase, due to certain "greenhouse" gases absorbing and reemitting infrared radiation emitted by the earth. The international scientific consensus is that the anthropogenic release of greenhouse gases that are normally stored in the earth increases this greenhouse effect, and thus increases climate change.

greenhouse gases: Gases that absorb and reflect terrestrial radiation. The anthropogenic increase in atmospheric concentration of some of them (carbon dioxide, methane, nitrous oxide, HCF) is causing climate change.

grouping: Qualitative or semiquantitative process that helps prioritize results by sorting or ranking.

higher heating value: Amount of energy that would be released by the complete combustion of a unit mass or volume of fuel (gas, oil, coal, etc.), assuming the water formed during combustion is returned to a liquid state and the other products are in a gaseous state.

human exposure: Concentration or amount of a substance that reaches the human being.

human toxicity (carcinogenic): Midpoint impact category representing the carcinogenic effects of substances on human beings.

human toxicity (noncarcinogenic): Midpoint impact category representing the noncarcinogenic effects of substances on human beings.

hydrofluorocarbons: Fluorinated gases, replacing chlorofluorocarbons (CFCs), which have a significant greenhouse potential.

hydrogen to carbon ratio: Ratio of hydrogen to carbon for fuel molecules. The larger the hydrogen to carbon ratio (H/C ratio) of combustible molecules, the less CO_2 will be produced.

impact pathway: Encompasses all the environmental processes from the substance emission to its final impact.

industrial ecology: Interdisciplinary science aimed at optimizing the use of energy, resources, and capital of a technological system by minimizing its environmental impacts. The technological system in this case is defined as a living system that interacts with natural systems.

input: Matter or energy flow entering the system.

input–output method: Method to calculate the inventory of emissions and extractions based on the economic flows generated by the product or service rather than on the basis of physical flows.

intake fraction: Fraction of a pollutant emission to the environment that ends up ingested by the population.

intermediate product flow: Flow linking one unit process to another unit process, expressing the quantity of each unit process needed for the subsequent unit process.

internalization of external costs: Expression of environmental impacts in financial terms, to combine them directly with economic costs.

International standard ISO 14000 : Series of norms produced by the International Organization for Standardization on environmental management systems for businesses to manage the environmental impact of their activities and to measure their environmental performance.

interpretation: Fourth phase of the life cycle assessment; identifies the life cycle stages at which intervention can substantially reduce the environmental impacts of the system or product and analyzes the uncertainties involved.

inventory of elementary flows/inventory of emissions and extractions: Quantitative description of flows of matter, energy, and pollutants crossing the system boundary. This includes the emissions of polluting substances to the environment, as well as the amounts of extracted resources from the environment (minerals, energy carriers, soil surface area, etc.) throughout the life cycle of the analyzed product or service.

ionizing radiation: Very high energy radiation capable of ionizing substances through which it passes, which can cause genetic mutation, cancer, and other negative outcomes. It often originates in radioactive substances with unstable nuclei, which emit ionizing radiation during the decay process.

key parameter: Parameter linking reference flows to the functional unit. It often measures environmental performance as a ratio of material needed per function, whereas the functional unit itself is additive and not a ratio.

life cycle assessment: Tool assessing and comparing the environmental impacts of products and services related to their function.

life cycle costing: Analysis of the financial flows of a product or a service over its life cycle.

life cycle impact assessment: Third phase of an LCA, linking data on raw material extractions and substance emissions associated with a product's life cycle with their environmental impacts. It consists of three steps: classification of emissions in different impact categories, characterization of the midpoint impacts, and damage (end point) characterization.

life cycle impact assessment method: Method modeling the impact pathways of substances to link, as accurately as possible, each inventory flow to its potential environmental damage.

Life Cycle Initiative: Launched by UNEP and SETAC, initiative aimed at developing and disseminating practical tools for evaluating solutions, risks, advantages, and disadvantages associated with products and services throughout their life cycle.

life cycle inventory: Second phase of the life cycle assessment, quantifying the different flows through the system.

life cycle management: Integrated approach aimed at minimizing environmental burdens associated with a product or service throughout its life cycle. Applied to company management, it aims to integrate environmental aspects into industrial processes by considering the impacts and costs of the supply chain.

life cycle stage: One of the following steps: resource extraction and preparation, provision of infrastructure and inputs, transportation, manufacturing, use and maintenance of products, disposal and recycling of waste.

life cycle thinking: Approach taking into account all life cycle stages of products and services in management decisions. It applies to environmental, economic, and social decisions and includes other concepts such as industrial ecology, risk analysis, ecolabels, environmental management systems, ecodesign, and life cycle management.

lower heating value: Effective heat released during combustion, determined by subtracting the heat of vaporization of the water vapor from the higher heating value.

marginal variations: Allocation method applicable when we can vary at will the ratio of coproducts in a way that corresponds to actual practice.

midpoint category: Category grouping the inventory results having similar effects (e.g., all the substance emissions that contribute to the greenhouse effect).

midpoint characterization: Step of the environmental life cycle impact assessment, weighting emissions and extractions to represent their contribution to each midpoint impact category.

midpoint characterization factor: Expresses the relative importance of substance emissions or extractions in the context of a specific midpoint environmental impact category.

midpoint impact score: Sum of each midpoint category of the masses emitted or extracted multiplied by the midpoint characterization factors.

module: Schematic representation of a process, used for system modeling.

