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Evaluation of environmental carrying capacity and application of the sustainability target method

Chatpet Yossapol
New Jersey Institute of Technology

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ABSTRACT

EVALUATION OF ENVIRONMENTAL CARRYING CAPACITY AND APPLICATION OF THE SUSTAINABILITY TARGET METHOD

by
Chatpet Yossapol

The environmental carrying capacity (CC) is defined as the capacity of the earth to absorb or tolerate potentially stressful burdens imparted at various scales and locations, that is, to accommodate the ecological stresses without showing permanent damage. The CC can be used as a reference dataset for Life Cycle Assessment (LCA) purposes and as a baseline for other environmental studies.

In this research, a set of impact-oriented U.S. CC is developed for both input- and output-related impacts. CC for eight common impact categories is evaluated: resource depletion, global warming, ozone depletion, acidification, eutrophication, photochemical ozone formation, human toxicity, and eco-toxicity. Numerous sources of information and various environmental models are used to estimate the CC at the appropriate scales. The CC for output-related impacts is mostly based on the threshold-oriented technique using threshold concentrations in environments. A CC is basically determined from the emission that causes the environmental conditions not exceeding the threshold levels.

The CC estimates are applied as the baseline reference for the Sustainability Target Method (STM), a Life Cycle Impact Assessment (LCIA) method, in three LCA case studies. The STM is a single-score LCIA method which offers an absolute metric for environmental performance evaluation. The STM not only compares alternatives in terms of environmental performance, but also evaluates the performance by identifying the significance of impact in relation to the earth's carrying capacity. The case studies

presented are the LCA of electrical energy generation using various fossil fuels, the production of various basic materials, and the production of a coffee maker. The results are compared with those of other LCIA methods: Eco-Indicator 95, Eco-Indicator 99, EPS, and EDIP.

The advantages of using the STM in conjunction with the CC estimates are that: it provides an absolute metric related to environmental sustainability; it allows economic consideration; it eliminates the subjective weighting procedure inherent in other LCIA methods; it deals with the temporal and spatial variations in life cycle stages; and it is flexible and not limited to the selection of impacts.

EVALUATION OF ENVIRONMENTAL CARRYING CAPACITY AND APPLICATION OF THE SUSTAINABILITY TARGET METHOD

by
Chatpet Yossapol

A Dissertation
Submitted to the Faculty of
New Jersey Institute of Technology
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Department of Civil and Environmental Engineering

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APPPROVAL PAGE

EVALUATION OF ENVIRONMENTAL CARRYING CAPACITY AND APPLICATION OF THE SUSTAINABILITY TARGET METHOD

Chatpet Yossapol

Dr. Daniel J. Watts, Dissertation Advisor Date
Executive Director of Otto H. York Center for Environmental Engineering and Science
and National Panasonic Endowed Chair in Sustainability Studies, NJIT

Dr. Lisa B. Axe, Dissertation, Dissertation Co-Advisor Date
Associate Professor of Civil and Environmental Engineering, NJIT

Dr. Reggie J. Caudill, Dissertation Co-Advisor Date
Executive Director of Multi-Lifecycle Engineering Research Center and
Professor of Industrial and Manufacturing Engineering, NJIT

Dr. Methi Wecharatana, Committee Member Date
Associate Chair for Graduate Studies
and Professor of Civil and Environmental Engineering, NJIT

Dr. Hsin-Neng Hsieh, Committee Member Date
Professor of Civil and Environmental Engineering, NJIT

BIOGRAPHICAL SKETCH

Author: Chatpet Yossapol

Degree: Doctor of Philosophy

Date: August 2006

Undergraduate and Graduate Education:

- Doctor of Philosophy in Environmental Engineering,
  New Jersey Institute of Technology, Newark, NJ, 2006
- Master of Engineering in Environmental Engineering,
  Chulalongkorn University, Bangkok, Thailand, 1994
- Bachelor of Engineering in Environmental Engineering,
  Khon Kaen University, Khon Kaen, Thailand, 1990

Major: Environmental Engineering

Presentations and Publications:

Luo, Y., Wirojanagud, P, and Yossapol, C. (2001). "Comparison of Major Environmental Performance Metrics and Their Application to Typical Electronic Products." Proceedings of 2001 IEEE International Symposium on Electronics and the Environment. Denver, May 2001.

Yossapol, C. et al. (2002). "Carrying Capacity Estimates for Assessing Environmental Performance and Sustainability." Proceedings of 2002 IEEE International Symposium on Electronics and the Environment. San Francisco, May 2002.

Rodprasert, N., Tantemsapya, N., Yossapol, C., and Wirojanagud, W. (2006). "Environmental Impact Evaluation of Fluorescent Lamp Using Life Cycle Assessment." Proceedings of 2006 International Conference on Hazardous Waste Management for a Sustainable Future. Bangkok, Thailand, January 2006.

This dissertation is dedicated to my parents, my sister, my brother,
and my beloved dear friend, Siriwan Limmongkol.

ขออุทิศความพยายามที่บากบั่น เพื่อบุพการีอันเป็นที่รักยิ่ง เพื่อน้องที่รักทั้งสอง
และเพื่อนรักที่จากไป ผู้คอยเป็นกำลังใจเสมอมา สิริวรรณ ลิ้มมงคล

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ABBREVIATIONS

ABS	Acrylonitrile Butadiene Styrene
ADI	Acceptable Daily Intake
ADP	Abiotic Depletion Potential
A_{eq}	Aspect Equivalent Level
AHP	Analytical Hierarchy Process
AP	Acidification Potential
A_R	Aspect Reference
AS	Acidifying Substance
ATSDR	Agency for Toxics Substances and Disease Registry
BAU	Business-as-Usual
BDP	Biotic Depletion Potential
BGS	British Geological Survey
BOD	Biochemical Oxygen Demand
CAA	Clean Air Act
CAAA	Clean Air Act Amendments
CC	Carrying Capacity
CERES	Coalition for Environmental Responsibility Economics
CFC	Chlorofluorocarbon
CIA	Central Intelligence Agency
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CML	Centrum voor Milieukunde (Center of Environmental Science, Leiden University, the Netherlands)

COD	Chemical Oxygen Demand
CPF	Cancer Potency Factor
CPM	Center for Environmental Assessment of Product and Materials Systems, Chalmers University of Technology, Sweden
DALY	Disability Adjusted Life Year
DU	Dobson Unit, 100 DU equals to 1 mm at the standard temperature and pressure
EB	Environmental Burden
EC	European Commission
EC_{50}	50% Effect Concentration
ECB	European Chemicals Bureau
ECC	Economic Carrying Capacity
ECE	Economic Commission for Europe
ECI	Environmental Condition Indicator
ECETOC	European Centre for Ecologicology and Toxicology of Chemicals
ECOTOX	EPA's Ecological Toxicology Database
EE	Eco-Efficiency
EESC	Equivalent Effective Stratospheric Chlorine
EDIP	Environmental Design of Industrial Products
EI	Environmental Impact
EI95	Eco-Indicator 95
EI99	Eco-Indicator 99
EIA	Environmental Impact Assessment
EKMA	Empirical Kinetic Modeling Approach

ELU	Environmental Load Unit
EP	Eutrophication Potential
EPS	Environmental Priority Strategies
EPA	United States Environmental Protection Agency
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GCM	General Circulation Model
GDP	Gross Domestic Product
GGP	Global Gross Product (World GDP)
GHG	Greenhouse Gas
GRI	Global Reporting Initiative
GTC	Gigatons (10^9 tons) Carbon
GWP	Global Warming Potential
HAP	Hazardous Air Pollutant
HPV	High Production Volume
HQ	Hazard Quotient
HTP	Human Toxicity Potential
I	Impact
ICI	Imperial Chemical Industries
IDEMAT	The Netherlands' Inventory Data of Materials
IPCC	Intergovernmental Panel on Climate Change
IPCS	International Programme on Chemical Safety
I_R	Impact Reference

IRIS	Integrated Risk Information System
IRWR	Internal Renewable Water Resources
IS	IPCC Scenarios
ISO	International Standards Organization
JCM	Java Climate Model
LA	Load Allocation
LC_{50}	50% Lethal Concentration
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LIME	Life Cycle Impact Assessment Method Based on Endpoint Modeling
LOAEL	Lowest Observed-Adverse Effect Level
MCL	Maximum Contaminant Level
MERC	NJIT Multi-lifecycle Engineering Research Center
MIR	Maximum Incremental Reactivity
MJ	Mega Joules (10^6 Joules)
MOS	Margin of Safety
MPI	Management Performance Indicator
MRL	Minimum Risk Level
MW	Molecular Weight
NAAQS	National Ambient Air Quality Standards
NADP	National Acid Deposition Program
NAE	National Academe of Engineer

NAPAP	National Acid Precipitation Assessment Program
NASA	National Aeronautics and Space Administration
NB	Natural Burden
NCC	Natural Carrying Capacity
NMVOC	Non-methane Volatile Organic Compounds
NOAA	United States National Oceanographic and Atmospheric
NOAEL	No-Observed Adverse Effect Level
NOEC	No-Observed Effect Concentration
NP	Nutrification (Eutrophication) Potential
NJIT	New Jersey Institute of Technology
NPS	National Park Service
NRC	National Research Council
NSC	National Safety Council
ODP	Ozone Depletion Potential
ODS	Ozone Depleting Dubstance
OECD	Organization for Economic Co-operation and Development
OPI	Operational Performance Indicator
OZIP	Ozone Isopleth Plotting
OZIPR	Ozone Isopleth Plotting for Research
P	Price
PAF	Potential Affected Fraction
PAH	Poly-aromatic hydrocarbon
PDF	Potentially Disappeared Fraction

PEC	Predicted Environmental Concentration
PNEC	Predicted No-Effect Concentration
POCP	Photochemical Ozone Creation Potential
ppm	part per million (10^{-6})
ppmv	part per million (by volume)
ppt	part per trillion (10^{-12})
QALYs	Quality Adjusted Life Years
RDI	Resource Depletion Index
RfC	Reference Concentration
RfD	Reference Dose
RIVM	The Dutch National Institute for Public Health and Environmental Hygiene
RP	Resource Productivity
RQ	Risk Quotient
SETAC	Society of Environmental Toxicology and Chemistry
SF	Site Factor
SP	Service Productivity
SRES	Special Report on Emission Scenarios of the Intergovernmental Panel on Climate Change
STM	Sustainability Target Method
TMDL	Total Maximum Daily Load
TRACI	Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts
U.S.	United States
UK	United Kingdom

UN	United Nations
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environmental Programme
UNFCCC	United Nations Framework Convention on Climate Change
USES	Uniform System for the Evaluation of Substances
USGS	United States Geological Survey
UV	Ultraviolet
UVB	Ultraviolet B
V	Value
VOC	Volatile Organic Compounds
VP	Value Productivity
V_R	Value Reference
USGS	United States Geological Surveys
WBCSD	World Business Council for Sustainable Development
WCI	World Coal Institute
WEP	Weighted Environmental Impact Potential
WGI	Workgroup I of the Intergovernmental Panel on Climate Change
WGII	Workgroup II of the Intergovernmental Panel on Climate Change
WGIII	Workgroup III of the Intergovernmental Panel on Climate Change
WHO	World Health Organization
WLA	Waste Load Allocation
WMO	World Meteorological Organization
WQS	Water Quality Standard

WR	Weighted Resource Consumption
WRE	Wigley, Richels, and Edmonds
WRI	World Resource Institute
WTP	Willingness-to-Pay
WWP	Weighted Working Environmental Impact Potential

CHAPTER 1

INTRODUCTION

1.1 Introduction

A number of businesses and industries have attempted to evaluate environmental aspects of their products and services using environmental performance indicators. The basic idea of an environmental performance evaluation framework is to identify and assess environmental performance indicators or environmental performance metrics that can be measured and tracked to facilitate continuous improvements or at the least to prevent further environmental degradation.

Life Cycle Assessment, LCA, is increasingly used as a tool for environmental performance evaluation since it covers all life-cycle stages of products, starting from resource extraction to end-of-life management. Moreover, an analysis of environmental impacts can be achieved through Life Cycle Impact Assessment or LCIA, an important component of the LCA framework. Various LCIA methods have been developed to serve this purpose including methods that offer single scores; however these methods are seriously hindered by weighting techniques that lack real-world sophistication and are frequently subjective.

The Sustainability Target Method, STM, is an advanced LCIA method for analysis of environmental performance of a product, a process, a service, or a system. The STM was formulated by Lucent Technologies and developed through the collaboration of Lucent Technologies, Agere Systems, and the NJIT Multi-lifecycle Engineering Research Center (Caudill et al. 2002). The STM can be used to evaluate the environmental impacts associated with business activities in terms of achieving a practical sustainability target. The STM provides a new perspective on environmental performance evaluation by using a non-subjective sustainability target as the basis for establishing an absolute indicator. This can be regarded as a development of an environmental performance evaluation methodology towards the precautionary principle, where preventive action should be taken before the environmental damage could be measured. The STM also provides a practical basis for applying the earth's carrying capacity as the reference for the calculations of its key parameters including Eco-Efficiency. Carrying capacity can be defined as the capacity of the earth to absorb or tolerate burdens of potentially stressful types imparted at various geographical scales and locations, that is, to accommodate the stresses without showing permanent damage.

The essential function of the STM is to form a link between the economic value of a system and the environmental carrying capacity. Therefore, it might consider that the STM accounts for two elements in sustainability's triple bottom-line namely, both the economic and environmental dimensions. The third element, the social dimension, may also be included if it can be expressed in terms of monetary value.

1.2 Problem Statement

Carrying capacity is fundamental to the STM; consequently, it is important to develop carrying capacity estimates based on robust scientific methods and supporting documents. Furthermore, the carrying capacity should be extended to the commonly regarded environmental impact categories that have been brought forward in other LCIA approaches. A set of scientifically sound carrying capacities will strengthen the credibility and extend the applicability of the STM. This dissertation research is motivated by the need for reliable carrying capacity estimates aligned with common impact categories that can be used as the STM reference values or other uses.

1.3 Objectives and Scope

The primary objective of this research is to develop methodologies to estimate environmental carrying capacity values based on strong environmental engineering and scientific principles. The impact categories taken into account are the ones commonly considered in LCA including global warming, stratospheric ozone depletion, acidification, eutrophication, photochemical ozone formation, human toxicity, eco-toxicity, and resource depletion. The carrying capacity estimates are further validated by comparing these values to those available in the literature.

The secondary objective of this research is to demonstrate and evaluate the applicability of the estimated carrying capacities in conjunction with the STM. This objective is accomplished by comparing the performance and results of implementing the STM with those obtained with other LCIA methods.

The scope of this dissertation covers topics related to the evaluation of carrying capacity and implementation of the STM based on these carrying capacities. The literature review in Chapter 2 covers the background of LCA and LCIA, critical assessments of widely-used advanced LCIA methods including the STM, reviews of environmental impact categories, and reviews of potential approaches and methodologies for estimating carrying capacity. Chapter 3 presents the objectives and hypotheses of this research. Chapter 4 presents the development of the threshold-oriented technique as used in this research as the primary method for evaluating emission-related carrying capacities.

Chapter 5 through Chapter 12 constitute the core of this dissertation. These chapters cover the methodologies, assessments, and evaluations of carrying capacity for the following impact categories: global warming, stratospheric ozone depletion, acidification, eutrophication, photochemical ozone formation, human toxicity, eco-toxicity, and resource depletion.

Chapter 13 demonstrates how the carrying capacity estimates are used in STM as applied in three case studies: at the process level, supply-line level, and product level. The results are compared with those of other LCIA methods. Chapter 13 also provides a preliminary set of carrying capacity estimates for Europe using the methodologies developed in this research. Sensitivity analysis is presented in Chapter 14, including an uncertainty analysis of using the carrying capacity in the STM. Lastly, conclusions and recommendations for further research are presented in Chapter 15.

1.4 Scholarly Contributions of This Research

The eventual goal of this research is to provide a practical basis for achieving a sustainable society by using a credible environmental performance evaluation tool for a product, a service, or a system. In the attempt to accomplish this goal, this research hopes to make the following unique scholarly contributions:

- Develop a technique to estimate the emission-related environmental carrying capacities;
- Develop a reserve-time horizon technique to estimate the consumption-related environmental carrying capacities;
- Formulate and validate a set of U.S.-based carrying capacity estimates; and
- Evaluate the environmental performance of selected case studies utilizing the STM based on carrying capacity estimates.

CHAPTER 2

LITERATURE REVIEW

2.1 Overview

The purpose of this chapter is to examine and review the research and progress relevant to the development of methods for life cycle impact assessment and environmental carrying capacity estimates. This chapter contains six sections beginning with this overview. Section 2.2 provides an introduction to Life Cycle Assessment and the background of Life Cycle Impact Assessment, which is a key component of the LCA framework. Review of six LCIA methods that offer single scores is made in Section 2.3. Section 2.4 provides an introduction to the Sustainability Target Method (STM). The STM is being developed as a comprehensive LCA-based metric that yields a systematic single score related to sustainability and resource productivity evaluation of products and services. The STM can also be considered as a single score LCIA method for the LCA context. The implementation of the STM requires a set of environmental carrying capacity values, which is a major focus of the research program discussed in this dissertation.

Section 2.5 starts with a review of common environmental impact categories used in LCA and in other LCIA methods. Then a survey of potential methodologies for obtaining carrying capacity estimates is provided. This survey includes the potential methodologies and approaches that have concepts similar to or the same as carrying capacity. Lastly, Section 2.6 summarizes this literature review and provides explicit problem statements that form the motivation for this dissertation research.

2.2 Life Cycle Impact Assessment Background

Life Cycle Impact Assessment (LCIA) is the third component of the traditional Life Cycle Assessment (LCA) framework. LCA is a “cradle-to-grave” approach for assessing environmental-related activities. “Cradle-to-grave” begins, for example, with the mining of raw materials from the earth to create products and services and ends when the materials are disposed of to the earth. LCA evaluates all stages of a product’s life-span and estimates the cumulative environmental burdens.

Basically, LCA is a framework for assessing environmental aspects[1] and potential impacts[2] associated with a product or service by compiling an inventory of environmental burdens, evaluating the potential environmental impacts associated with the burdens, and interpreting the results for decision making. Different from other environmental performance evaluation frameworks such as Eco-Efficiency (WBCSD 1992), the Global Reporting Initiative’s Sustainability Report (GRI 2002), the ISO 14031 (ISO 1999), and the Ecological Footprint (Wackernagel et al. 1999) (the review of these frameworks and guidelines is presented in Appendix A), LCA may bridge the gaps of some issues that need to be advanced. As summarized by NAE (1998) and Olsthoorn et al. (2001), these issues include:

- The need for more standardization and a practicable approach;
- Measurement of sustainability;
- Life cycle assessment framework; and
- A narrower but deeper analysis of environmental impacts.

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[1] An environmental aspect is an element of an organization's activities, products or services that can interact with the environment.

[2] An environmental impact is any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organization's activities, products or services.

Life cycle consideration can be incorporated into the environmental performance evaluation by following the LCA approach of SETAC (Society of Toxicology and Chemistry). In LCA, the LCIA step is a means to assess technical issues of environmental impacts. By using LCIA, the exempted narrow and deep analysis of environmental impacts can be implemented.

The term "life cycle" refers to the major activities in the course of the product's life-span including its manufacture, use, disposal, and reuse. LCA serves several purposes including environmental reporting and declaration, environmental accounting, internal decision-making, being a supporting tool for design for the environment, and education for interested audiences. The LCA conceptual framework consists of four major components: goal and scope definition, life cycle inventory, life cycle impact assessment, and life cycle interpretation as presented in Figure 2.1 (Fava et al. 1993; ISO 1998a).

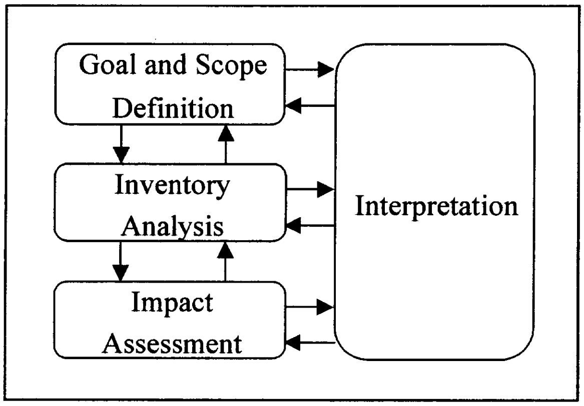

Figure 2.1 Life cycle assessment framework.
(Source: Fava et al. 1993)

LCIA builds on the analysis of the Life Cycle Inventory (LCI), which lists the environmental emissions and resource consumption throughout different stages of a product's life. LCIA is the evaluation of potential human health and environmental impacts of the resource consumption and environmental releases identified during the LCI. The impact assessment addresses ecological and human health impacts as well as resource depletion. Furthermore, LCIA establishes a linkage between a product or process and its potential environmental impacts. The development of measures of actual impact on ecological and human health as well as resource depletion leads to impact assessment. In practice, LCIA converts the results of an LCI to a group of common impact measures such as global warming or ozone depletion, that allows interpretation of the total environmental impacts of the system being evaluated (Fava et al. 1993).

In the U.S., SETAC and EPA are the leading organizations that point out the importance and need for the development of advanced LCIA (Fava et al. 1993; EPA 1993). Often, the LCIA development is limited due to the scarcity and relatively poor quality of data. In Europe, LCIA has been driven by a desire for public policy to obtain a single numerical indicator or single score that can be readily interpreted. Many LCIA methods have been proposed and developed but none of them is yet used sufficiently widely to be considered as a standard. Thus, LCIA is still an emerging research topic for scientific analysis. Many impact assessments are currently being performed; some of them yield only qualitative and subjective results. Studies designed to lead to a more sophisticated LCIA will surely contribute to LCA development.

The standardized ISO 14042 “Life Cycle Impact Assessment” approach describes LCIA as consisting of seven steps where the first three steps are mandatory (ISO 1998b) and the remaining steps are optional depending on the goal and scope of the LCA. These LCIA steps are:

Selection and definition of impact categories: identifying relevant environmental impact categories (e.g., global warming, acidification). An impact category can be defined as a group or class of inventory inputs/outputs that share common environmental attributes; such as mutual mechanism of actions that can lead to an endpoint.

Classification: assigning LCI results to the impact categories (e.g., classifying CO_2 emissions to global warming).

Characterization: modeling LCI impacts within impact categories using science-based equivalency factors[1] (e.g., modeling the potential impact of methane on global warming).

Normalization: expressing potential impacts in ways that can be compared (e.g., comparing the total emissions or resource use for a given area on a per capita basis).

Grouping: sorting or ranking the indicators (e.g., sorting the indicators by environmental safeguard: resources, human health, ecosystem health).

Weighting: emphasizing the most important impacts.

Evaluating and reporting: gaining a better understanding of the LCIA results and communicating with stakeholders.

An LCIA provides a systematic procedure for classifying and characterizing the environmental impacts. A typical concept of LCIA requires information about loadings (environmental emissions) and resource uses in the form of a numerical indicator or index for each impact category. The indicators are the basis for making comparisons or considerations. LCIA indicators are approximations and simplifications of aggregated loadings and resource use. The typical results of an LCIA provide a checklist showing the relative differences in potential environmental impacts for each option, in other

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[1] An equivalency factor or characterization factor is a factor that describes the relative harmfulness of an environmental intervention within an environmental impact category. A factor is a result of modeling environmental impact.

words, an environmental profile. For example, an LCIA would identify which product or process causes more greenhouse gases or could potentially harm human health more (through the characterization phase). A set of traditional LCA results does not allow making comparisons among products or services in terms of overall environmental performance. However, such comparison can be accomplished using LCIA single scores.

For an LCIA to yield a single score, the scores of every impact category in the LCA are aggregated to produce a single indicator or a single score. By using this LCIA single score, comparison among systems, products, services, or materials is more explicit than by using multiple indicators of each impact category. Furthermore, the overall superiority of a product may be identified by using a single score, provided that the comparison is based on the same functional unit (i.e., products that provide the same service) or the same benchmark.

An aggregated single score can be evaluated by applying the weights for individual impacts, or impact categories, in the weighting step of LCIA. In other words, the multiple results of all impact categories are converted into one single quantity by using weighting factors. The weighting can be defined as the qualitative or quantitative element through which the relative importance of different environmental impacts can be weighted against each other (Udo de Haes 2000).

Weighting methodologies are not as scientifically-based as the characterization methodologies. As a result, weighting factors are less used. The weighting factors are less scientifically-based because there is no absolute answer as to why one impact is more important than the others (Vogtlander and Bijma 2000). Therefore, subjective value judgment is often used to assign the weighting factors rather than natural sciences. The use of value judgments for this purpose is probably the most controversial issue in LCA (Udo de Haes 2000; Hertwich and Hammitt 2001). Furthermore, it is also of concern that the use of a single score may cause loss of the transparency of LCA. As a result, most of today's LCAs are still presented in the form of multiple results corresponding to impact categories, or environmental profile, rather than single results.

According to Udo de Haes (2000), the outcome of the weighting is fully dependent upon the values of those who make the judgment. Because basic weighting techniques may contain some degree of subjectivity, weighting is an optional element of LCA as suggested by the ISO procedure. ISO 14042 states, "weighting across categories shall not be part of a comparative assertion disclosed to the public" (ISO 1998b).

In spite of the difficulties, the results of weighting can be practical and applicable. A single score is still useful, for instance, for internal use, where choices between products/materials have to be made and agreement on weighting factors exists (Vogtlander and Bijma 2000). A single score can also be used to interpret LCA results for non-environmental specialists (Vogtlander and Bijma 2000). Furthermore, as pointed out by Emblemsvag and Bras (1999), LCA will not survive long in the commercial world without comparability and benchmarkability. Therefore, single score LCIA methods, which allow a comparative assessment for the overall superiority of products or services, still need to be developed.

Consequently, some studies to develop systematic weighting factors have been initiated. According to Finnveden (1997) and Goedkoop (1998), weighting techniques to convert normalized multiple results to a single result may be classified into six categories as follows:

- Use of social evaluation;
- Use of the costs incurred in preventing the environmental impacts;
- Use of the energy consumption necessary to prevent the environmental impacts;
- Use of only one environmental impact to avoid the use of weighting factors;
- Use of judgment of and evaluation of the impact by experts; and
- Use of the degree by which a target level is exceeded.

Examples of basic weighting techniques for LCIA are presented by the works studied by Ahbe et al. (1990 as cited in Goedkoop 1998), Stone and Tolle (1998), Sangel et al. (1999), Lee (1999), Itsubo et al. (2004), and Soares et al. (2006). Generally, these weighting techniques rely on value judgment to identify the weights among environmental impact categories (Table 2.1). An example of surveyed basic weighting factors is presented in Table 2.2. In this table, relative rankings of impact categories are presented in the parentheses.

Table 2.1 Summary of Weighting Techniques Used in LCIA

Weighting technique	Description	Reference
Distance to target	Use of correlations between the seriousness of an impact and the distance between current levels and target levels.	Ahbe et al. 1990 as cited in Goedkoop 1998
Use of Analytical Hierarchy Process (AHP)	Use the AHP to identify the weights of environmental impacts using relative importance between impacts on a pair-wise basis.	Stone and Tolle 1998
Societal weighting	Use of fuzzy ordering method to obtain weights using a matrix of relative weights of environmental impacts.	Sangel et al. 1999
Combined distance to target and relative significance factors	Modify the distance to target method by applying the relative significance factors of environmental impacts. The relative significance factors can be derived from weighting identifying methods.	Lee 1999
Panel approach and Multi Criteria Decision Aid	Use of value judgment based on multiple criteria. Definite scores are assigned among choices, according to distance-to- target before calculate weighting factors.	Soares et al. 2006

Table 2.2 Importance of Impact Categories as a Percentage and Rank of Total Impact (Source: Schmidt and Sullivan 2002)

Impact category	Survey experts	Survey Germany	Survey EU
Energy	17 % (1)	7 % (7)	7 % (8)
Raw materials	1 % (7)	5 % (8)	4 % (9)
Water	≈ 0 % (8)		
Global warming	16 % (2)	8 % (5)	12 % (4)
Ozone depletion	16 % (2)	17 % (1)	14 % (3)
Acidification	12 % (5)	13 % (2)	8 % (6)
Nutrification	9 % (6)	4 % (9)	15 % (2)
Summer smog		7 % (6)	3 % (10)
Human toxicity		17 % (1)	18 % (1)
Eco-toxicity	15 % (3)	12 % (3)	7 % (7)
Waste	14 % (4)	9 % (4)	12 % (5)

In addition to the basic weighing techniques mentioned above, there are a few full-stage LCIA methods developed as a tool offering single scores. Many scientific approaches are used in these attempts to make the weighting in LCIA as scientifically based as possible. The fully developed LCIA methods offering single scores available today use different concepts and techniques to convert LCI results to single scores. These fully developed LCIA methods (LCIA methods hereafter) including Eco-Indicator 95 (EI95), Eco-Indicator 99 (EI99), Environmental Priority Strategies (EPS), Environmental Design of Industrial Products (EDIP), IMPACT2002+, and Life Cycle Impact Assessment Method Based on Endpoint Modeling (LIME) are reviewed in the following section. The Sustainability Target Method (STM), a newly developed LCIA method, is reviewed in a subsequent section.

Some of the fully developed LCIA methods (all LCIA steps are integrated) employ complex procedures with extended library databases, some of them are implemented in commercially-available software, e.g., SimaPro (PRe' 2006) and EPS Design System (CPM 2002). The integration of an advanced LCIA method and an LCI database in a software tool offers the ability to put LCA into practice in an economical and more convenient way. The use of LCA software tools is increasingly popular but none of the LCIA methods is regarded as a standard. Therefore, there is still room for the development and improvement of full-stage LCIA methods offering single scores. As pointed out by Graedel (1998), a complete and quantitative LCA has never been carried out nor is likely to be. It is therefore better to start with less sophisticated approaches and work towards the more refined ones.

2.3 Life Cycle Impact Assessment Methods

2.3.1 Eco-Indicator 95 Distance-to-Target Method (Goedkoop 1998)

Eco-Indicator 95 Method (EI95) was developed by PRe' Consultants for the Dutch Government in conjunction with several manufacturing companies and research agencies. In EI95, impacts are grouped in the categories of human health, ecosystem health, and resources. The EI95 method yields a single score based on the conventional LCIA method that considers classification, characterization, normalization, and weighting. The environmental burdens are first aggregated within a number of environmental impacts with which they are associated. The impacts are then categorized according to the degree to which they contribute to the overall environmental situation. Equivalency factors are used in the characterization stage to calculate the impact scores for individual impact categories. The result is a profile of characterized impact scores. Then these scores are normalized to three different damages (fatalities per one million population, health complaints regarding health impairment, and five percent of ecosystem impairment) based on a European scale.

Next, the different impact categories are weighted and summed to form a single Eco-Indicator value, an "imaginary" unit with which impacts can be compared. EI95 generates a single score by using weighting factors for a variety of adverse environmental impacts produced during the life cycle stages. The degree by which a defined target level (the level for which the damage is acceptable) is exceeded is used to weight the different environmental impacts. This "distance-to-target" weighting derivation method is based on the critical pollution load (target level), which considers the scarcity of environmental absorption capacity by relating a load to a critical load. The greater the gap between the current environmental impact (emission) and the target level, the higher the rating given to the seriousness of the impact. A single score is the sum of the weighted points from the environmental burdens that have been taken into account. There are nine impact categories in EI95 as illustrated in the Figure 2.2.

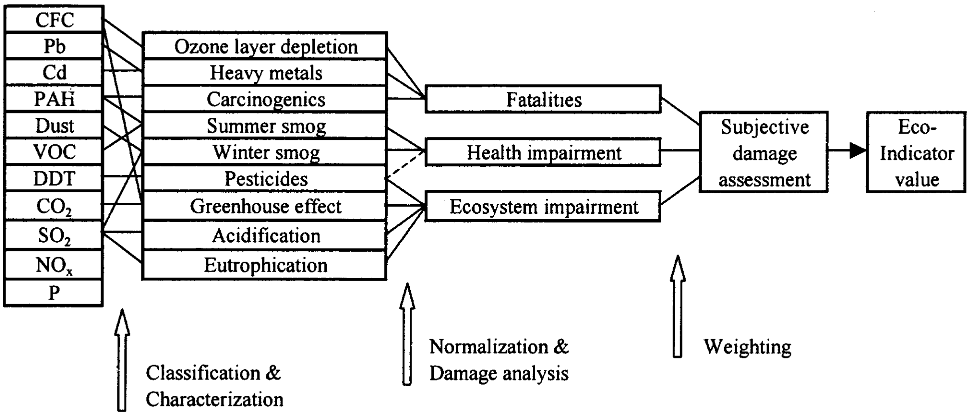

Figure 2.2 Structure of Eco-Indicator 95 method.
(Source: Goedkoop 1998)

The target levels for EI95 were developed for Europe. In EI95, there is a correlation between the seriousness of an impact and the distance between the current emission level and the target emission level (the so-called target value in EI95). Thus, if an impact has to be reduced by a factor of 10 to achieve a target sustainable level and another impact has to be reduced by a factor of 5, then the first impact is regarded as being twice as serious. In other words, the reduction factor is the weighting factor. These reduction factors or the weighting factors were largely determined based on data from governmental documents such as those from the Dutch National Institute for Public Health and Environmental Hygiene (RIVM). Table 2.3 presents the reduction factors used in EI95. These reduction factors were derived based on the emission amounts of respective pollutants and the environmental conditions in Europe in 1990.

Table 2.3 Reduction Factors in Eco-Indicator 95
(Source: Goedkoop 1998)

Impact Category	Reduction Factor	Criteria for Target levels
Ozone layer depletion	100	Probability of one death per year per million population
Heavy metals	5	Air- Lead content in blood of children, limited life expectancy and learning performance of people Water- Cadmium content in air
Carcinogens	10	Probability of one death per year per million population
Summer smog	2.5	Prevent smog period and health complaints
Winter smog	5	Prevent smog period and health complaints
Pesticides	25	5% of ecosystem is disrupted
Greenhouse effect	2.5	Increase in temperature for 0.1°C per decade
Acidification	10	5% of ecosystem is damaged
Eutrophication	5	5% of ecosystem is damaged

In EI95 the procedure can be expressed in a simple equation form as (Goedkoop 1998):

$$I = \sum_i W_i \times \frac{E_i}{N_i} \times \frac{N_i}{T_i} = \sum_i W_i \times \frac{E_i}{T_i} \qquad (2.1)$$

Where I denotes the eco-indicator value, W_i is the weighting factor (reduction factor) which express the seriousness of impact I, E_i is the contribution of a product or service life cycle to an impact I, N_i is the normalization value or the current extent of the European impact I, and T_i is the target value for impact i.

The major drawback of EI95 is the subjectivity of the reduction factors. This procedure considers that the different damages of the three safeguard measures are equivalent (one fatality per one million population, a health complaint regarding health impairment, and five percent of ecosystem impairment). This equivalence is determined without a scientific basis. Subjectivity is also found in the determination of reduction factors. Even though there are criteria for the determination, the reduction factors for some impact categories are subjectively selected (see also Subsection 2.5.2).

EI95 does not take into account resource depletion (availability of resources) as an impact. Instead the environmental damage for resource consumption is presented in terms of energy used for material production. However, EI95 does indicate that the resource depletion impact is already incorporated in the waste emission (less resource use means less waste emission) (Goedkoop 1998). The toxic substances that cause human health impact considered in EI95 are also limited to only heavy metals, carcinogens, summer smog, winter smog, and pesticides. For heavy metals, the developer selected lead and cadmium as the surrogates for other pollutants according to the available information. Hence only the environmental conditions affected by these two pollutants were considered. This may be seen as a drawback because different heavy metals cause different damages to human body.

Grant (2000) criticized EI95 because the method deals poorly with land use and biodiversity impacts. EI95 also does not deal with the temporal and spatial variations of the impacts. Dubreuil (1997) demonstrates this limitation by mentioning the concern regarding the emissions of SO_2 in the winter smog impact in Europe, where the origin of the problem is mainly the combustion of low quality coal. However, other activities, such as metal production, are also associated with emissions of SO_2. Europe imports a significant amount of metals in relation to its metal consumption. EI95 considers that SO_2 emissions are always associated with the winter smog impact regardless of where the metals are produced. In other parts of the world where winter smog does not exist and is not related to smelting, the eco-indicator score based on the European perspectives for metal production is not relevant.

2.3.2 Eco-Indicator 99 Damage-Oriented Method (Goedkoop and Spriensma 2001)

The Eco-Indicator 99 Method (EI99) is a modification of EI95. This method was also developed using European environmental conditions. The EI99 was intended to minimize the subjectivity of the weighting procedure of its predecessor; the EI95. EI99 has added and adjusted the impact categories offered in EI95. The weighting procedure in EI99 is based on the use of the judgment of an LCA expert group as contrasted to the distance-to-target method used in EI95. In an attempt to overcome the most critical and controversial step in EI95, the weighting step in EI99 the number of subjects (grouped impact categories) to be weighted is reduced to only three types of environmental damage or safeguard subject: human health, ecosystem quality, and resources. This smaller number of subjects to be weighted was used because it was more convenient and less complicated for the expert panel to make judgments on the weighting. The results returned from the panel experts, 45 opinions of a Swiss LCA interest group, revealed that the weightings for human health were about the same as those for ecosystem quality at 40% while the weighting factor for the resource depletion was about 20%. Based on this summary, these weights are used as the default in the EI99 weighting step to aggregate the scores across the three environmental damage areas to yield a single score (see Figure

2.3). As in EI95, the single score obtained from EI99 is recommended by the developer for internal use only.

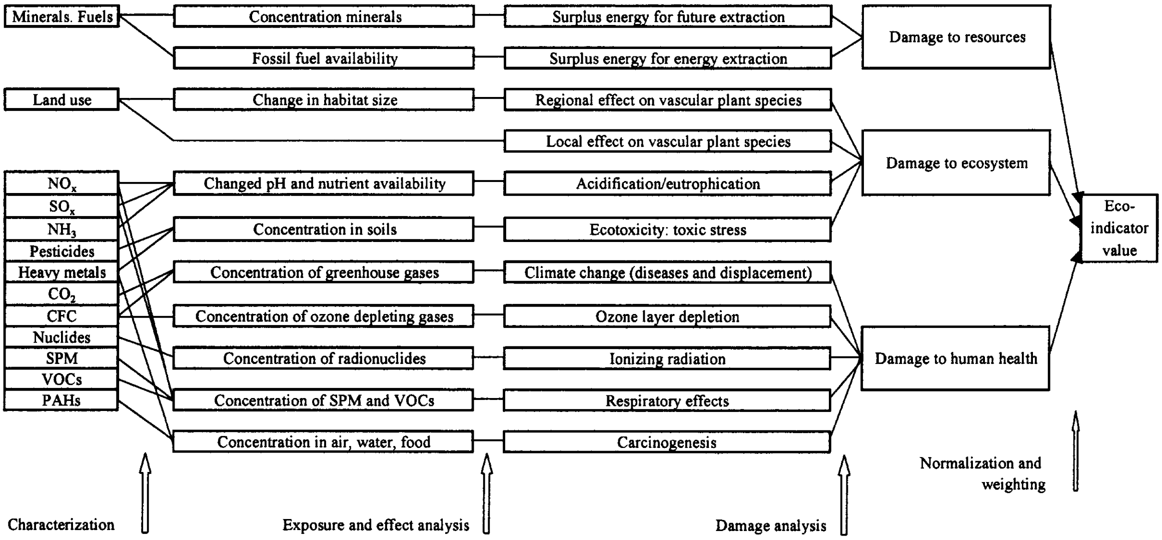

Figure 2.3 Structure of Eco-Indicator 99 method.
(Source: Goedkoop and Spriensma 2001)

EI99 assessment starts with the characterization, which places the environmental burdens into 11 impact categories. Then the characterized burdens are assessed to identify the sensitivity (effect) of the receiving environments using exposure and impact analysis. In order to do this, a number of environmental models and analysis techniques are used. Then the sensitivity of the environmental conditions are evaluated and expressed in the damage analysis in terms of Disability Adjusted Life Years (DALYs) for the impacts that cause human health damage, Potentially Disappeared Fraction (PDF) or Potential Affected Fraction (PAF) for ecosystem quality damage, and Mega Joules (MJ) for energy use in resource extractions (Figure 2.3). Table 2.4 presents the impact categories considered in EI99 and their corresponding damage categories.