Monte Carlo analysis: Data-intensive statistical analysis estimating the uncertainty of the final results of a model due to input parameter uncertainties; also determines the significance of a difference between two scenarios.

multifunctional product: Product with multiple functions.

nonrenewable primary energy: Energy contained in the energy carriers at the point of extraction from the environment that is either irreplaceable or replaced very slowly through natural processes.

normalization: Optional step of the environmental life cycle impact assessment, expressing a given impact per functional unit relative to the total impact in that category to better understand the magnitude of the damage. It thus compares the respective contribution of the considered product or service to the current total effect on a global, continental, or regional level for a given category (midpoint or damage).

normalization value: Reference value to which the results of the impact characterization results are compared for normalization.

normalized score: Damage score reported relative to a normalization value, thus giving the respective contribution of the product or service considered to the current total effect on a global, continental, or regional level for a given category.

octanol-water partition coefficient: Ratio between the equilibrium/steady-state concentration of a chemical substance in octanol and the concentration of that substance in water; used to estimate the bioconcentration factor indirectly.

output: Matter and energy flows leaving the system.

ozone layer depletion: Destruction of ozone in the stratosphere due to certain molecules, such as CFCs. The stratospheric ozone layer is crucial to terrestrial life because it absorbs harmful ultraviolet radiation.

peer reviewer: Normally, an independent expert performing the critical review.

photochemical oxidation: Impact category related to the formation of ozone in the troposphere (lower atmosphere) from volatile organic compounds and NO_x. This ozone is a strong oxidant, causing respiratory problems and limiting plant growth.

physical allocation: Allocation based on physical causality, namely, a property or parameter representative of the causality relationship between production and emissions. This method applies if there is a direct causal relationship between the physical parameter and the amount of emissions or resource used (i.e., emissions proportional to the physical quantity considered).

process subdivision: Detailed description of a system that results in multiple products, examining if certain subprocesses may only be relevant to one of the coproducts.

product flow: Flow linking a unit process to the economy.

reference flow: For a given functional unit, amounts of goods or services purchased to fulfill the function and generate this functional unit.

reference substance: Substance on which impact scores are based within a given impact category.

reliability: Data quality indicator, based on the measurement method and the verification procedures.

respiratory effect (inorganics): Respiratory diseases (asthma, chronic bronchitis, etc.) due to inorganic substances such as particulates, SO_2, and NOx.

risk assessment: Studies the risk or probability of severe impacts occurring from an installation (such as a nuclear power plant) or the risks of using a chemical substance.

robustness: Insensitivity to small differences in the assumptions. The robustness generally reflects the resistance of the estimate to aberrant data/outliers.

scenarios: Alternatives studied that are compared with one another during a life cycle assessment.

screening: Quick and simple analysis, evaluating the order of magnitude of each life cycle stage contribution.

sensitivity analysis: Tests the robustness of the results and their sensitivity to data, assumptions, and models used.

substance flow analysis/mass balance analysis: Quantifies the flow and accumulation in the environment of either a single substance, such as mercury, or a group of substances, such as inorganic nitrate compounds.

substitution: Replacement of a product from the market whose production is avoided by that of a coproduct from the studied system.

supply chain: Set of processes, upstream of the production site, that provide what is needed for the considered product or service.

sustainable development: Global and comprehensive approach aimed at addressing the needs of the present generations without compromising the ability of future generations to meet their own needs.

system: Group of dynamically interacting elements, organized to achieve one or more functions. It is identified by the elements it contains, called *processes*, the links between these elements, and the boundaries that delineate it from the surroundings (environment and economy).

system boundaries: Delimitation of the processes to be considered for modeling the system studied, including all the processes necessary to fulfill its function.

system expansion: Method of accounting for coproducts by avoiding allocation. Resources and emissions associated with substituted product(s) are avoided and subtracted from those of the main system.

technological correlation: Correlation between the technology and materials under study and those of which the data is representative.

temporal correlation: Time difference between the study and the period of which the data is representative.

terrestrial acidification: Natural phenomenon amplified in recent years by an increase in certain atmospheric pollutants (mainly NOx and SO_2), resulting in a loss of mineral nutrients for trees and vegetation.

terrestrial ecotoxicity: Toxicity with respect to living terrestrial organisms, excluding human beings.

terrestrial nutrification: Excessive enrichment of a terrestrial environment by nutrients (nitrogen, phosphorus).

unit process: Smallest elements of the system, each corresponding to a unique activity or group of operations, for each of which inputs and outputs are quantified.

useful energy: Energy actually used by the consumer, taking into account the energy efficiency of equipment involved.

wastewater sludge: By-product of wastewater treatment.

wastewater treatment plant: Infrastructure for the treatment of wastewater.

weighting: Optional step of the environmental life cycle impact assessment, defining the relative importance of characterization scores, based on the relative social value attributed to the various midpoint or damage categories.

weighting factor: Relative importance of characterization scores, based on the relative social value given to various midpoint or damage categories.

willingness to pay: Most-used approach for monetization, consisting of defining weighting factors based on the amount of money an organization or individual would be willing to pay to avoid a given damage.

win-win situation: Case with reduction of both environmental burdens and costs.

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