Table 2.4 Damage Categories and Impact Categories in Eco-Indicator 99
(Source: Goedkoop and Spriensma 2001)

Damage Categories	Impact Categories
Human health (DALY)	Carcinogenic Respiratory impacts caused by organic substances Respiratory impacts caused by inorganic substances Climate change Ionizing radiation Ozone layer depletion
Ecosystem quality (PDF)	Ecotoxicity Acidification and eutrophication Land occupation and land conversion
Resources (MJ)	Extraction of minerals Extraction of fossil fuels

In the normalization step, the relative contribution of the calculated damage to the total damage of the reference system is determined. EI99 normalization references are based on European data. The normalized scores of the predicted damages are then aggregated using the weighting factors mentioned earlier. The EI99 procedure can be expressed in simple equation form as:

$$I = \sum_i W_i \times E_i \times \frac{D_i}{N_i} \qquad (2.2)$$

Where:

I eco-indicator value

W_i weighting factor of the damage i (e.g., 40% or 0.4 for human health damage)

E_i contribution of a product of service life cycle to an impact i (e.g., kg of SO_2 emission)

D_i damage factor of the contribution E_i (e.g., DALY/kg of SO_2)

N_i normalization value or the current extent of the European damage i (e.g., total European human health damage in DALY/yr).

EI99 has a few advantages over EI95. One of which is that EI99 focuses on the "damage-oriented" or endpoint assessment (actual impacts or damages are quantified) compared to the midpoint assessment (the assessment is based on the potential impact with regard to environmental burdens) as in EI95. In other words, EI99 evaluates the "actual damage" instead of the "potential damage" as in EI95. EI99 also eliminates the EI95's subjective assumption that the three different damages of safeguard measures are equivalent (one fatality per one million population, a health complaint regarding health impairment, and five percent of ecosystem impairment).

Overall, EI99 provides a systematic LCIA single score with the impact categories covering major environmental issues. EI99 is compatible with conventional LCI, as demonstrated in the SimaPro software (PRe' 2006). There is also a potential to apply EI99 to other regions of the world (Gomez 1998 as cited in Goedkoop et al. 1998).

However, EI99 still has some weaknesses that can be pointed out here. Subjective judgment still exists in the determination of the weighting factors (from polled panel experts). Other weaknesses, which are shared with EI95, are the incapability to deal with temporal and spatial variations and the interpretation or the meaning of eco-indicator scores. It is difficult for those not familiar with EI95 and EI99 methods to understand what would be the practical meaning of the EI95 or EI99 points.

There are also a few major assumptions implicit in EI99 that should be pointed out here because these points lead to some shortcomings. The first one is the assumption that only one species of plant can be used as the representative of an entire ecosystem in the determination of ecosystem quality damage. Another assumption that raises questions is that the resource depletion impact can adequately be expressed by the energy used for the extraction of minerals and fossil fuels (i.e., the more energy needed, the more scarce is the resource) instead of the actual damage caused by the extractions. Moreover, there are only a few minerals and fossil fuels have information available on energy consumption for their extraction, raising even more uncertainty.

2.3.3 Environmental Priority Strategies (EPS) (Steen 1999)

The EPS approach assembles all data from the LCI into a single value expressed in terms of Environmental Load Unit (ELU). The environmental performance of products/ services can be compared using ELU. An ELU is an assigned monetary value of the damage posed by an impact. The impact assessment process, which consists of the classification, characterization, and weighting, are implicit in ELU in one process.

The EPS employs a valuation of environmental impacts based on the Swedish Parliament's and the UN's general environmental objectives for the external environment. EPS defines five safeguard subjects to be included in the assessment: natural resources, biological production, human health, biodiversity, and aesthetic values. In the evaluation, the impact on each of these safeguard subjects is determined and quantified. Figure 2.4 illustrates the system flow diagram of the EPS.

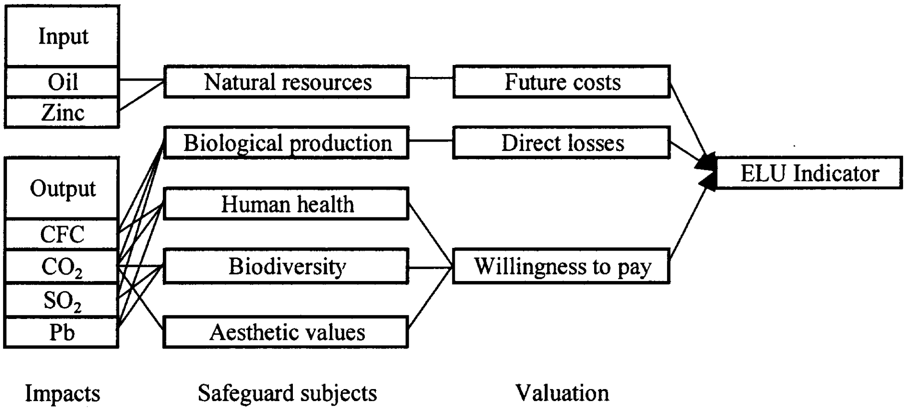

Figure 2.4 Structure of EPS method.
(Source: Goedkoop 1998)

A number of methodologies and data sources are used to estimate ELU, which is equivalent to Euro currency, for safeguard subjects. Three types of ELU valuation were implemented. The natural resources safeguard subject was valued by actual commodity prices for future extraction costs. These are the costs that must be spent in order to extract the remaining resources. For oil and coal, the costs of alternative fuel (vegetable oil) were used for oil and the price of wood was used to value coal.

For the biological production safeguard subject, the current prices of production are used. The production losses are measured directly from the estimated reduction in agricultural yields and industrial damage.

For the human health, biodiversity, and aesthetic values safeguard subjects, the monetary value was determined from the willingness to pay (WTP) to avoid damages. In other words, WTP is the value paid to restore the protected impact to its original condition after it has been affected. The change in environmental impacts is quantified through the unit effects, which are defined as the changes in the protected impacts. For instance, WTP can be seen as the sums that a society is prepared to pay for ill health or the death of its citizens, the extinction of plants and animals, and impairment of natural aesthetic values. The unit effect can be a reduction of one kilogram in seed production or a loss of man-year due to a particular disease (e.g., 85,000 ELU/person-year of life loss). The EPS procedure can be expressed in a simple form of equation as:

$$ELU = \sum_{i} W_i \times I_i \times E_i \tag{2.3}$$

Where:

ELU EPS indicator in ELU

W_i weighting factor of impact i

I_i impact index (ELU) of environmental contribution i (e.g., ELU/kg)

E_i contribution of a product of service life cycle to an impact i (e.g., kg).

A feature of EPS is providing the universal indicators (currency unit) that are understandable to industrial designers and help them make decisions based on the lowest (economic) environmental cost. The social costs of production and consumption are also implied in the evaluation of ELU. Furthermore, EPS is flexible and easily incorporates information on new materials, processes, or energy uses. This is because the cost data for material consumption, process, and energy uses are readily available. EPS has been used to evaluate the environmental impact of different materials for a component of the automobile industry (EPS was initiated by Volvo of Sweden). The most recent version is EPS 2000, which provides default ELUs for a number of environmental impacts.

EPS also carries out a sensitivity analysis of both the data and the weighting factors. A calculation to determine the uncertainty for each weighting factor is made providing that the data from the inventory phase are accompanied by uncertainty factors. This sensitivity analysis enables EPS users to examine the accuracy of the results. However, sometimes it is not clear from what information or assumptions the uncertainty factor is derived.

With regard to limitations, EPS neglects the consideration of environmental damages. EPS also lacks transparency in the method for valuation of biodiversity. Another limitation is the omission of temporal and spatial variations. The default ELU evaluation (EPS 2000) is based on prices of commodities, costs of production, and value of the prevention of environmental damage. Most factors are for global conditions in the 90's and represent average emission rates. Hertwich et al. (1997) criticized EPS because the prices were based on an unstable situation. For example, the current prices paid in royalties for timber may be underestimated compared to the full replacement value of the forest. The updating of these prices and costs due to temporal and spatial variations would require a significant amount of time and resources.

2.3.4 Environmental Design of Industrial Products (EDIP) (Wenzel et al. 1997)

EDIP is a method for incorporating environmental considerations into product development. The method, which was developed in Denmark, describes how to calculate and assess the environmental impact of a product during its life cycle stages. The method also describes how the results of the life cycle assessment can be incorporated into the product development plans as well as how to simulate various product alternatives. Basically, EDIP offers guidelines for implementing a full-stage LCA (Danish EPA 2001b). However, LCIA of a product can be carried out following its documentation or using the software tools, i.e., EDIP PC Tool (Danish EPA 2001a) and SimaPro software (PRe' 2006). Figure 2.5 illustrates the system flow diagram of EDIP's LCIA.

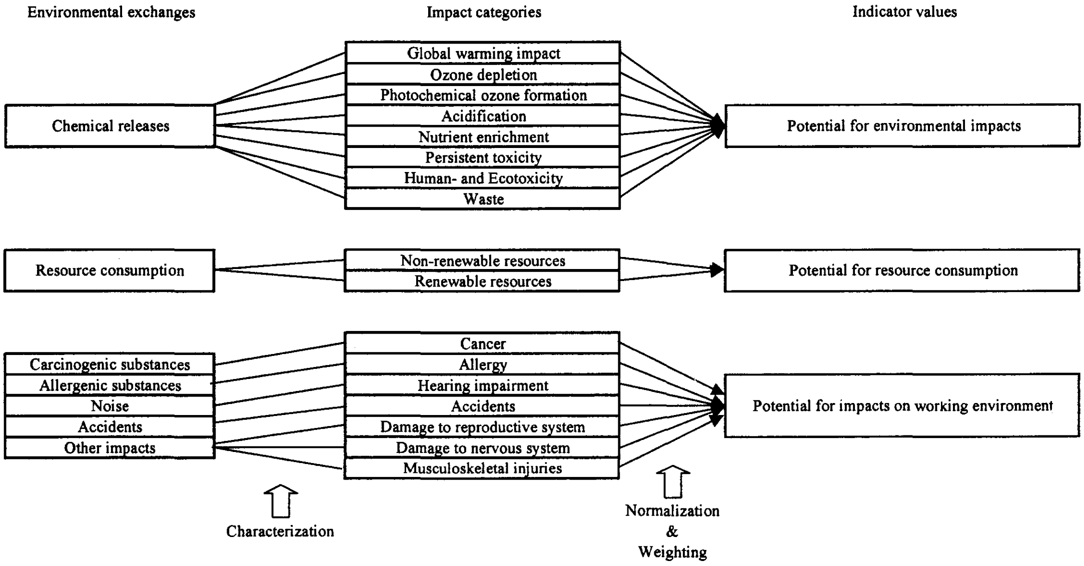

Figure 2.5 Structure of LCIA in the Environmental Design of Industrial Products.

The impact assessment procedure of EDIP framework relies on classification, characterization, and normalization with weighting using the concept of the person-equivalent to interpret the impact potentials. The three safeguard subjects considered in EDIP are environmental impacts, resource consumption, and impacts on the working environment or impact to human health.

The environmental impact group consists of eight impact categories: global warming, ozone depletion, photochemical ozone formation, acidification, nutrient enrichment, persistent toxicity, human- and eco-toxicity, and waste. Classification and characterization in EDIP are similar to those of EI95 and EI99. After environmental burdens are characterized, they are normalized to person-equivalent for the inventoried area in different scale (local-Denmark, regional-Europe, and global). The normalized environmental impacts then can be aggregated in the weighting step using weighting factors. Weighting factors in EDIP are similar to the distance-to-target method in EI95. EDIP defines the weighting factors as the relative environmental impact potential of emissions in the year 1990 compared to the target levels in the year 2000 for the default calculations. Therefore, EDIP's weighted environmental impact potential (WEP) can be formulated as (Wenzel et al. 1997):

$$WEP = \sum_{i} WF_i \times NEP_i \qquad (2.4)$$

Where:

WEP EDIP weighted environmental impact potential (person-equivalent)

WF_i weighting factor of impact category i

$$= \frac{\textit{Environmental impact potential emission in 1990 of category i}}{\textit{Target emission in 2000 of category i}}$$

NEP_i normalized environmental potential (person-equivalent)

$$= \frac{EP_i}{T \times ER_i}$$

EP_i total LCA potential impact of impact category i (e.g., kg CO_2-eq)

T duration of service (year)

ER_i reference potential impact in inventoried year of impact category i (e.g., kg CO_2-eq/person/year1990).

The resource consumption group consists of renewable and non-renewable resources. Resource consumptions are assessed individually and no characterization is made. Consumption of a resource can be normalized to person-equivalents for the global scale regardless of the site-specific characteristics because the "one world market" is assumed. The normalized resource consumption then can be aggregated in the weighting step using weighting factors. A weighting factor of resource consumption in EDIP is defined as the reciprocal of the supply horizon for the resource. Therefore, EDIP's weighted resource consumption (WR) can be formulated as (Wenzel et al. 1997):

$$WR = \sum_i WF_i \times NR_i \qquad (2.5)$$

Where:

WR EDIP weighted resource consumption (person-equivalent)

WF_i weighting factor of resource i

$$= \frac{1}{\textit{Supply horizon for resource i}}$$

Supply horizon for resource i is the amount of known reserves relative to annual consumption of resource i and it is:

For non-renewable resources

$$= \frac{\textit{Known reserves}}{\textit{Annual global consumption}}$$

For renewable resources where consumption exceeds regeneration

$$= \frac{\textit{Known reserves}}{\textit{Annual consumption} - \textit{annual regeneration}}$$

For renewable resources where consumption does not exceed regeneration

$= \textit{Infinite}$

NR_i normalized resource consumption (person-equivalent)

$$= \frac{RC_i}{T \times RR_i}$$

RC_i total LCA consumption of resource *i* (e.g., kg Aluminum)

T duration of service (year)

RR_i global resource consumption in 1990 (e.g., kg Al/person/year1990).

The impacts on the working environment safeguard subject are classified within seven impact categories: cancer, damage to the reproduction system, allergy, damage to the nervous system, hearing impairments, musculoskeletal injuries, and accidents. The impacts are summarized in terms of exposure time of the impact per employee per year. Therefore, characterization is not needed since this form is ready for normalization. The impacts on working environment are normalized with the background impact of Denmark in the year 1990. The normalized impacts based on working environment impacts then can be aggregated in the weighting step using weighting factors, which are defined as the relationship between the number of work-related injuries reported for persons exposed to the type of impact in question in Denmark and the total time exposure in Denmark to the relevant type of impact in the year 1990. EDIP's weighted working environmental impact potential (WWP) can be formulated as (Wenzel et al. 1997):

$$WWP = \sum_i WF_i \times NWP_i \qquad (2.6)$$

Where:

WWP EDIP weighted potential impacts on the working environment (person-equivalent)

WF_i weighting factor for impact i on working environment

$$= \frac{\textit{Reported injuries for impact i per year in Denmark}}{\textit{Total annual time of exposure to the impact i in Denmark}}$$

NWP_i normalized potential for impacts on the working environment = $\frac{WP_i}{T \times WR_i}$

WP_i impact potentials for the working environment for impact category i (year)

T duration of service (year)

WR_i exposure time of the impact per employee per year in Denmark for impact category i in 1990 (year/person/year1990).

A survey reported that most of the LCA experts who participated in the survey found EDIP to be the most advanced, complete and consistent LCA method available (Sorensen 2002). The former version of this LCIA method is EDIP97. The update EDIP2003 supports country-specific LCA through the characterization factors (Dreyer et al. 2003; Hauschild and Potting 2004; Potting and Hauschild 2004; Hettelingh et al. 2005). Several environmental models were used to evaluate the characterization factors for local and regional scale impact categories, i.e., photochemical ozone formation, acidification, nutrient enrichment, eco-toxicity, human toxicity, and noise.

A major characteristic of EDIP's LCIA is a toxicity assessment to determine the equivalency factors for chemical substances. As pointed out by Toffel and Marshall (2004), the impact assessment for human toxicity in EDIP incorporates each substance's toxicity, biodegradability, and dispersion in the environment. Human toxicity potential values are calculated for relative toxicity from exposure via environmental media.

Other advantages of EDIP are the transparency of the calculations and the flexibility of the selection of impact categories. Unlike EI95, EI99, and EPS in which the number of impact categories is pre-defined, EDIP allows an unlimited number of impact categories to be assessed. The characterized impact potentials from other methods can be further assessed using EDIP's normalization and weighting procedures.

Even though EDIP considers environmental impact, resource consumption and impacts on the working environment as the safeguard subjects, a further aggregation of the scores evaluated for these three safeguard subjects is not carried out because a valid method to lump these scores together is not yet determined. The recommendation for EDIP interpretation is based on the impact potential profiles of each of the individual main groups. Hence, EDIP does not offer a true "single score" that combines all three safeguard subjects. However, a few assessments using EDIP are implemented by assessing only one safeguard subject, i.e., the environmental impact or the resource consumption (as in SimaPro Software, Nilsson 2001; Yang and Nielson 2001; Dreyer et al. 2003). These EDIP implementations allow the comparison of environmental contributions thus still serving the LCA purposes.

EDIP also shares limitations similar to those of EI95, EI99, and EPS in terms of carrying a degree of subjectivity in the weighting procedure. In EDIP, subjective judgment is reflected because the determination of the future target emission levels is based on value choices (mostly Danish emission policy). The EDIP documentation provides the scientific background using information from Denmark and Europe specifically. However, there are a few attempts to evaluate a group of normalization references and weighting factors for other regions such as China (Yang and Nielson 2001; Lin et al. 2005).

2.3.5 IMPACT2002+ (Jolliet et al. 2003)

The IMPACT2002+ LCIA method, which was developed in Switzerland, was a modification of EI99. IMPACT2002+ shares several features with EI99, one of which is the aggregating of impact categories into a smaller number of groups for weighting purpose (safeguard subjects). IMPACT2000+ has grouped the total of 14 midpoint impact categories into four damage categories. The term "damage category" used in this method and EI99 is as same as the term "safeguard subject" used in EPS and "indicator value" used in EDIP. However, while EI99 and EDIP have three damage categories, IMPACT2002+ has four and EPS has five. The damage categories considered in IMPACT2002+ are human health, ecosystem quality, climate change, and resources. Climate change is the damage category that has been added to those in EI99. The structure of IMPACT2002+ is presented in Figure 2.6.

IMPACT2002+ uses most of the characterization factors obtained from EI99 and other sources such as CML (Center of Environmental Science at Leiden University, the Netherlands) and EcoInvent Database. IMPACT2002+ also uses newly developed methodologies to evaluate the characterization factors for human toxicity and eco-toxicity. The Human Damage Factors (HDF) were calculated for carcinogens and non-carcinogens, employing intake fractions, estimates of dose-response slope factors. Both human and eco-toxicity damage factors are based on mean responses of the receptor. The risk assessment models were used to determine the HDFs for several thousand chemicals. Generic factors were calculated at the Western Europe continent level offering the spatial differentiation for 50 watershed and air cells. Damages in IMPACT 2002+ are expressed in terms of Disability Adjusted Life Years (DALYs), Potentially Disappeared Fraction (PDF), kg CO_2, and MJ for grouping the impact categories into the damage categories of human health, ecosystem quality, climate change, and resources, respectively.

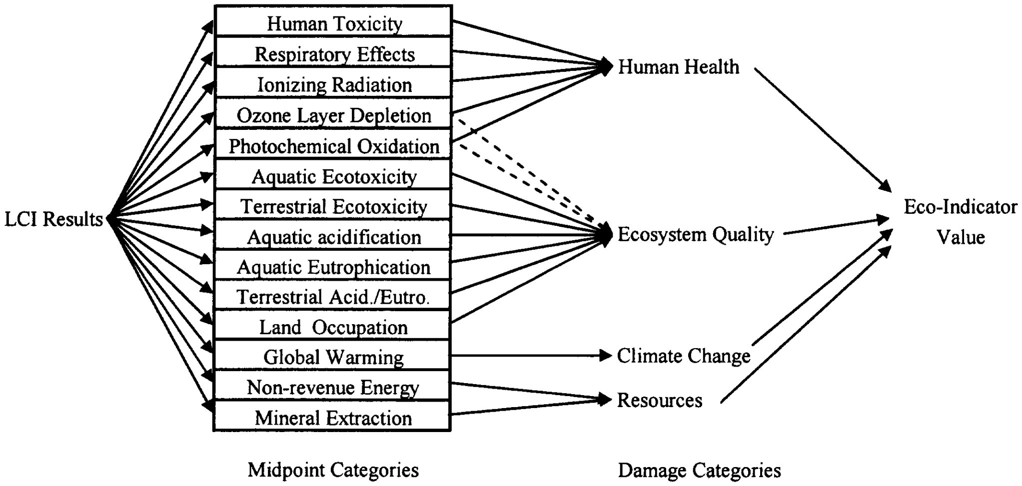

Figure 2.6 Structure of IMPACT2002+ method.
(Source: Jolliet et al. 2003)

For the normalization, the developer calculated the normalization factors by using the ratio of impact per unit of emission divided by the total impact of all substances of the specific category, per person per year. And the calculated normalization factors are 0.0077 DALY/person/year, 4,650 PDF.m^2.year/person/year, 9,950 kg CO_2/person/year, and 152,000 MJ/person/year for the damage categories of human health, ecosystem quality, climate change, and resources, respectively.

The weighting factors for aggregating the damage categories to obtain a single score were not evaluated in IMPACT2002+. The developers suggested that the interpretation of the LCA results should be made separately for those impact categories. However, it is the choice of the users to determine the damage factors by themselves should an aggregation be needed. The default weighting factors of one, which means each damage category is as important as the others, are used in SimaPro (Pre' 2006).

IMPACT2002+ has major advantages over EI99 in terms of providing the spatial variation of human toxicity and eco-toxicity impacts and global climate change is emphasized as a damage category. However, the weakness, inherent also in EI99, is the treatment of the weighting process.

2.3.6 Life Cycle Impact Assessment Method Based on Endpoint Modeling (LIME)

(Hayashi et al. 2004; Itsubo et al. 2004; Hayashi et al. 2006)

The LIME approach assembles the data from LCI into a single score expressed in two forms; a monetary value in Japanese Yen and a dimensionless index. The LIME combines both midpoint and endpoint assessments into its evaluation process before grouping the endpoint categories into four safeguard subjects; human health, social assets, biodiversity and primary production. Human health and social assets are related to human welfare while biodiversity and primary production relate to ecosystem welfare. The structure of LIME is presented in Figure 2.7.

LIME uses a set of Japanese-based LCI data for implementation in LCI process. The characterization of LCI results are made using characterization factors from several sources of information as well as from newly developed Japanese-based characterization factors for some local impact categories (Hayashi et al. 2004). The midpoint impact categories are then re-grouped into different endpoint categories according to their incurred damage using damage assessments. Then the quantities in endpoint categories are distributed to four different safeguard subjects using damage factors obtained from damage analyses. Damages are expressed in terms of Disability Adjusted Life Years (DALYs), Japanese Yen, Extinct Number of Species (EINES), and Dry Ton for the safeguard subjects of human health, social assets, ecosystem, and primary production, respectively. An LCA single score in terms of monetary value can be obtained by multiplying the damages in all safeguard subjects by the corresponding "monetary weighting factors". The monetary weighting factors were estimated using the results from the interview for determination of the willingness to pay (WTP) to prevent the damages from about 400 Japanese household samplings. Table 2.5 shows the monetary weighting factors of all safeguard subjects obtained using this approach. And the monetary single score can be formulated using Equation (2.7).

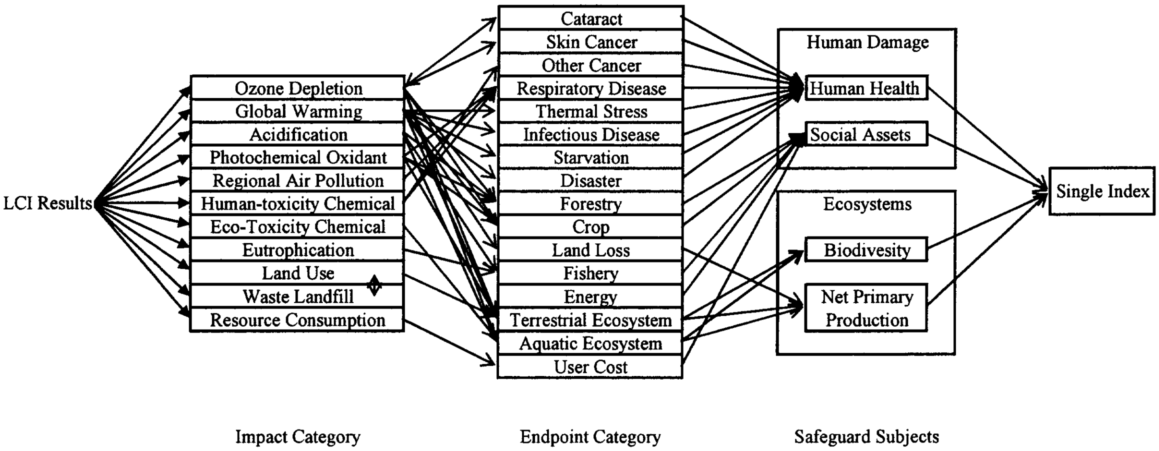

Figure 2.7 Structure of LIME method.
(Source: Hayashi et al. 2006)

Table 2.5 Monetary Weighting Factors in LIME
(Source: Itsubo et al. 2004)

Safeguard Subject	Monetary Valuation	
	Unit of Damage	Monetary Weighting Factor (Japanese Yen/Unit)
Human Health	1 DALY	9.76×10^6
Social Assets	10,000 Japanese Yen	1.00×10^4
Biodiversity	1 Species loss	4.80×10^{12}
Net Primary Production	1 Ton	2.02×10^4

$$I_1 = \sum_e \sum_s \left(Inv._s \times Damage\ Factor_{s,e} \times Weighting\ Factor_{1,e} \right) \qquad (2.7)$$

Where:

I_1	a monetary LCA index (Japanese yen)
$Inv._s$	a result of LCI of substance *s* (kg)
$Damage\ Factor_{s,e}$	a damage factor (damage/kg) of environmental loading substance *s* for safeguard subject *e* (damage amount/kg) $= \sum_t Damage\ Function_{s,e,t}$
$Damage\ Function_{s,e,t}$	a damage amount (damage amount/kg) incurred by a safeguard subject *e* through a specific process of environmental impact *t* caused by the loading of one unit of an environmental loading of substance *s*
$Weighting\ Factor_{1,e}$	an amount of monetary weighting factor for one unit of damage to a safeguard subject (yen/unit damage amount).

A national level WTP for a safeguard subject could be obtained by multiplying the household level WTP by the total number of households in Japan. The "dimensionless weighting factors" could then be obtained by normalizing the national level WTPs by the damages at the national level, which was calculated using the national pollution release inventory. The 2002 dimensionless weighting factors were 0.31, 0.21, 0.23, and 0.26 for safeguard subjects of human health, social assets, primary production, and biodiversity, respectively. And the dimensionless single score can be formulated using Equation (2.8).

$$I_2 = \sum_e \sum_s \left(\frac{Inv._s \times Damage\ Factor_{s,e}}{Normalization\ Value_e} \times Weighting_{2,e} \right) \tag{2.8}$$

Where:

I_2	a dimensionless LCA index
$Weighting\ Factor_{2,e}$	a dimensionless weighting factor of an annual damage amount of a safeguard subject
$Normalization\ Value_e$	annual amount of damage incurred by safeguard subject e $= \sum_s \left(Annual\ Env.\ Loading \times Damage\ Factor_{s,e} \right)$
$Annual\ Env.\ Loading_s$	annual amount of environmental loading (kilograms) of environmental loading substance s in Japan.

Like EPS, a feature of LIME is to provide a universal indicator (currency unit) that is understandable and more tangible to the decision makers and other interested audiences. The social costs of preventing environmental damage are also included in the evaluation. One notable feature of LIME is the combination of midpoint and endpoint impact categories. The different assessments used among those two types of impact categories are well defined. Several rigorous approaches were used in the evaluation of characterization factors and damage factors (Hayashi et al. 2004)

With regard to limitations, value judgment still exists in the determination of the weighting factors (Itsubo and Inaba 2003). Other weaknesses are the incapability of dealing with temporal and spatial variations. LIME was based on a Japanese LCI database and environmental conditions (Narita et al. 2004). As of now the LIME software is only available in Japanese, make it difficult for other LCA practitioners to study. An application of LIME to other parts of the world would require a significant amount of time and resources to update the database library, especially when LIME relies heavily on prices and costs which are fluctuating.

2.3.7 Summary and Opportunities for Increasing LCIA Sophistication

Six LCIA methods yielding single scores have been reviewed in the previous subsections. Some key properties and characteristics of these LCIA methods are summarized in Table 2.6. An LCA/LCIA single score is practical and applicable. It is useful for internal use to make a comparison among products, services, or materials by identifying the overall superiority or ranking of product or service alternatives, which is a benefit for decision-making. In terms of the superiority among choices, use of a single score is more explicit than using multiple scores of every impact category. Furthermore, LCA Single scores provide a way for businesses/organizations to communicate with their customers and audience about the sustainability characteristics of their products and services.

Although the available LCIA methods reviewed here have merit in terms of a presenting single score, there are still some disadvantages that can be identified. These disadvantages, drawn from the literature review, are:

Table 2.6 A Comparison of Features of Six LCIA Methods

Feature	Eco-Indicator 95 (EI95)	Eco-Indicator 99 (EI99)	Environmental Strategies System (EPS)
Developer	The Netherlands PRe' Consultants National Reuse of Waste Research Programme (NOH) Agency for Energy and the Environment (NOVEM) National Institute of Public Health and Environmental Protection (RIVM)	The Netherlands PRe' Consultants National Institute of Public Health and Environmental Protection (RIVM)	Sweden Centre for the Environmental Assessment of Products and Material Systems Chalmers University of Technology National Board for Technical and Industrial Development The Swedish Waste Research Board
Software tool	SimaPro 7	SimaPro 7	SimaPro 7, EPS 2000 Design System
Goal of the method	Environmentally-aware design of products	Environmentally-aware design of products	Increase of total welfare of products and processes
LCIA steps			
Classification	Yes	Yes	Yes
Characterization	Yes	Yes	Yes
Normalization	Yes	Yes	No
Grouping	Yes	Yes	Yes
Weighting	Yes	Yes	Yes
Evaluation and reporting	Yes	Yes	Yes
Method indicator and unit	Eco-Indicator 95 indicator value, point	Eco-Indicator 99 value, millipoint	Monetary value, Environmental Load Unit (ELU) -equivalent to Euro currency
Weighting principle	Distance-to-target, political weighting	Damage-oriented, panel expert	Actual price and willingness-to-pay to avoid damages
Evaluation basis	Relative impacts	Potential effects	Actual effects or risks
Site specific	No	No	No
Time, specific year	Varies on impact categories	No	2000
Information needed	Small	Moderate	High

Feature	Environmental Design for Industrial Products (EDIP)	IMPACT2002+	Life Cycle Impact Assessment Method Based on Endpoint Modeling (LIME)
Developer	Denmark Technical University of Denmark The Confederation of Danish Industries and the Danish EPA	Switzerland Swiss Federal Institute of Technology Lausanne (EPFL)	Japan National Institute of Advanced Industrial Science and Technology
Software tool	SimaPro 7, EDIP Software Tool (Beta)	SimaPro 7	Software Tool in Japanese
Goal of the method	Reduction of impacts associated with industrial product designs	Combine midpoint/endpoint approach	Combine midpoint/endpoint approach
LCIA steps			
Classification	Yes	Yes	Yes
Characterization	Yes	Yes	Yes
Normalization	Yes	Yes	Yes
Grouping	Yes	Yes	Yes
Weighting	Yes for weighting across impacts in main groups, No weighting across main groups	Yes	Yes Monetary/dimensionless indexes
Evaluation and reporting	Yes	Yes	Yes
Method indicator and unit	Impact potential, person-equivalent	UMPACT value, millipoint	Monetary value- Japanese yen and LIME value -dimensionless
Weighting principle	Distance-to-target, separate weighting for environmental impacts, resource consumption, and working environment	Damage-oriented, non-weighted	Conjoint analysis- value judgement
Evaluation basis	Relative impacts	Potential effects	Potential effects
Site specific	Yes for some impact categories	No	No
Time, specific year	1990 and 2000	No	Yes-varies
Information needed	Moderate	Moderate	High

1. Subjectivity of weighing procedure

Even though the sophisticated LCIA methods are capable of a full stage implementation, these methods, as well as the basic weighting techniques, are still not free from subjective value judgment. The subjectivity is present in the weighting factors where they are derived from value systems such as expert panels and governmental policies. These value-based weighing factors are controversial and may decrease the credibility or broad applicability of the entire LCA process (Udo de Haes 2000; Hertwich and Hammitt 2001). Because LCIA methods are complicated, the subjectiveness of the weighting procedure may be less obvious and may be overlooked by the audiences.

2. Weighting factors should be flexible

Even though the weighting procedure is based on either governmental policies or expert panels' judgment, the weights logically reflect the perceived seriousness of the environmental impacts at that time and place. The degree of seriousness is temporal and spatial according to several factors, e.g., the accomplishment of mitigation measures, better environmental conditions due to natural self-purification, new technology, and further exploration (of natural resources) or improved scientific understanding. Finnveden (2000) and Hellweg et al. (2003) also pointed out in a similar way, that current impacts should be weighted differently than the future impacts. Because weighting of environmental impacts is flexible, their use should be implemented cautiously and updated regularly.

3. Need of metrics/indicators relative to sustainability

In LCA, not only are environmental impacts measured, but they should be analyzed and interpreted in a meaningful way. Because LCA may used both internally and externally, its use as a communication tool between businesses and their external stakeholders will be more acceptable when the results are understandable and methods are transparent. The LCIA methods reviewed here have defined their own indicator units (Table 2.7). These units, i.e., point or millipoint for EI95, EI99, IMPACT2002+, and LIME, ELU for EPS, and person-equivalent for EDIP, are difficult to understand to the external stakeholders in terms of the environmental context. Among the units presented here, only the EPS's ELU (equivalent to Euro currency) and the LIME's damage index (in Japanese yen) might be considered as non-abstract and universal.

Table 2.7 Units for Single score of Six LCIA Methods

LCIA Method	Unit
EI95	Point, millipoint
EI99	Millipoint
EPS	ELU (Environmental load unit) - equivalent to Euro
EDIP	Person-equivalent
IMPACT2002+	Millipoint
LIME	Japanese yen and millipoints

These environmental-related indicators offer a way for comparative assessment of products and services using the same LCIA method. They identify the superiority among products or services by the "less is better" approach: the one posing lower environmental burdens is the one with better environmental performance. However, the precautionary principle, where the preventive action should be taken before the environmental damage can be measured, is not satisfied because the seriousness of the environmental impacts posed by environmental emissions and resource consumptions of the products/services is not identified. It is also difficult for external audiences to understand the possible physical meaning of these indicators. Heijungs (2005) discussed the fact that units used in LCA today are not universal and not common to external audiences. Therefore, indicators that may fulfill this gap are the ones that are more universal and more easily related to sustainability.

Recently, the concept of sustainable development has been more widely considered as a primary value for businesses (Schwarz et al. 2002). Consequently, the sustainability concept has been put into practice by identifying the indicators or metrics for measuring environmental performance as related to sustainability. And since the sustainability concept consists of three dimensions (or a triple bottom line: economic, environmental, and social equity), therefore, environmental performance metrics related to sustainability levels or sustainability metrics can be defined as the metrics that are designed to consolidate key measures of environmental, economic, and social performance (Schwarz et al. 2002). Examples of frameworks and approaches that attempt to standardize an environmental performance evaluation related to sustainability are Eco-Efficiency (WBCSD 2000), GRI's Sustainability Report (GRI 2002), ISO 14031 (ISO 1999), Ecological Footprint (Wackernagel et al. 1999) (the review of these frameworks and approaches is presented in Appendix A). Generally, these frameworks/approaches provide the guidelines for how to arrange an environmental report. They also present recommended lists of environmental performance indicators related to the concept of sustainable development. However, these frameworks and approaches do not generally provide the step-by-step procedures in detail. In the context of LCA, there are some LCA/LCIA study groups mentioning and introducing the other dimensions of sustainability (economic consideration and social equity) into their methodologies (Dickinson 1999; Bage and Samson 2003; Guinee et al. 2004; Stewart and

Jolliet 2004; Udo de Haes et al. 2004; Hunkeler and Rebitzer 2005; Weidama 2005; Labuschagne and Brent 2006). Even though it is just an early stage, but it shows the direction of how to develop more advanced and applicable LCA/LCIA methodologies.

Environmental performance indicators can be expressed as absolute or relative indicators, as indexed indicators (percentage with respect to total), as aggregated indicators (quantities of the same unit are summed), and weighted evaluations (conversion factors are used to depict quantities of varying importance) (ISO 1999). The approach that is gaining more publicity currently is the eco-efficiency concept proposed by the World Business Council for Sustainable Development (WBCSD) (Schmidheiny 1992). The WBCSD states that a basic business contribution to sustainable development is eco-efficiency, which is (WBCSD 2000):

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

According to Raynolds (1997), a basic form of eco-efficiency environmental performance indicator is the link between resource inputs and/or pollution outputs and units of products or services. In other words, higher eco-efficiency implies the creation of more goods and services with less resource use, and less waste and pollution. Eco-efficiency combines the two eco-dimensions of economy and ecology to relate product or service value to environmental influence. The measurement of eco-efficiency is widely practiced using the following ratio (WBCSD 2000):

$$\textit{Eco-Efficiency} = \frac{\textit{Product or Service Value}}{\textit{Environmental Influence}} \qquad (2.9)$$

The Eco-Efficiency framework relates environmental impact to the value added of a product or service. The comparative assessment between products/services may be implemented using eco-efficiency ratios.

4. Not all relevant environmental impacts are considered

An initial step in LCA is the goal and scope definition. At this step, the selection of impact categories that should be considered is made. An LCA should include all relevant environmental aspects and impacts. The impact category selection may also be made based on the availability of information in addition to the goal and scope definition. Most of LCIA methods in the review have a pre-defined and limited number of impact categories. The modification of this limitation requires a great number of changes. A method that is flexible in the selection of impact category would be more preferable in accordance with the LCA context. Especially for the damage-oriented LCIA methods, which aim to interpret an LCA in the form of damage indicators at the level of the societal concern (Jolliet et al. 2004).

5.) Other issues

Bare et al. (1999) addressed other issues for further LCIA development as follows:

- Environmental backgrounds and thresholds: the inclusion or the exclusion of environmental backgrounds and thresholds in LCIA;
- Uncertainty analysis: the necessity and practicality regarding the sophistication of the uncertainty analysis for model and data uncertainties;
- Midpoint and endpoint determinations: the advantages of incorporating the endpoint determination from environmental impact assessment (e.g., endpoint is the quantifying of fish kills and tree loss as opposed to emission of acidification potential of substances for the midpoint);

- Eco-toxicity: the difficulty of assessing and capturing the comprehensiveness of the environmental health impact category; and
- Role of risk assessment: the use of information and results from other supporting environmental analyses such as toxicological benchmarks from risk assessment.

Hunkeler and Biswas (2000) also added that in order to evolve LCA into a more objective index, the aspects that need to be addressed: 1) are based on quantifiable, investigator independent, systematic, life cycle data, 2) must be scalable, so that the results can be normalized to be representative within and across product lines, and 3) should combine environmental, technical, and market-based information.

Spatial differentiation in LCA is also necessary to strengthen the credibility of LCA. Spatial differentiation is gaining more attention among LCI/LCIA developers (Ross and Evans 2002; Scharnhorst et al. 2004; Udo de Haes et al. 2004; Heuvelmans et al. 2005; Hettelingh et al. 2005). Modern LCA should focus on the implementation of spatial differentiation in both LCI and LCIA. And also pointed out by Ciroth et al. (2004), the omission of uncertainty assessment may downgrade the usability of LCA. Sugiyama et al. (2005) recently presented a new approach to use standard statistics to consider the uncertainty in industrial-based LCI.

A recent survey conducted to probe the needs from LCA practitioners indicates that issues that need immediate attention within UNEP-SETAC Initiatives are the transparency of methodology, scientific confidence, scientific co-operation, and factors that are recommended for future LCA development (Stewart and Jolliet 2004).

In summary, a consensus and standard for LCIA methods as well as an objective weighting procedure is not available currently. Some governments have attempted to adopt their own standard methods supporting single scores, e.g., EI99 for the

Netherlands, the EPS for Sweden, EDIP for Denmark, IMPACT2002+ for Switzerland, and LIME for Japan, but none of these methods has yet been accepted as the one for worldwide use. Hence, there are opportunities for increasing the sophistication of LCIA by addressing the issues mentioned above. In addition, the attempts to develop a refined system for measuring environmental performance following the LCA framework are increasingly under attention since the introduction of the ISO 14040 series. Therefore, further contributions to LCA and LCIA development are surely needed.

Further research to advance the development of a single score LCIA may be designed on the basis of two criteria: ease of explanation to non-specialists and transparency for specialists (Vogtlander and Bijma 2000). The following section reviews a recently developed LCIA method, the Sustainability Target Method (STM) that is an attempt to meet these criteria, i.e., simplicity and transparency to all audiences.

As pointed out in the preceding literature review, the major weakness of current LCIA development is the weighting procedure. To bridge this gap, further research may be divided into two directions. The first one is to seek a weighting method that could be widely accepted as the standard, which is difficult. And the second one is to seek a means to aggregate the discrete environmental impacts without subjective weighting. The STM is being developed as an LCIA-based, non-weighting methodology that bases the aggregation on the sustainability concept rather than value-choice weighting process.

2.4 The Sustainability Target Method

2.4.1 The Development of the STM

The Sustainability Target Method (STM) is an environmental performance metric formulated by Lucent Technologies and developed through the collaboration of Lucent Technologies, Agere Systems, and the NJIT Multi-lifecycle Engineering Research Center (MERC) (Caudill et al. 2002). The STM can be used to evaluate the environmental impacts associated with business activities in terms of achieving a practical sustainability target. In the context of LCA, the STM can also be considered as an LCIA method that provides meaningful environmental performance indicators related to the sustainability of the product.

In general, the STM interprets different types of environmental impacts based on a single dimensionless indicator called Environmental Impact (EI), which is the environmental impact per unit production (Dickinson 1999; Mosovsky et al. 1999; Mosovsky et al. 2000; Dickinson et al. 2001; Mosovsky et al. 2001; Dickinson et al. 2002). EI provides the basis for calculating the indicators called Resource Productivity (RP) and Value Productivity (VP), which are relative indicators expressing the level of production and value provided, respectively, per unit of environmental impact. RP and VP provide a quantitative basis for comparing environmental impacts between products or processes. RP and VP are also used to calculate Eco-Efficiency (EE), which is a practical absolute indicator for sustainability. EE is essentially a product's economic contribution (percentage of GDP) divided by its environmental burden (percentage of carrying capacity). The carrying capacity is used as the baseline reference for the sustainability target in the STM.

The STM can be used to evaluate the environmental performance across the complete life cycle stages of a product because it is being developed according to the LCA framework. The key function of this method is to make a link between the economic value of the product and the portion of the environmental carrying capacity that the product consumes. Therefore, the STM accounts for two elements of sustainability: economic and environmental considerations. The third element, social considerations, has not been explicitly incorporated and will be explored in further developments (Caudill et al. 2002). The details of the STM methodology are presented in Appendix B.

2.4.2 Characteristics of the STM

Combining economic and environmental evaluations

The STM takes into account economic considerations by application of the economic value of a product or service. The application of the economic value may also be extended to social considerations, which may be quantified in terms of monetary value of social perception of the willingness-to-pay to protect or prevent environmental damages and social value of the product or service.

Dealing with temporal and spatial variations

The STM attempts to take into account the temporal and spatial variations. Mostly, the aspects and life cycle stages of a system are temporal and spatial variations. The impacts demonstrate spatial variation when local or regional scales are considered and there are temporal variations when the aspects are not influenced simultaneously. Temporal and spatial variations are dealt with in the STM calculations where EI is the aggregation of EI from several aspects or life cycle stages of a system. The overall EI is the sum of EI from different life cycle stages of the product, which are temporal and spatial distributions.

Measuring the environmental impacts related to sustainability

In the STM, the significance of the environmental impact is not only identified, it is also expressed as a group of indicators related to the sustainability target. In other words, the sustainability target is used as the ruler to measure environmental performance of systems. This makes the STM a tool for absolute measure, with which the environmental performance or the level of environmental stewardship of a product can be identified without any further comparison with others. The evaluation of environmental impacts as related to the sustainability target may also be considered as an approach toward the precautionary principle in the examination of alternatives in the decision-making process.

Assessing environmental impacts without weighting procedure

The weighting procedure is used traditionally in LCIA to identify the relative importance among various impact categories. With the STM, weighting is avoided because all of the environmental impacts are expressed based on the same scale, which is the earth's carrying capacity. Exceeding the earth's carrying capacity is unsustainable regardless of the specific impact category. The STM is considered as a combined midpoint and endpoint assessment, i.e., the environmental damage is assessed relative to the sustainability target.

Being flexible for the selection of impact categories

The impact assessment of the STM is simple and straightforward. Similar to the EDIP, the impact categories are not attached to the calculation processes; therefore, the STM is flexible and not limited to the selection of impact categories. Users can choose the impact categories that suit their use of the assessment. The characterized impact potentials from other methods can be further assessed in the STM.

Providing environmental condition assessment

An environmental condition indicator (ECI) provides the information on environmental health and conditions at different scales. This type of indicator links specific business activities or emissions to environmental impacts, i.e., establishes a relationship between pollutants and impacts. According to the National Academy of Engineering (NAE 1999), this type of indicator is of greatest interest to industry and stakeholders because it may be used to estimate environmental performance toward the sustainability of human activities. Indicators in the STM may be considered as an ECI because they are derived from environmental conditions. In other words, the sustainability level is the ultimate target for improvement or maintenance of environmental conditions.

2.4.3 The Importance of the Carrying Capacity to the STM

The STM defines the economic carrying capacity (ECC) as the natural carrying capacity (NCC) minus the natural burden (NB)[1]. The ECC is the portion available to industry and commerce to support the needs of society (Dickinson et al. 2001). Since ECC is the one that is associated with economic value, therefore, only the ECC is used as the reference in the STM calculations. The STM is formulated from a business perspective; the term "carrying capacity" hereafter refers to the economic carrying capacity.

Carrying capacity is important to the STM because it is used to calculate the key references, i.e., the Impact Reference Levels (I_R) and the Aspect Reference Levels (A_R) (see Appendix B). The utility of the STM is based heavily on the soundness of the

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[1] Natural carrying capacity is the capacity of the earth to absorb the environmental burdens caused by both natural and anthropogenic sources. The economic carrying capacity is the portion of the burdens that is associated with economic value (caused by anthropogenic sources) while the natural burden is the portion that is caused by natural sources and is not associated with economic value.
(Economic carrying capacity = Natural carrying capacity – Natural burden)

methodologies and approaches used in carrying capacity estimates. Therefore, it is important to develop estimates of carrying capacity based upon the best available scientific evidence and supporting documents. Scientifically-rigorous carrying capacity estimates will strengthen the advantages of the STM over other LCIA methods. This research is motivated by the need to create scientifically-based methodologies to estimate carrying capacity that are as sound as possible. The following section reviews topics relevant to carrying capacity development.

2.5 Impact Categories and Methodologies for Carrying Capacity Estimates

2.5.1 Impact Categories

The number of impact categories to be included in an implementation of environmental performance evaluation is limited by practicality. A proposal by SETAC divides the impact categories into input related and output related categories (Udo de Haes et al. 1999a; 1999b). With the exception of "land use", this list has also been adopted by the ISO for the ISO 14040 series (ISO 2001).

- Input Related Categories
    - Category 1: Extraction of abiotic resources
        - Subcategory: Extraction of Deposits
        - Subcategory: Extraction of Funds
        - Subcategory: Extraction of Flow resources
    - Category 2: Extraction of biotic resources
    - Category 3: Land use
        - Subcategory: Increase of land competition
        - Subcategory: Degradation of life support functions
        - Subcategory: Bio-diversity degradation
- Output Related Categories
    - Category 1: Climate change
    - Category 2: Stratospheric ozone depletion
    - Category 3: Human toxicity
    - Category 4: Eco-toxicity

Category 5: Photo-oxidant formation
Category 6: Acidification
Category 7: Nutrification (Eutrophication)

This list is not exhaustive and other impact categories may be important in some situations. These other impact categories may include noise, odor, and others. In LCA practice, selection of impact categories depends on the system boundaries and the specification of the project. However, an LCIA method must cover the common impact categories so it can be considered as a generic method. To keep the overall structure of impact categories within the same system, the categories can be divided into subcategories. Then the aggregation can also be performed at the subcategory level.

In general, impact categories can be divided into homogeneous and non-homogeneous impact categories (Assies 1998). A homogeneous impact category is an impact that is incurred by several burdens and these burdens can be interchangeable (or have trade-off capability) within this impact category in the sense of environmental damage. For example, the global warming impact is a homogeneous impact category because it is caused exclusively by emissions of CO_2, CH_4, and other greenhouse gases. These greenhouse gases are interchangeable and can be aggregated within the impact. Examples of other homogeneous impact categories are climate change, stratospheric ozone depletion, and acidification. By contrast, a non-homogeneous impact category is an impact caused by a specific burden. For example, human toxicity impact is a non-homogeneous impact category because the adverse impact on human health is caused by different toxic substances and these substances are not substitutable in terms of human exposure. This is because different toxic chemicals affect different organs and systems and it is not possible to calculate the synergistic and additive effect on human health caused by all toxic chemicals. As a result, the aggregation or trade-off in terms of toxicity potential among toxic chemicals is not valid (Udo de Haes et al. 1999a).

The CML (Center of Environmental Science, or Centrum voor Milieukunde, at Leiden University, the Netherlands) divides environmental impacts into three groups: baseline impact category, study-specific impact category, and other impact categories (Guinee 2001). The baseline impacts, which are mandatory, are presented in Table 2.8 along with the impact categories recommended by SETAC and the ones used in the six LCIA methods reviewed in Section 2.3. Study-specific impacts include radiation, odor, noise, waste heat, and casualties. Other impacts may include the depletion of biotic resources.

A survey in 2004 revealed the impact categories that are most significant in the opinion of LCA practitioners (Stewart and Jolliet 2004). According to this survey, impact categories could be prioritized into three levels, i.e., the "Required" impact categories which are the most significant, the "Nice to Know" impact categories, which are moderately significant, and the "Low Priority", which are not significant. Table 2.9 shows the list of impact categories distributed within these three levels.

Impact categories in LCA are defined on the basis of general environmental concerns. The data aggregation between burdens in a homogeneous impact category is made in the impact assessment phase through the characterization step using "equivalency factors" or "characterization factors" in the LCIA. Some impact categories, such as acidification and eutrophication, may be divided into subcategories to take account of differences in the characteristics/means of pollutants that contribute to the same impact.

Table 2.8 Impact Categories Used in Other Studies and Methods

Default impact categories	SETAC	CML Baseline Impacts	EI95	EI99 IMPACT	EPS	EDIP	LIME
Input related categories							
Resource depletion	X	X		X	X	X	X
Land use and land occupation	X	X		X	X		X
Output related categories							
Global warming	X	X	X	X	X	X	X
Stratospheric ozone layer depletion	X	X	X	X	X	X	X
Photochemical ozone formation	X	X	X	X	X	X	X
Toxic to human health							
General toxic substances	X	X			X	X	X
Carcinogens			X	X			X
Substances causing winter smog			X				
Pesticides			X	X			
Heavy metals			X				
Substances damaging respiratory system				X			X
Radiation				X			
Toxic to ecosystem							
Acidification	X	X	X	X	X	X	X
Eutrophication	X	X	X	X	X	X	X
Ecosystem quality	X	X		X	X	X	X
Production							
Crop and food					X		X
Natural					X		X
Irrigation water					X		X
Solid wastes					X		X
Noise pollution					X		
Impacts on working environment							
Cancer						X	
Allergy						X	
Hearing impairment						X	
Accident						X	
Damage to reproductive system						X	
Damage to nervous system						X	
Musculoskeletal injuries						X	

Table 2.9 Significance of Impact Categories
(Source: Stewart and Jolliet 2004)

Required	Nice to Know	Low Priority
Climate Change Ozone Depletion Habitat loss as a result of deliberate actions Human Toxicity Eco-Toxicity Acidification and Eutrophication Photo-Oxidants Extraction of Minerals Energy from Fossil Fuels Nuclear Radiation Water Usage	Salinisation Erosion Soil Depletion Habitat loss as a result of indirect actions Noise Use of GMOs	Health of workers Safety Landscape Extraction of biotic resources

2.5.2 Survey of Potential Methodologies for Carrying Capacity Estimates

The Ecological Footprint concept defines carrying capacity as "the maximum rates of resource harvesting and waste generation that can be sustained indefinitely without progressively impairing the productivity and functional integrity of relevant ecosystems wherever the latter may be located" (Rees 1996; Wackernagel et al. 1999). Ecological Footprint uses area requirements (footprint area) for environmental aspects as the aggregated indicator to measure sustainability. The environmental aspects such as human consumption are translated into areas of productive land that are required to provide resources and to assimilate waste products. The area requirement of the ecological footprint is expressed on the basis of per capita per year. When the footprint is greater than the total area where the people live, the sustainability criteria are not met. Use of area requirements as an indicator is a simple but effective way to communicate with all audiences.

The ecological footprint is among the most popular metrics for environmental reports at the national level. The advantage of the ecological footprint is that the measure is translated in terms of an aggregated indicator. By using the area requirements, the comparative assessment for sustainability between nations/demographic areas can be carried out. The simple criteria of sustainability are twofold: finite land area (bio-capacity) and population density, i.e., the carrying capacity is the basis for demographic accounting. The carrying capacity (in terms of land area) is exceeded when the footprint of a nation is greater than its actual land area. In essence, the Ecological Footprint uses productivity and functional integrity as measures of damage to the ecosystem. (More discussion on Ecological Footprint is in Appendix A)

However, in this research "carrying capacity" is defined as the capacity of the earth to absorb or to tolerate potentially stressful burdens imparted at various scales and locations, that is, to accommodate the stresses without showing permanent damage (Yossapol et al. 2002). In other words, a carrying capacity is the maximum magnitude of an environmental burden that causes no permanent damage to the environment. This definition is in accordance with the sustainability concept where a sustainable impact may well cause certain environmental effects, but not any effects which endanger our own needs or those of future generations in the long term (United Nations' World Commission for Environment and Development 1987).

Hauschild and Wenzel (1998) also define the carrying capacity in a similar manner to the one in this research, "… the magnitude of the impact which produces no detectable effects, either acute or chronic, in the recipient", while Catton (1986 as cited in Rees 1996) defines the carrying capacity of an environment as the maximum persistently supportable load. Carrying capacity is assessed for each individual type of environmental impact and for the associated burdens. For example, the carrying capacity for global warming impact may be determined based on the level of greenhouse gas (GHG) emissions that will allow stabilization of global temperature.

The first point of concern about the carrying capacity estimate is the level in the environmental mechanisms at which it can be defined as "sustainable". A carrying capacity is considered individually for an associated impact. Hence, a practical definition of the "sustainable level" or "carrying capacity" for one impact category may be different from other impact categories. Until recently, the concept of environmental carrying capacity has not been widely discussed by other environmental or LCA studies.

The concept of sustainable level can be substituted by the term “critical levels”, the levels of environmental burdens that maintain sustainable conditions of the environmental impact of concern. In some circumstances where scientifically-based critical levels are not feasible (e.g., there are still some necessary uses of ozone depleting substances), the more subjective but also more realistic “target levels” may be used instead. The target levels are subjective in terms of including political judgments and other uncertainty factors. In EI95, “target values” were defined as the environmental impacts (emissions or releases) that cause impairment, but only to an acceptable degree (see also Subsection 2.3.1).

Another example of subjective judgment associated with the use of “target levels” is the selection between midpoint and endpoint targets. Midpoints are considered to be links in the cause-effect chain (environmental mechanism) of an impact category, prior to endpoints, at which characterization factors or indicators can be derived to reflect the relative importance of emissions or extractions (Bare et al. 2000). In midpoint approaches, cause-effect information is generally presented in the form of qualitative relationships, statistics, and numbers reported in the literature while the endpoint approaches do not need to deal separately with the environmental relevance of the category indicators (Bare et al. 2000). The selection of either midpoint or endpoint approaches, including the impact indicators, plays an important role in subjective judgments of the evaluation of the carrying capacity. However, a more important factor for the evaluation of carrying capacity is the availability of the required data and scientifically-rigorous approaches. In some cases, limited or inaccessible data may force the carrying capacity to be evaluated by a less sophisticated approach.

This subsection reviews the works by Goedkoop (1998) and Hauschild and Wenzel (1998), which adopted methodologies to determine the European target values for EI95 and Denmark/Europe/Global carrying capacity for EDIP. These two concepts are similar to the carrying capacity for the STM. A preliminary set of U.S. carrying capacity values that were modified from EI95's target values by Dickinson et al. (2001) is also reviewed.

1. EI95's Target Values (Goedkoop 1998)

Target values for Europe were evaluated by dividing the "normalization values" by the "reduction factors". The normalization values were the existing (1990) anthropogenic emission of pollutants normalized to a per capita basis. Therefore, emission of a pollutant for Europe can be scaled up from an emission data of country A by:

$$\textit{European emission (kg/yr)} = \frac{\textit{Emission from Country A} \times \textit{European pop.}}{\textit{Country A population}} \qquad (2.10)$$

When the emission data were missing for some impacts, the extrapolation was made using the country's energy consumption. This was calculated on the assumption that the energy consumption reflected the emission pattern. The emissions were normalized and aggregated within the same impact categories: the greenhouse effect, ozone layer depletion, acidification, eutrophication, heavy metals, carcinogens, winter smog, summer smog, and pesticides.

To determine the reduction factors, the degree of impairment varied among impact categories. Therefore a reduction factor was judged according to the degree by which a policy target emission rate was exceeded (distance-to-target). The reduction factors ranged from 2.5 for a less serious condition impact category to 100 for a very serious condition impact. These reduction factors were judged based on policy on target emission level information from governmental documents such as the Dutch National Institute for Public Health and Environmental Hygiene (RIVM) and other sources of information.

There were several criteria for assigning the reduction factors (see also Table 2.2), some of which were based on less sophisticated approaches. Even though there were criteria for the determination, the reduction factors for some impact categories seem to have been subjectively selected. For example:

- The criterion for greenhouse effect was defined as the emission that caused 5% impairment of the ecosystems and the increase of mean global temperature of 0.1° C per decade. By considering that the current emission (at that time) was not exceeding the threshold levels, then the reduction factor of 2.5 was simply selected.

- Lead was used as the reference chemical for heavy metal in the air impact category. Lead concentration in the air in Europe was considered to be less than the harmful level for exposure to children. However, a reduction factor of five was selected for lead (as well as other heavy metals) in the atmosphere. The reduction factor of five was also used for an emission target level of cadmium, the reference substance for heavy metals, in water.

- The reduction factor for summer smog was selected by determining the damage to crops. The acceptable level for ozone concentration was 0.03 ppm under which crop damage would not occur. The level of the summer smog ozone would be reduced by 90% from the observed 0.3 ppm to 0.03 ppm, which was the target level. With this ozone reduction rate, VOCs and NO_x, which are the primary photochemical smog precursors, must be reduced by 60-70%. Therefore, a reduction factor of 2.5 was selected.

Goedkoop (1998) indicated that the difficulty in determining a more sophisticated set of reduction factors in EI95 lead to the building of a newer model, EI99. Table 2.10 presents the calculation of EI95 target values. The normalization values presented in this table were the estimated European emission of pollutants in each impact.

Table 2.10 Normalization Values, Reduction Factors, and Target Values EI95
(Source: Goedkoop 1998)

Impact	Scale	Normalization values	Reduction factor	Target value	Unit[1]
Greenhouse effect	Europe	6.5E+12	2.5	2.60E+12	GWP kg/yr
Ozone layer depletion	Europe	4.6E+08	100	4.60E+06	ODP kg/yr
Acidification	Europe	5.6E+10	10	5.60E+09	AP kg/yr
Eutrophication	Europe	1.9E+10	5	3.80E+09	NP kg/yr
Heavy metals	Europe	2.7E+07	5	5.40E+06	Pb-eq kg/yr
Carcinogens	Europe	5.4E+06	10	5.40E+05	PAH-eq kg/yr
Winter smog	Europe	4.7E+10	5	9.40E+09	SO_2-eq kg/yr
Summer smog	Europe	8.9E+09	2.5	3.56E+09	POCP kg/yr
Pesticides	Europe	4.8E+08	25	1.92E+07	Active ingredient kg/yr

The limitations in applying these target values for the STM are 1) the target values were derived on an emission basis, which is different from the carrying capacity concept; 2) they lack scientific rigor in selecting reduction factors; 3) the target value relevant to resource availability is not included; and 4) they are Europe-based and therefore not easily adaptable to other parts of the world without additional data.

2. EDIP's Carrying Capacity (Hauschild and Wenzel 1998)

Hauschild and Wenzel (1998) evaluated a set of "environment's carrying capacity" as the reference potential impact to calculate normalized environmental potential (see also Subsection 2.3.4) for some common impact categories. The reference potential impact was calculated from the carrying capacity normalized by population (e.g., kgCO_2-eq/person/year in 1990).

In their work, the carrying capacity was evaluated from different sources and methods, e.g., the carrying capacity for the emission of CO_2 and other greenhouse gases was estimated from the absorption capacity of major sinks. They also suggested that a

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[1] GWP = Global warming potential (CO_2-eq.), ODP = Ozone depleting potential (CFC11-eq), AP = Acidification potential (SO_2-eq), NP = Nutrification potential (PO_4^{3-}-eq), PAH = Polyaromatic hydrocarbon (benzo[a]pyrene-eq), POCP = Photochemical ozone creation potential (Ethylene-eq).

carrying capacity might also be estimated on a pre-industrial basis, for example, by using a model to determine the deposition scenarios for acidification of forest systems in Scandinavia. The results of the modeling showed that the era that did not result in an exceedance of the critical load for any forest system would be the pre-industrial scenario from around 1890. The emission of acidifying substances at that time was around 5 % relative to that of 1990. The carrying capacity for the EU could then be derived using a 95% reduction of the 1990 emission rate. Table 2.11 summarizes the EDIP's carrying capacity for four impact categories that have a similar concept to carrying capacity in this dissertation research.

Table 2.11 EDIP's Carrying Capacity
(Source: Hauschild and Wenzel 1998)

Impact category	Method	Scale	Carrying capacity	Unit[1]
Global warming	Natural removal capacity	Global	2.0E+7	kt CO_2-eq/yr
Stratospheric ozone depletion	Natural removal capacity Sustainable level	Global Global	44 281	kt CFC-11-eq/yr kt CFC-11-eq/yr
Acidification	Interpolation from pre-damaged level Interpolation from pre-damaged level	Denmark Europe	62 2420	kt SO_2-eq/yr kt SO_2-eq/yr
Nutrient enrichment	Interpolation from pre-damaged level Interpolation from pre-damaged level	Denmark Denmark	29.6 0.57	kt N/yr kt P/yr

The carrying capacity for photochemical ozone formation impact was not determined in this work. However, Hauschild and Wenzel (1998) suggested two possible methods to determine carrying capacity: use of threshold levels and use of pre-industrial emissions. They also pointed out that the carrying capacity could be estimated for both VOCs and NO_X. For the estimate on a pre-industrial basis, the carrying capacity could be defined from the pre-industrial or natural emission scenario.

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[1] CO_2-eq is the same as GWP, CFC-11 is the same as ODP, and SO_2-eq is the same as AP.

The carrying capacity derived for EDIP is not as comprehensive as the one in EI95. In other words, the carrying capacity estimate for EDIP is damage-oriented while EI95 target values are based on the distance to emission target. Therefore, EDIP carrying capacity has to be determined individually because each impact category is associated with different burdens. The major advantage is that the damage-oriented approach fits well with the definition of the carrying capacity in the STM context. However, the carrying capacity for acidification and nutrient enrichment is based on a Denmark and European scale. Another limitation is this set of carrying capacity estimates excludes some key impact categories such as resource consumption and human health impact.

3. Modified EI95's Target Values for the U.S. (Dickinson et al. 2001)

EI95's target values for seven impact categories were modified by Dickinson et al. (2001) to estimate a U.S.-based carrying capacity. This preliminary U.S. carrying capacity was estimated by normalizing EI95's target values using the European land area. Then the calculated U.S. carrying capacity was simply obtained by multiplying the normalized EI95's target values by the U.S. land area. Because this set of carrying capacity estimates was calculated from EI95's target values, it shares the same limitations. This set of carrying capacity values was determined for the purpose of testing STM application in the early stages of STM development. Table 2.12 presents the preliminary U.S. carrying capacities based on EI95's target values. From this table, the EI95 target values for Europe were obtained from the ones presented in Table 2.10. A modified carrying capacity was obtained by multiplying the EI95 target value by the ratio of [9.36×10^6 $km^2/4.43\times10^6$ km^2], where the former figure is the European land area and the latter is that of the U.S.

Table 2.12 Preliminary U.S. Carrying Capacity Based on EI95's Target Values
(Source: Dickinson et al. 2001)

Impact category	EI95 Target Value (Europe)	Modified Carrying Capacity (U.S. Scale)	Unit
Acidification	5.6E+09	1.2E+10	AP kg/yr
Eutrophication	3.8E+09	8.0E+09	NP kg/yr
Heavy metals	5.4E+06	1.1E+07	Pb-eq kg/yr
Carcinogens	5.4E+05	1.1E+06	PAH-eq kg/yr
Winter smog	9.4E+09	2.0E+10	SO_2-eq kg/yr
Summer smog	3.6E+09	7.5E+09	POCP kg/yr
Pesticides	1.9E+07	4.1E+07	Active ingredient Kg/yr

2.5.3 Summary

As stated, the concept of environmental carrying capacity is rather new. Therefore, only a few studies focusing on the evaluation of the environmental absorption capacity are available. This section has provided a background and a review of the available information, approaches, and methodologies that can be used as the carrying capacity or used to evaluate the carrying capacity. The methodologies that have been reviewed are the works by Goedkoop (1998) for EI95, by Hauschild and Wenzel (1998) for EDIP, and by Dickinson et al. (2001), which was originally intended for the STM at the early development stage. In addition, other potential methodologies that may be used to evaluate carrying capacity for some impact categories are also reviewed.

There are some limitations in applying the EI95's target values for the STM. First, the target values were derived on an emission basis, which is different from the carrying capacity concept. Second, they lack scientific rigor in selecting reduction factors. Third, the target value relevant to resource availability is not included. And fourth, they are strictly Europe-based. The modified EI95's target values for U.S. by Dickinson et al. (2001) also share these limitations except the last one.

The EDIP's carrying capacity was derived from the damage-oriented method while the EI95's target values were based on the distance-to-target method. The major advantage of the EDIP's carrying capacity is that the damage-oriented carrying capacity fits well with the definition of the carrying capacity in the STM context. However, the carrying capacity for acidification and nutrient enrichment is based on a Denmark and European scale. Another limitation is that some key impact categories such as resource consumption and human health impact are not included in the EDIP's carrying capacity.

However, it can be concluded that there has been no set of carrying capacity estimates that fits STM perfectly. The major types of carrying capacity estimates that suit the STM's need will be ones that are U.S. based (or at least U.S. inclusive) so that the STM can be applied initially for U.S. businesses. Furthermore, they should be a based on a damage-oriented evaluation with scientific rigor so the STM is more useful to the LCA practitioners than other LCIA methods. Therefore, a set of carrying capacity estimates that meets these characteristics will be beneficial for the STM applications.

2.6 Summary of Literature Review and Problem Statements

Even though there are a number of environmental performance evaluation frameworks available for reporting environmental performance indicators, there is no single standard approach or even a perception of one. The areas that need to be advanced to make the use of environmental performance assessment more universal are: need for a more standardized and practicable approach, need for the measurement of sustainability, need for the use of the life cycle assessment approach, and need for a narrower but deeper analysis of environmental impacts.

LCA has the advantage over other frameworks and approaches because it takes into account the life cycle considerations. Furthermore, a narrow and in depth analysis of environmental impacts can be accomplished through the LCIA. A few LCIA methods have been proposed and developed, however, none of them is yet universally accepted as a standard. A research topic receiving attention is the search for a standard LCIA method that offers a meaningful single score. By using a single score, the comparison among products, services, or materials is more comprehensible than by using multiple indicators of individual impact categories, e.g., the superiority among alternatives is clear cut. Furthermore, the overall sustainability superiority of a product may also be identified.

A few available LCIA methods that offer single scores have some drawbacks that need to be improved. The major disadvantages of these methods drawn from the literature review are: subjectivity of weighting procedure, weighting factors are not flexible, need of metrics/indicators relative to sustainability, limited impact categories, and the units of these single scores are difficult to understand. Hence, there are opportunities for increasing the LCIA sophistication and utility by addressing these issues. The STM is an LCIA method that has been developed in an attempt to bridge the gaps. The STM is expected to advance LCIA development by dealing with the following technical features: combining economic and environmental evaluations, measuring the environmental impacts related to sustainability, assessing environmental impacts without a subjective weighting procedure, being flexible in the selection of impact categories, and providing environmental condition assessment

Carrying capacity is essential to the STM because it is used to calculate the key environmental performance indicators. The credibility of the STM is based heavily on the soundness of the methodologies used in the carrying capacity estimates. Therefore, it is important to develop the most scientifically sound carrying capacity estimates using reliable supporting documents so that the application of the STM is favorable over other LCIA methods. Consequently, the use of LCA may also be more universal.

This research is motivated by the need to create scientifically-based methodologies to estimate carrying capacity as soundly as possible. These carrying capacity estimates will strengthen the advantages of the STM over other LCIA methods. Moreover, the use of the STM in conjunction with these appropriate carrying capacity estimates will extend the utility of single scores in LCA.

CHAPTER 3

OBJECTIVES AND HYPOTHESES

3.1 Objectives

The primary objective of this research is to develop methodologies for estimating environmental carrying capacity values based on strong environmental engineering and scientific principles. The impact categories taken into account are the ones commonly used in LCA including global warming, stratospheric ozone depletion, acidification, eutrophication, photochemical ozone formation, human toxicity, eco-toxicity, and resource depletion. To accomplish this objective, several sources of available and accessible information as well as science and engineering methods are reviewed and applied to estimate a set of carrying capacities. Once determined, the carrying capacity estimates in this research are compared to the values derived by different concepts available in the literature. It is not necessarily expected that this comparison will show exact compatibility because some of the carrying capacities have been developed for other times and places. Furthermore, the approaches used are also different.

The carrying capacity is intended to be used to derive a set of reference values for the Sustainability Target Method or the STM. Hence, the secondary objective of this research is to demonstrate the application of the estimated carrying capacity in conjunction with the STM in environmental performance evaluations. To accomplish this objective, various LCA case studies are conducted using the STM in conjunction with the carrying capacity estimates developed in this research. The results are compared to those obtained from other LCIA methods.

3.2 Hypotheses

This research is an exploratory study that attempts to develop methodologies for carrying capacity estimates. Consequently, the hypotheses need to be formulated accordingly. In addition, the hypotheses need to be developed on the basis of an assessment to illustrate the advantages and disadvantages of the use of STM in conjunction with the carrying capacity estimates in order to implement an LCA compared to that of other LCIA methods. The significance of this research is its ability to strengthen the STM valuation by providing a set of scientifically sound carrying capacity estimates. Therefore, the hypotheses developed for this research are:

1. *A set of carrying capacity values can be estimated from the approaches developed in this research, providing that the best available data and resources from public reports and existing studies are used.*

Since the concept of environmental carrying capacity is rather new, consensus approaches that can be used to estimate carrying capacity for various types of environmental impacts are yet to be available. Therefore, the focus of this research is to develop feasible approaches for carrying capacity estimates at this early stage. It might be noted that a complete and comprehensive set of carrying capacity values may never be obtained. Acquiring of the most rigorous information and data surely requires a great amount of resources. Therefore, this research has set a criterion for developing approaches that are transparent and easy to apply. The information and data used for carrying capacity estimates should also be available to the public. This will also demonstrate that a set of carrying capacity estimates is not too difficult to build where the time comes, that the STM is used in other parts of the world where data and information is scarce. Less scientifically sound carrying capacity estimates may be used in the STM at an early stage to promote its use. However, STM users should work toward a more sophisticated set of carrying capacity estimates afterwards.

2. *The carrying capacity values estimated here should be significantly different from the values derived by different concepts available in the literature.*

It should be noted here that the term "carrying capacity" in the context of this research is defined as the capacity of the earth to absorb or tolerate potentially stressful burdens imparted at various scales and locations, that is, to accommodate the stresses without showing permanent damage (Yossapol et al. 2002). Therefore, an output-related carrying capacity may be derived from the threshold emission amount of the constituent of concern that begins to worsen an environmental attribute. As presented in Chapter 2, however, the EI95 target values (Goedkoop 1998) and the EDIP carrying capacity (Hauschild and Wenzel 1998) were derived from the known or estimated emission amount of the constituent of concern. The carrying capacity of the same geographical scale estimated in this research, i.e., from the "receptor-side", and the ones derived from the different concept in the literature, i.e., from the "emission-side", should therefore be different. And this difference should be significant because it is conservatively assumed here that the pollutant emissions today are already exceeding the environmental carrying capacity.

This hypothesis can be examined by comparing the carrying capacity values estimated in this research to the ones in other works, which are the studies by Goedkoop (1998) to determine the European target values for EI95, by Hauschild and Wenzel (1998) to determine the Denmark/Europe/Global carrying capacity for EDIP, and by

Dickinson et al. (2001) to modify the EI95 target values to the U.S.-based target values in the early stages of the STM development. The comparisons are discussed mainly in terms of the difference in the carrying capacity values and their characteristics. An explanation of the discrepancy or the concordance are provided. Moreover, the methodologies as well as their advantages and disadvantages are also discussed.

3. *The STM, in conjunction with the carrying capacity values developed here, can be used to conduct an LCA to identify superiority among alternatives in the same fashion as do other LCIA methods. The results obtained from the STM should show similar trends in terms of significant impacts.*

This hypothesis can be examined by using the STM to evaluate the environmental performance of three case study situations. The first case study is the comparison of environmental performance at the process level (four alternatives of energy generation sources) while the second case study is the comparison at the supply-line level (four basic material productions). Last, the third case study is the evaluation of the environmental performance at the product level (a household coffee maker). The results of these three case studies are compared with those of other LCIA methods (EI95, EI99, EPS, and EDIP). The comparisons are discussed mainly in terms of the contributions of individual environmental impacts to the overall single scores for all case studies and in terms of the change in the ordinal ranking of environmental performance for the first two case studies (four energy generation sources and four basic material productions). These comparisons are also made through a qualitative assessment to demonstrate the values and characteristics of the STM and other LCIA methods. A major characteristic of the STM compared to the other LCIA methods is its capability to identify the environmental performance of products related to sustainability as an absolute metric. Even though the other LCIA methods consist of intrinsic subjective judgment in weighting procedure, the value judgment is somehow perceived from the seriousness of ongoing environmental problems. Therefore, the concordance in terms of the contributions of individual environmental impacts to the overall single scores should be observed.

4. *By conducting LCA case studies, the advantages and disadvantages from using STM in conjunction with the carrying capacity values developed here can be identified.*

In Chapter 2, the characteristics of the STM have been presented. However, a thorough analysis of all three case studies should emphasize the advantages and reveal any disadvantages. The analysis should be made for not only the LCA results but also on the way the LCA is conducted. The analyses will be made for the LCA case studies conducted using the STM as well as other LCIA methods.

5. *The STM single scores are sensitive to different carrying capacity values. In other words, a range of carrying capacity values also gives a possibly wide range of STM results.*

The validation of this hypothesis can be demonstrated by a sensitivity analysis of a case study. The results, i.e., the ordinal ranking in terms of environmental performance of alternatives, which is due to a marginal change in individual carrying capacity values, can be investigated. The criterion of the sensitivity analysis is the alteration in ordinal ranking of the case study alternatives corresponding to the change in carrying capacity values. This hypothesis will be valid should the ordinal ranking in terms of environmental performance of alternatives be shifted when the carrying capacity values are varied. A conclusion that can be drawn from this finding will be how the carrying capacity plays a significant role in the application of STM and its results. A rough or less sophisticated carrying capacity may lead to a bias and misleading results for the superiority in terms of environmental performance among product alternatives. This would demonstrate that the credibility of the STM valuation is based heavily on the soundness of carrying capacity estimates. A scientifically sound carrying capacity therefore would be needed in order to minimize any bias.

3.3 Basic Principle of Practicality

The implementation of the above hypotheses and the development of the STM approach need to be guided by an additional operational principle. A goal of the STM, and of LCA in general, is to develop tools that are of practical use to product and process designers in making choices that leads to sustainable products and practice. Therefore, the carrying capacity approach must be easily implemented by skilled individuals, not necessarily environmental professionals, using information and data that is easily available.

3.4 Summary

The objectives of this research are twofold: to develop and evaluate environmental carrying capacity based on environmental engineering and science principles and to demonstrate the application of the estimated carrying capacity in conjunction with the STM in environmental performance evaluation. There are five hypotheses developed for this research.

1. A set of carrying capacity values can be estimated from the approaches developed in this research, providing that the best available data and resources from public reports and existing studies are used.
2. The carrying capacity values estimated here should be significantly different from the values derived by different concepts available in the literature.
3. The STM, in conjunction with the carrying capacity values developed here, can be used to conduct an LCA to identify superiority among alternatives in the same fashion as do other LCIA methods. The results obtained from the STM should show similar trends in terms of significant impacts.
4. By conducting LCA case studies, the advantages and disadvantages of use of the STM in conjunction with the carrying capacity values developed here can be identified.
5. The STM single scores are sensitive to different carrying capacity values. In other words, a range of carrying capacity values also gives a possibly wide range of STM results.

The scope of this research is developed with an aim to examine these hypotheses. After the carrying capacity is developed, the hypotheses are examined by case studies. The outcome of the examination of these hypotheses should provide a significant contribution to research on LCA development.

CHAPTER 4

METHOD

4.1 Less-is-Better and Only-Above-Threshold

Typical LCI data do not provide information on spatial and temporal variations. This absence limits the ability of LCIA to predict actual impacts or relate a burden to actual impact (Potting et al. 1999). Usual LCIA methods overcome the limitation by following a source-oriented or less-is-better approach. A comparative assessment between the differences in emissions of alternatives can be made by using either the impact category basis or the single score basis. The alternative with the smallest emissions is obviously the best one. Figure 4.1 presents an example of a comparative assessment of alternatives within an impact category. In this example, substances A through D are aggregated to produce a single score. The aggregation can be made after characterization factors or potential impacts are applied to all substances. Alternative 2 is the best in terms of environmental performance because of its smallest aggregated environmental emissions.

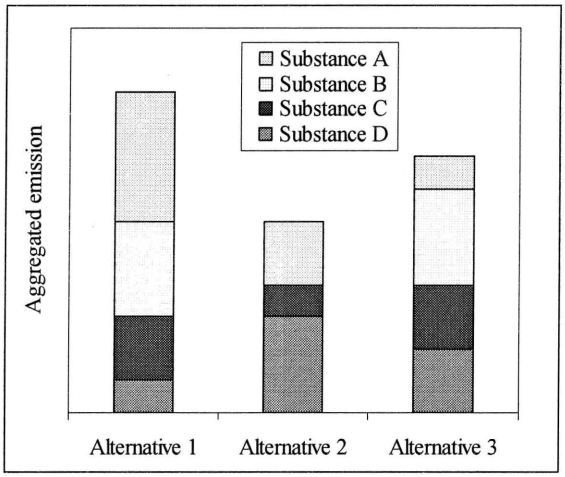

Figure 4.1 Example of a comparative assessment of product alternatives within an impact category following the less-is-better approach.

As pointed out by Potting et al. (1999), the less-is-better approach might result in a poor correspondence between the expected actual impact and the impact predicted by the LCIA. They also criticized the less-is-better approach in terms of its inability to discriminate between processes with emissions causing concentrations below and above a threshold level, which is the intensity level that is just barely perceptible. An approach that may be used for an improved impact assessment is the only-above-threshold approach (White et al. 1995 as cited in Potting et al. 1999).

In the only-above-threshold approach, the actual environmental impacts posed by an environmental burden are identified and compared to a threshold level. The major characteristic of this approach is the ability to distinguish between processes with emissions causing environmental intensities below and above a threshold level (Potting et al. 1999). In the only-above-threshold approach, additional information about actual site conditions is needed to predict whether or not an emission from a process exceeds threshold levels. It is likely advantageous to practice the LCA with help from environmental fate and transport modeling as used in risk assessment and environmental impact assessment (Assies 1998; Owens 1999). The relation between LCA frameworks using the less-is-better and the only-above-threshold approaches are shown in Figure 4.2.

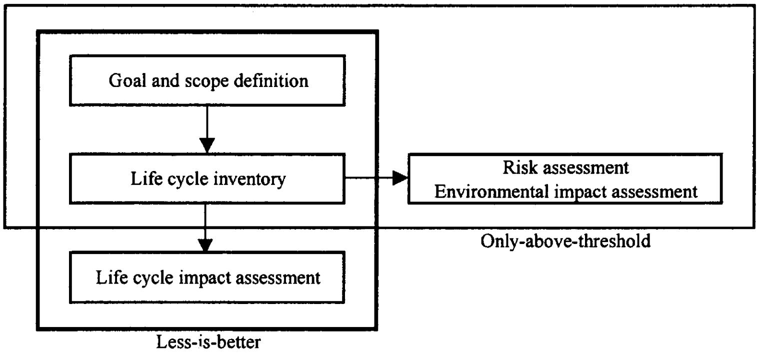

Figure 4.2 The relation between the less-is-better and the only-above-threshold LCA frameworks. (Source: Potting et al. 1999)

The major advantage of the less-is-better approach is its limited requirement for information which makes the approach simple to use and allows the assessment of processes without knowing spatial and temporal variations. The major advantage of the only-above-threshold approach is its accuracy in predicting actual impact because it takes into account these variations as well as environmental background information. Furthermore, it provides more meaningful results for impact assessment in terms of damage endpoints and implies the degree of environmental harm (Assies 1998). However, Heijungs and Huijbregts (1999) criticized the only-above-threshold approach because using environmental modeling for each impact category in every LCA would be very time consuming and cost prohibitive. Table 4.1 summarizes the advantages and disadvantages of the less-is-better approach compared to the only-above-threshold approach.

Table 4.1 Summary of Advantages and Disadvantages between the Less-Is-Better Approach and the Only-Above-Threshold Approach
(Source: Assies 1998; Heijungs and Huijbregts 1999; Potting et al. 1999)

Advantage/Disadvantage	Less-is-better	Only-above-threshold
Correspondence between predicted and actual impact	No	Yes
Ability for temporal and spatial variations	No	Yes
Implies degree of environmental harm	No	Yes
Limited requirement for data and less time spent	Yes	No
Ability to assess "non-defined" processes	Yes	No
Simplicity of use	Yes	No

4.2 Threshold-Oriented Carrying Capacity Evaluation Technique

The STM methodology (Appendix A) may be compatible with the only-above-threshold approach, where the sustainable level in the environmental performance evaluation of a system is analogous to the circumstances when the threshold level is being reached in the only-above-threshold approach. In the STM, the sustainable level is used not only as an indicator of the physical endpoint, which is of direct societal concern, it is also used as the common criterion for human heath, natural environment, and natural resources.

An assessment to compare the environmental performance between alternatives can be made using the sustainable levels as the benchmarks. The better alternative is the one that is associated with less environmental harm compared to its sustainable level. Figure 4.3 illustrates a comparative assessment of product alternatives in the STM. In this illustration, Alternative 2 is the most preferable alternative in terms of environmental performance (0.3X related to sustainability target threshold) while Alternative 3 is the least preferable alternative since its environmental performance exceeds the sustainability target threshold (1.2X). From this example, not only the alternative that has the better environmental performance is identified, but the alternative performance related to the sustainable level is also evaluated.

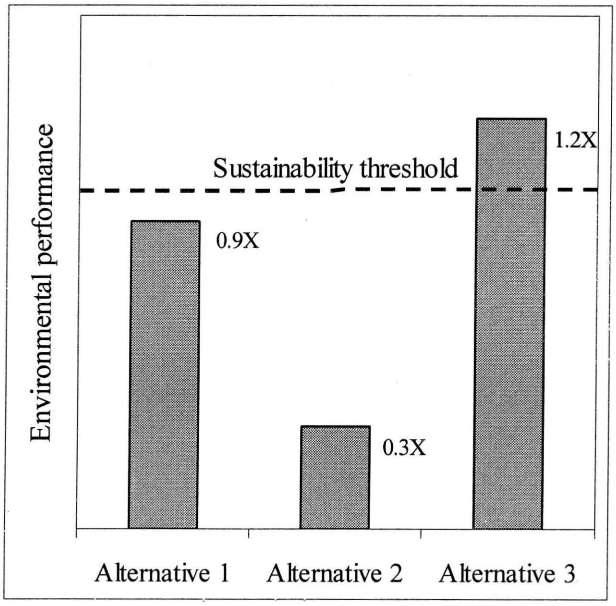

Figure 4.3 An assessment of alternative related to the sustainable level in the STM.

Even though the STM may be analogous to the only-above-threshold approach, there is a slight difference in terms of impact assessment. The feature of the only-above-threshold approach not necessary for the STM is the assessment of the actual environmental impacts due to various degrees of environmental burdens. This is because the STM does not need to investigate what are the actual environmental impacts posed by all levels of burden, rather the STM relates environmental impacts to the sustainable level or the threshold level. The STM indicators are interpreted in terms of the degree of environmental harm incurred as a function of the sustainable level. This comparison can be made by assuming that there is a linear relationship between the environmental cause and the resulting effect (see Figure 4.4). This characteristic is advantageous because this cause-effect linear relation needs an impact assessment to be evaluated only once at the sustainable level for an impact category. This impact assessment will identify the environmental burden (cause) resulting in the environmental condition (effect) at the threshold level.

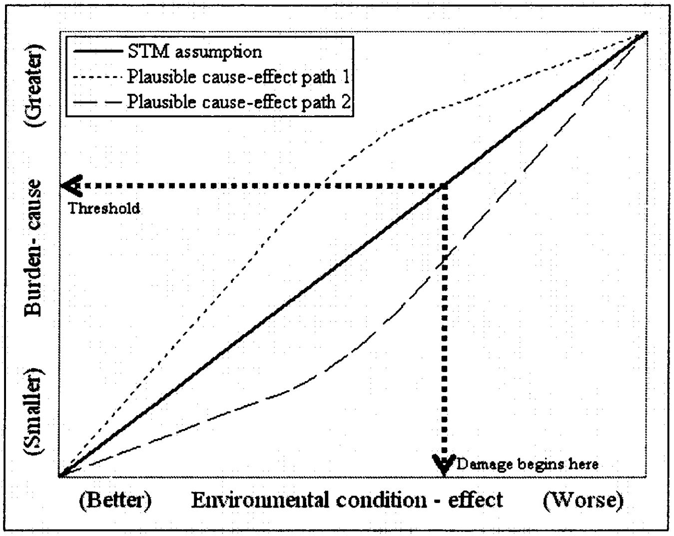

Figure 4.4 The assumption of linear relation between environmental burden (cause) and condition (effect) used in the STM.

By assuming a linear relation between the cause and effect, the following equation can be established for a given emission-related impact category:

$$\frac{\text{Environmental burden}}{\text{Carrying capacity}} = \frac{\text{Environmental concentration}}{\text{Threshold level}} \qquad (4.1)$$

This equation is the anchor of the approaches used to evaluate the emission-related carrying capacities in this research. It indicates that the sustainability of a system can be determined either from the cause (pollutant emission) or the effect (environmental concentration). Therefore, an environmental performance assessment of a system can be made by calculating its environmental emission compared to its carrying capacity. A carrying capacity can be evaluated from the burden that causes the environmental condition reaching the threshold level. This threshold-oriented carrying capacity evaluation technique, hereafter referred to as the "threshold-oriented technique", is used in the context of emission-related carrying capacity evaluation throughout this research.

Risk assessment is explicit in the STM calculation while the environmental impact assessment is used in the carrying capacity evaluation. Usually, a non-probabilistic risk assessment is expressed in terms of risk quotient (RQ) or hazard quotient (HQ) (Murin et al. 1996; Hill et al. 2000; Morello-Frosh et al. 2000). The RQ or HQ can be defined as the ratio between predicted environmental concentration (PEC) and the threshold concentration (Predicted No-Effect Concentration-PNEC):

$$RQ = \frac{PEC}{PNEC} \qquad (4.2)$$

In the STM, EE (Eco-Efficiency) is the practical absolute indicator calculated relative to the sustainable level. This key indicator has the criterion for sustainability at

EE ≥ 100 %. A product with EE > 100 % indicates less impact than the sustainable level while products with EE <100 % are not sustainable. Analogous to the risk quotient in risk assessment, EE can also be used to interpret the degree of sustainability and a risk quotient of a system (Table 4.2). A risk quotient can be thought of as a probability that a system exceeds the sustainability level. This interpretation appearing in an environmental report may be more understandable to audiences than the current common reporting style that is based on the comparative source-oriented or less-is-better approach.

Table 4.2 Risk Quotient of Sustainability in the STM

Eco-Efficiency (%)	Degree of sustainability	Risk quotient of sustainability
0.1	1,000 times worse	10^3
1	100 times worse	10^2
10	10 times worse	10
100	Just sustainable	1
1,000	10 times better	10^{-1}
10,000	100 times better	10^{-2}
100,000	1,000 times better	10^{-3}

Environmental impact assessments are commonly used to identify the environmental effects of an activity, usually at a specific location and at a point of time. The major objectives of the environmental impact assessments are to consider all possible issues associated with a proposed project and to provide the information to facilitate decision-making. For this research, the environmental impact assessment is used in the opposite way in evaluation of the carrying capacity. This technique can be referred to as "inverse modeling" or "inverse calculation". The carrying capacity is estimated on an impact category or subcategory basis. In other words, the approach here is to consider environmental impact scenarios and then to select a level of environmental challenge that produces no un-correctible environmental damage.

First, a perceptible damage function is identified (e.g., no change in global temperature in the global warming impact). Then a specific environmental condition that causes the damage is also identified and specified, which basically are represented by the threshold levels in this case (e.g., an acceptable atmospheric CO_2 concentration that causes no temperature change). The threshold levels are different and specific from place to place and scale to scale according to the economic system available in the context of the STM. Next, the area-specific and time-dependent information is gathered as necessary and plugged into appropriate environmental fate and transport modeling to investigate the cause of the defined threshold. The carrying capacity of an impact category is derived here from an environmental burden that causes the environmental condition to reach the defined threshold level (Figure 4.5).

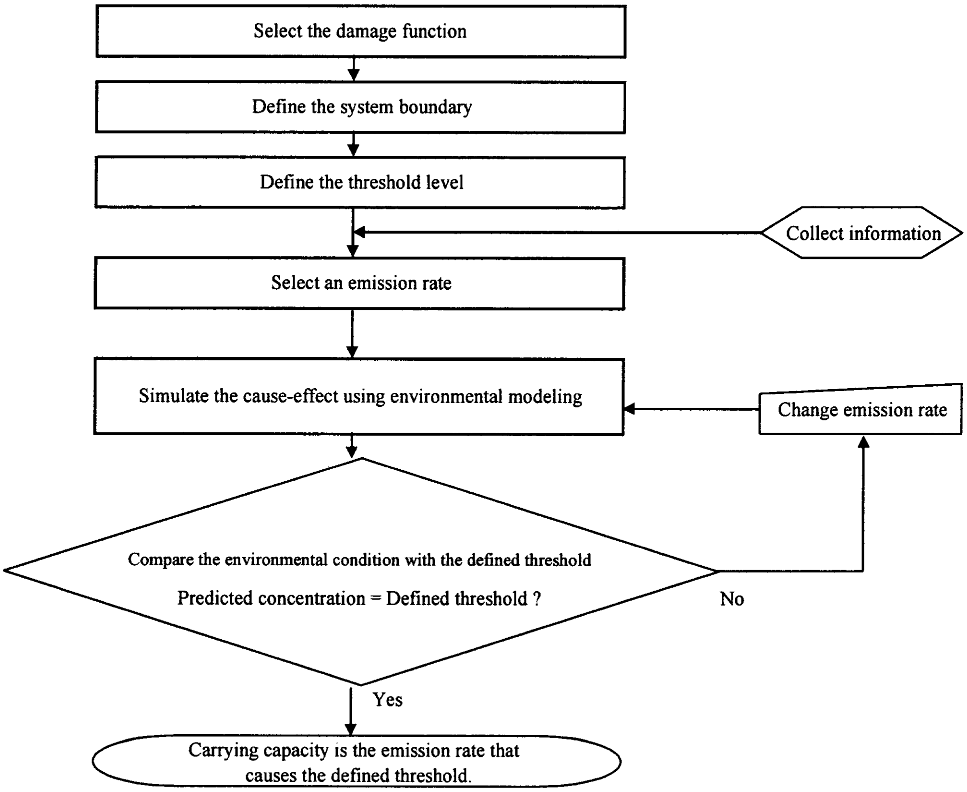

Figure 4.5 Schematic diagram illustrates the algorithm of the threshold-oriented carrying capacity evaluation technique for emission-related impact category.

However, the carrying capacity evaluation of some impact categories in this research may omit some of the steps mentioned above if the perceptible and scientifically based information is available. For example, the perceivable threshold levels for most environmental systems are available through the literature, international agreements, and regulations. Therefore, it is not necessary to estimate the acceptable threshold level from hazard or toxicity assessment. Similarly, the cause-effect relation of the global warming impact (the impacts caused by the emission of greenhouse gases) has been widely studied and is available from the literature. In such cases, modeling for cause-effect relation of greenhouse gas emission and its effect is not needed in this research and the carrying capacity can be directly determined from the level of greenhouse gas emission that causes the impact to reach the criterion threshold level.

The procedure of the threshold-oriented technique may also be adjusted from case to case due to the limitation of information, resources, and environmental modeling. Adjustments may be necessary in order to achieve an initial working set of carrying capacity estimates, particularly when all of the desired information is not available. In these cases the assumptions are noted and suggestions for future refinement of the carrying capacity estimates are made. The details for carrying capacity evaluation of consumption-related impact categories are slightly different from the emission-related impact category and are described in more detail in Chapter 12.

4.3 Perception of Carrying Capacity Estimates

The approaches for carrying capacity estimates employ relevant environmental models to simulate the effects of emissions on various types of environmental impacts. For example, general circulation models (GCMs) may be used to simulate the effect of the earth's surface temperature and atmospheric carbon dioxide concentration from the emission of greenhouse gases. An acceptable "target level" of rising temperature and increasing carbon dioxide concentration can then be made. This acceptable level will be based on practical steps to mitigate the effects of such damage, however there cannot be a realistic "sustainable level" because some damage will still occur. Hence, the development and simulation of environmental models must be very precise and effective because the stakes are very high. As a result, there are a number of international cooperative activities working on issues such as global warming and stratospheric ozone depletion, acidification, and eutrophication.

Therefore, as long as the international agreements on a well-defined "target levels" are available, it is feasible to use them as a surrogate for the carrying capacity for each respective impact category. However, the target levels are not available for all impact categories under consideration because some environmental issues are locale-dependent and the target value for some impacts has not been reached. As an example, the Kyoto Protocol calls for the ratification, acceptance, and approval from the Parties to the UNFCCC (United Nations Framework Convention on Climate Change) so that the next steps for mitigation of the global warming issue can be progressed. However, even though the criteria for the ratification have recently been fulfilled, agreements on the details of greenhouse gas emission projection and implementation of mitigation are still being worked out.

Another conclusion from reviewing how international agreements on environmental issues are settled is the use of an approach similar to the threshold-oriented approach to project the emission allowance. An allowable emission estimate starts from defining the "target level" for the environmental concentration in question.

Next, an appropriate environmental model is used to predict the emission of the pollutant that causes environmental concentrations to reach the target levels. Thus, the carrying capacity is derived from the predicted allowable emission.

A single well-defined target level may be available from the literature or from other studies. However, it is difficult to select the most publically appropriate one if they are available from several sources. Hence, using the most perceived or acceptable target level, which may be more or less stringent than the well-defined target level, may be more appropriate. However, these target levels are used only if they were derived on a scientific basis. In this research, the target levels are based primarily on an extensive application of regulatory levels and standards. This application assumes that the regulatory levels and standards are general agreements and are surrogates for non-damage environmental health (only-above-threshold approach), which is fundamental to the sustainability concept. Of course, regulatory levels are subject to change based on new information and perspective. Such change may result in change to the acceptable threshold levels as well. However, the assumption in this research is that this approach provides a valid working level, realizing that additional data may later be required. Also, the assumption of using regulatory levels as a surrogate of the entire environmental systems is to be clearly stated.

Selecting appropriate environmental models and datasets is as important as the selection of target levels. A sophisticated model should provide an accurate result, but this is not guaranteed. Environmental modeling should also rely on the best set of assumptions in order to yield a model that is both realistic and useful. It is very difficult to assemble a complete, detailed description of all the environmental conditions. It is thus necessary to make numerous simplifying assumptions or statements about the condition of the environment. However, an excessively simple model may be misleading. Alternatively, an excessively detailed model may be too complex and thus unlikely to be used. The estimates for carrying capacity should rely on simple but applicable models yet should be transparent to LCA practitioners so that they can justify the methodology or even make further modifications based on new data.

Environmental models range from very simple models that can be processed by a handheld calculator to very complex models that require a large amount of time and resources for their use. The selection of a model must be based on the purpose of the work and the availability of resources. Other factors which must be taken into account are: the level of detail and accuracy needed for the analysis, the degree of technical competence used in the simulation modeling, and the detail and completeness of the database. Appropriate data should be available before a model is chosen. A model that requires detailed and precise input data should not be used if the data is not available.

There are several methodologies and approaches that can be used to estimate carrying capacities depending on the individual impact category and its associated model and threshold levels. The use of these methodologies and approaches requires scientific judgment coupled with a reasonable time frame and sufficient resources. Therefore, carrying capacity estimates in this research, although still at an early stage, have employed transparent yet useful models presented in the following chapters. Moreover, the carrying capacity estimates developed in this research are designed to provide usable and comparative values that can be rationally modified when necessary.

4.4 Economic Carrying Capacity and Natural Burden

The STM defines the economic carrying capacity (ECC) as the natural carrying capacity (NCC) minus the natural burden (NB), $ECC = NCC - NB$ (Dickinson et al. 2001). In other words, ECC is the portion of the total carrying capacity that is available to industry and commerce to support the needs of society. ECC is used as the reference in the STM calculations. The natural burden is not a focus of this research. Consequently, the term "carrying capacity" in this research is referred to as the economic carrying capacity. In some applications, the natural burden may need to be discussed or considered.

Generally, environmental emission can be divided into natural emission (or natural burden) and human-induced emission. It can be assumed that the human-induced environmental emissions are always associated with economic value. Therefore, the carrying capacity estimates in the following chapters are the ones that result from human activities. The natural burden of the output-related impact categories, which is the uncontrollable natural background emission such as natural emissions of greenhouse gases, is difficult to estimate and therefore is not included in this research. In some impact categories such as the toxicity impacts, it is assumed that the toxic substances are solely released from human activities. As a result, the background concentrations of the toxic substances accumulated by natural emissions are assumed to be insignificant and can be negligible. Analogous to the output-related carrying capacity, the economic carrying capacity for water consumption is the portion that is left over after the natural take-out.

4.5 Temporal and Spatial Variations in the STM and Carrying Capacity Estimates

An advantage of the STM is its potential to deal with temporal and spatial variations in LCA. The temporal and spatial variations are taken into account where the Environmental Impact (EI) is the aggregation of EI from several aspects or life cycle stages of a system as illustrated below. EI is quantified by adding normalized environmental aspect levels to obtain an aggregated quantity (Dickinson et al. 2001):

$$EI = \frac{A_1}{A_{R1}} + \frac{A_2}{A_{R2}} + \frac{A_3}{A_{R3}} + \ldots + \frac{A_N}{A_{RN}} \tag{4.3}$$

Where A_N denotes an Aspect Level (e.g., kWh/year for energy, lb/year for consumption) and A_{RN} is the Aspect Reference Level indicating the level at which the aspect would have a significant environmental impact.

In an LCA context, LCI typically presents the data in terms of impact associated with a process or product. The EI per unit production rate can be expressed as (Dickinson et al. 2002):

$$EI_{PR} = \frac{EI}{P} = \frac{A_1^*}{A_{R1}} + \frac{A_2^*}{A_{R2}} + \frac{A_3^*}{A_{R3}} + \ldots + \frac{A_N^*}{A_{RN}} \tag{4.4}$$

Where P is the production rate and A^*_N is the aspect quantity per product unit (A_N/P). For each aspect, the reference level A_R depends on the environmental impacts caused by the aspect, the associated carrying capacities, and the relationship of V_R (the Value Reference Level corresponding to the Aspect Reference Levels) to total economic output. For each impact, part of the economic carrying capacity can be associated with the reference firm in the same proportion as the ratio of V_R to total global or regional economic output, depending on the geographical scale of the impact. This is the "Impact Reference Level", I_R (Dickinson et al. 2001):

$$I_{RN}\ (Global) \quad = \quad CC_{Global,N} \times V_R / GGP \qquad (4.5)$$

or $$I_{RN}\ (Regional) \quad = \quad CC_{National,N} \times V_R / GDP \qquad (4.6)$$

Where $CC_{Global,N}$ and $CC_{National,N}$ are the carrying capacities at the global and national scales respectively. GGP (Global Gross Product) and GDP (Gross Domestic Product) are total global and national economic product ($/yr) respectively. The above equations demonstrate the relationship between an Impact and its corresponding carrying capacity. They also demonstrate that the boundary or scale of an economic value and the Impact must correspond to each other. In other words, the carrying capacity must be evaluated for the same year and at the global or national scale that is in accord with the GGP or GDP, respectively. The temporal dimension of an Environmental Impact (EI) can be addressed by using a carrying capacity and the GGP or GDP that are based on the same year. For an input-related Environmental Impact, the spatial dimension can be addressed by evaluating the carrying capacity specifically at the national scale depending on where the resource is extracted. However, the spatial dimension for an output-related Environmental Impact can be addressed through three different steps for the impact categories that are local or regional rather than global issues.

- First, the carrying capacity (CC) is evaluated specifically from nation to nation depending on where the system is taking place.
- Second, site-dependent impacts that take into account the fate, transport, and exposure of the pollutant emission may be applied.
- Third, a locally-specific carrying capacity that takes into account the sensitivity of the receiving environment may be applied.

For the first step, an output-related carrying capacity at the national scale can be evaluated following the algorithm described in Section 4.2. An important assumption made here is that this carrying capacity can be calculated from an assumed uniform pollutant emission over the areas across a national boundary. In other words, areas across a national boundary are assumed to be a part of the unit receiving environment. The site-dependent and locally-specific considerations in the second and third steps may provide a better picture depending upon the availability of information. A site-dependent Impact (I'_N) can be determined by applying a site factor (SF) that includes transport, fate and exposure factors, $I'_N = I_N \times SF$. A locally-specific carrying capacity (CC') can be determined by assuming that a nation has the same degree of environmental sensitivity to the impact as the area where the system to be evaluated resides.

Among the output-related impact categories studied in this research, the adverse impacts posed by emissions of greenhouse gas (global warming) or ozone depleting substances (stratospheric ozone depletion) are considered to be operable on a global scale, i.e., any place on the earth will experience the same effect. Analogously, photochemical ozone formation, acidification, and eutrophication impact categories are considered to be operable on a regional or continental scale while human toxicity and eco-toxicity impact categories are considered to be functioning at a local scale. For resource depletion, which is an input related-impact category, the impacts can be on a national scale or a global scale considering that the trades of resources can be made globally. However, within a single analysis care should be taken to make certain that the scale selected truly reflects the sources used for the product or process.

Basically, the Environmental Impact of a product is the aggregation of multiple life cycle stages, ranging from the extraction of materials to the end-of-life management of the product. For the entire spectrum of aspects and life-cycle stages of a product, process, or service, the overall Environmental Impact should take into account the temporal variation and spatial variations, e.g., when and where those life-cycle stages took place.

The time and spatial variations in a complex system such as an entire corporation can also be dealt with in a similar manner to that discussed above. Conducting an environmental performance evaluation using the STM for a corporation level is more complex than one at the supply line level because the corporate level deals with more life cycle environmental aspects. To illustrate this point with an example, a manufacturing firm imports different parts from several locations for its assembly lines. These parts were manufactured at supply line levels at specific sites. An STM implementation of a part of a supply line deals only with the carrying capacity associated with a specific site. When conducting an environmental performance assessment for the manufacturing firm, the temporal as well as the spatial variations must be taken into account. Furthermore, environmental aspects associated with the manufacturing firm itself must also be included. These environmental aspects can include material consumption, electricity consumption, and waste generation.

The STM implementation at the corporate level involves carrying capacity with temporal and spatial variations. The variations in carrying capacity depend on the specific time and place where the products at the supply line level were manufactured. In addition, the variations in carrying capacity also depend on the number of aspects to be considered. More details on the preceding manufacturing processes of those products at the supply line level result in more variations in carrying capacity.

4.6 Summary

This chapter has discussed the commonly used less-is-better approach for LCA. Due to some limitations of this approach, LCIA development may need to shift to a more reliable approach, the only-above-threshold approach in order to provide information about the "distance" a product must go before it can be seen as sustainable. The STM is compatible with the only-above-threshold, where the sustainable level is analogous to the threshold level. Risk assessment is utilized in the interpretation while environmental impact assessment is utilized in the carrying capacity evaluation.

The threshold-oriented technique is developed exclusively for the evaluation of the carrying capacity in the context of the STM. This technique can be thought of as an "inverse modeling" or "inverse calculation" of the environmental impact assessment. For the first step of this technique, a perceptible damage function is identified. Then a specific environmental condition that causes the damage is also identified and specified as the threshold level. Next, the location-specific and time-dependent information is collected and put into an appropriate environmental model to investigate the pollutant emission rate. The carrying capacity is, therefore, an emission that causes the environmental condition to reach the threshold level. However, the results of the carrying capacity evaluation may be adjusted from case to case as new information becomes available that changes the limitations of information, resources, and environmental modeling.

The value of the STM is based on the soundness of the carrying capacity estimates. Therefore it is important to develop reliable and rational carrying capacity estimates based on the best available consensus scientific approaches or agreements. Where available, international agreements, regulatory levels, or consensus emission target levels may be used as the carrying capacity or used to estimate the carrying capacity. It should be noted that some carrying capacity estimates cannot be accomplished at this stage because they require a great amount of time and resources to do so. Scarcity of information and scientific-based approaches also hinders the study. When this is true, approximations and assumptions have been made; these are carefully noted and represent areas where additional work and modification will be necessary in the future.

Selecting appropriate environmental models and datasets is also important. Environmental modeling should rely on the best set of assumptions that yields a model that is both realistic and useful. However, an excessively detailed model will be too complex and thus unlikely to be used. Furthermore, it is not economical to do so. On the other hand, an excessively simple model may be misleading. Carrying capacity estimates should rely on simple but applicable models, yet should be transparent to users so that they can justify the methodology or can make further modifications should they be required. Therefore, carrying capacity estimates in this research have employed transparent yet useful models as seen and demonstrated in the following chapters.

CHAPTER 5

GLOBAL WARMING IMPACT

5.1 Overview

Global warming or the greenhouse effect is created by an accumulation of CO_2, methane, water vapor, and other greenhouse gases in the atmosphere of the earth, which traps the heat that radiates from the earth's surface back into space. By absorbing infrared radiation, greenhouse gases (GHGs) impact the flow of natural energy through the climate system. The climate must somehow adjust to the higher concentrations of GHG in order to maintain the balance between the energy arriving from the sun and the energy escaping back into space. There is a growing consensus within the scientific community that GHGs of manmade origin are responsible for the global warming impact currently observed (Houghton et al. 2001). Many GHG-emitting activities are essential to the global economy and a fundamental part of modern life. Carbon dioxide from the burning of fossil fuels is the largest single source of GHG emissions from human activities. The annual global CO_2 emissions from fossil fuel consumption have been increasing, reaching 5.4 Gt C/yr in the 1980s and 6.3 Gt C/yr in the 1990s (Marland et al. 2002).

Ideally, the carrying capacity for CO_2 should be estimated from the absorption capacity of the major global sinks, i.e., oceans and lands. However, it is difficult to model a global carbon balance because it is not static in time and there are a large number of factors involved. The current air-ocean flux of human-induced GHG was estimated at 0.6 to 2.8 Gt C/yr (Takahashi et al. 1997; Houghton et al. 2001; Sarmiento and Gruber 2002; Jacobson et al. 2003; McNeil et al. 2003) while the air-land flux was estimated at 0.8 to 1.4 Gt C/yr (Houghton et al. 2001). Sarmiento and Gruber (2002) estimated a higher air-land flux at 1.9 Gt C/yr mentioning that the re-growth of farmlands and forests are the major sinks. Potter et al. (2003) estimated the land sinks for North America, Eurasia, and the southern hemisphere between 0.5-0.9 Gt C/yr.

Hauschild and Wenzel (1998) suggested a uniform estimate of the environmental carrying capacity assessed from the balance of the carbon cycle to be 3.8 Gt C/yr for LCA purposes, which is at the high end of the range mentioned above. The impairment of ecosystems and the increase in global temperature can also be used as indicators in setting up a target threshold level. In EI95, the target for GHG emissions was defined as 5% impairment of the ecosystems and an increase of mean global temperature of 0.1 °C per decade (Goedkoop 1998). At this impairment level, a reduction factor of 2.5 was selected to calculate the target value from the estimated European 1990 emission of 0.65 Gt GWP/yr[1]. As a result, the European target value of 0.26 Gt GWP/yr is used in EI95.

The carrying capacity for the global warming impact can be estimated following the threshold-oriented technique. This requires that the threshold level be identified in order to calculate the carrying capacity. The carrying capacity for the global warming impact may have been exceeded already since it is believed that climate change has already taken place because of massive emissions of GHGs. As a result, the threshold level should be determined from a stabilization level that mitigates the damage seen already due to human-induced GHG emissions. This chapter presents an overview of the future emission and mitigation scenarios for GHG emissions. The possible stabilization levels are reviewed and a threshold level is specified. The carrying capacity can be determined from the emission scenario that maintains the specified threshold level.

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[1] GWP denotes global warming potential which is often referred to as CO_2-equivalent.

5.2 Greenhouse Gas Emission and Mitigation scenarios

5.2.1 Greenhouse Gas Emission Scenarios

A greenhouse gas emission scenario can be defined as a description of potential future human-induced emissions based on multiple driving forces including economic factors, demographic factors, policy factors, technological factors, and human responses (Metz et al. 2001). The Intergovernmental Panel on Climate Change (IPCC) developed an earlier set of reference emission scenarios, known as IS92, in an attempt to project future climate effects from its previous assessment reports. The IS92 consists of six different pathways or scenarios of which scenario IS92a is the mid-range scenario assuming the best estimate value of climate sensitivity (Houghton et al. 1996). The IS92a projects the human-induced GHG emissions approximately at 20 Gt C/yr in 2100 (Houghton et al. 2001). A general circulation model (GCM) consisting of an atmospheric circulation model and an ocean circulation model, predicted that the IS92a would result in an increase in global mean surface temperature relative to 1990 of about 2.5 °C by 2100, an increase in average sea level of about 0.30 meter, and an increase in average CO_2 concentration to the level of about 750 ppmv (Houghton et al. 2001).

The increasing political and research interest in global warming has led to an extensive number of emission scenarios. As a result, a database was developed to collect, manage, and analyze the emission scenarios to enable researchers to access and identify the strengths and weaknesses of the scenarios for further improvement (Morita and Lee 1998). Over 400 emission scenarios from this database were reviewed and evaluated by a team appointed by IPCC (Nakicenovic et al. 1998). Six modeling teams of specialists from 18 countries then developed a total of 40 scenarios contributing to IPCC based on the scenarios from the database (Nakicenovic 2000a). The new set of emission scenarios was later summarized in the form of four "marker scenarios" published in the Special Report on Emissions Scenarios (SRES) by IPCC, which was intended for use in the following assessment report (Nakicenovic 2000b). The new set of emission scenarios consists of four scenario families (A1, A2, B1, B2), each of which has a theme called a storyline that describes future trends of scenario factors and key parameters. The family A1 was divided into three distinguishable groups (A1FI, A1T, and A1B) based on sources of energy used and technologies involved. Consequently, there are six illustrative SRES scenarios developed, each of which, as well as other IPCC scenarios, is considered equally probable (Nakicenovic 2000b). A short description of these SRES scenarios is presented in Table 5.1.

Table 5.1 Description of the IPCC-SRES Emission Scenarios
(Source: Houghton et al. 2001)

A1. The A1 scenarios all describe a future world of very rapid economic growth and global population that peaks in mid-century and declines thereafter, and the rapid introduction of new and more efficient technologies. Major underlying themes are convergence among regions, capacity building, and increased cultural and social interactions, with a substantial reduction in regional differences in per capita income. The difference between the A1FI, A1B, A1T and scenarios is mainly in the source of energy used to drive this expanding economy. A1FI: Fossil-fuel Intensive, coal, oil, and gas continue to dominate the energy supply for the future. A1B: Balance between fossil fuels and other energy sources A1T: emphasis on new Technology using renewable energy rather than fossil fuel.
A2. The A2 scenario describes a very heterogeneous world. The underlying theme is self-reliance and preservation of local identities. Fertility patterns across regions converge very slowly, which results in continuously increasing global population. Economic development is primarily regionally oriented and per capita economic growth and technological change are more fragmented and slower than in other storylines.
B1. The B1 storyline and scenario family describes a convergent world with the same global population that peaks in mid-century and declines thereafter, as in the A1 storyline, but with rapid changes in economic structures toward a service and information economy, with reductions in material intensity, and the introduction of clean and resource-efficient technologies. The emphasis is on global solutions to economic, social, and environmental sustainability, including improved equity, but without additional climate initiatives.
B2. The B2 storyline and scenario family describes a world in which the emphasis is on local solutions to economic, social, and environmental sustainability. It is a world with continuously increasing global population at a rate lower than A2, intermediate levels of economic development, and less rapid and more diverse technological change than in the B1 and A1 storylines. While the scenario is also oriented toward environmental protection and social equity, it focuses on local and regional levels.

Compared to the IS92 scenarios, the SRES scenarios cover a larger range of emission distribution as well as relative future climate change predictions (Table 5.2). A study by Dai et al. (2001a) found that a variant IS92a emission scenario by including the initial business-as-usual (BAU) pathway in the emission profile provided future climate profiles similar to that to those of the SRES A1 family.

Table 5.2 Projected Manmade Greenhouse Gas Emissions and Predicted Temperature and Sea level rise According to IPCC Emission Scenarios
(Source: Houghton et al. 1996; 2001)

IPCC Scenario	Approximated values in 2100			
	GHG emissions (Gt C/yr)	CO_2 concentration (ppmv)	Increased temperature relative to 1990 level (°C)	Sea level rise relative to 1990 level (m)
1990 level	7	360	-	-
IS92a	20	750	2.5	0.30
IS92b	18	650	1.0 - 3.5	0.1 - 0.95
IS92c	5	500		
IS92d	10	550		
IS92e	36	900		
IS92f	26	700		
A1FI	28	950	4.5	0.50
A1T	4	600	2.5	0.35
A1B	13	700	3.0	0.35
A2	29	850	3.5	0.40
B1	5	550	2.0	0.30
B2	13	600	2.5	0.35

The most probable emission scenario has not been decided even though different reference emission scenarios have been developed; as a result, the uncertainty of the scenarios in terms of probability distributions is not provided. There are some attempts to take into account the uncertainties of parameters in the climate modeling. One of which is the study by Stott and Kettleborough (2002) that considered the uncertainties of GHG origins. They predicted that the global-mean surface temperature would rise from 0.3-1.3 °C relative to 1990 level for a short-term prediction (2020-2030). A study by Webster et al. (2002), was an attempt to include the uncertainties of GHG emissions, and found that the SRES scenarios were biased towards the higher temperature because of the strongly optimistic assumptions about the reductions in sulfur emissions. On the other hand, Knutti et al. (2002) concluded from their study considering the uncertainty analysis of radiative forcing and climate sensitivity that the SRES scenarios might significantly underestimate the probability of a strong warming.

5.2.2 Greenhouse Gas Mitigation Scenarios

A mitigation scenario can be defined as an emission scenario that assumes policies to mitigate climate change (Metz et al. 2001). Mitigation scenarios are developed to investigate the stabilization level that would prevent damage or future damage to climate systems caused by human-induced GHG emissions. According to the IPCC, mitigation scenarios can be classified into four categories based on type of mitigation: concentration stabilization scenarios (the most widely developed), emission stabilization scenarios, safe emission corridor scenarios, and other mitigation scenarios (Metz et al. 2001).

The IPCC developed a set of illustrative pathways for stabilizing CO_2 level at 350, 450, 550, 650, and 750 ppmv over the next three hundred years (Houghton 1994). The emission profiles of these mitigation scenarios, known as WGI (Work Group I) or S (Special) scenarios, were constructed based on the IS92a reference emission scenario. A simple climate model was used to make temperature and CO_2 concentration projections (Harvey et al. 1997; Houghton et al. 2001). Stabilization levels were set to be reached at different time spans from 2100 to 2250. The CO_2 emissions at 2300 are projected at the levels of approximately 1 to 5 Gt C/yr compared to the 1990 level (7 Gt C/yr). Similar climate profiles were obtained using a complex general circulation model by Dai et al. (2001a; 2001b; 2001c) for the 550 ppmv emission scenario.

Wigley et al. (1996) revised the WGI scenarios by adding an additional economic constraint that the emission profiles initially track a BAU pathway, which is an idealization of the assumption that the initial departure from BAU would be slow compared to the immediate reduction in emissions for the WGI scenarios. The emission and CO_2 concentration profiles according to these scenarios, known as WRE (Wigley, Richels, and Edmonds) scenarios, are slightly higher than those of the WGI scenarios in the early years due to the BAU constraint. However, time periods to reach stabilization levels and CO_2 emission levels for 2300 for both sets of scenarios are alike.

More than 150 out of 500 emission scenarios collected in the SRES database were identified as mitigation scenarios (Metz et al. 2001). Approximately half of them are concentration stabilization scenarios and most of these use atmospheric CO_2 concentrations of 550 ppmv as the target stabilization level. Following the previous illustrative example used in the IPCC studies on mitigation scenarios, these studies adopted a common concentration of 550 ppmv. However, there is no consensus, policy, or agreement implying that this CO_2 concentration level is the most desirable one.

Following the publishing of the SRES, an evaluation process for the SRES-based mitigation scenarios (the so-called post-SRES mitigation scenarios) based on concentration stabilization was prepared (Morita et al. 2000). A total of 76 post-SRES mitigation scenarios with CO_2 stabilization concentrations of 450, 550, 650, and 750 ppmv were evaluated by nine modeling teams. The year 2150 was selected as the target year to reach the stabilization levels. An assessment of the post-SRES emission scenarios was published in a recent IPCC report (Metz et al. 2001). Again, the IPCC has not drawn a conclusion as to which would be the best or the most probable mitigation scenario.

5.3 Threshold Level for Carrying Capacity Estimate

The ultimate objective of the United Nations Framework Convention on Climate Change (UNFCCC) and the IPCC is to stabilize the atmospheric CO_2 concentrations at a level that would prevent dangerous human interference with the climate system. Even now, "dangerous" remains undefined, and no official body has taken responsibility to provide a definition. This is because there are still some uncertainties on the scientific aspects of the cause-effect relationship between GHG emissions and climate change (Karl 2001). The IPCC has concluded that defining "dangerous" is a political decision.

For carrying capacity estimation, CO_2 concentration is used as the endpoint indicator for the threshold level selection, following most of the mitigation scenarios studied by the IPCC. For each CO_2 stabilization level, the range of costs and benefits of climate change may be evaluated in terms of sea level rise, water stress, biodiversity, social and economic impacts, and possibilities for adaptation. Any political decision on the dangers of GHG concentrations would have an influence on the emissions control policies of all countries, as it would ultimately set an absolute level of emissions globally.

The benefits of stabilizing atmospheric GHG concentrations will be affected by climate sensitivity. The range of benefits arising from the different stabilization levels varies according to uncertainties about climate sensitivity. A summary of the benefits of the stabilization of atmospheric GHGs is discussed below (Watson 2001).

Stabilization at 750 to 1000 ppmv. Significant impacts associated with warming up to 3 °C would not be prevented. For average to high climate sensitivity, global mean warming would exceed 4 °C, with land areas generally much warmer, resulting in many severe effects and posing risks of large scale, high impact events in future centuries.

Stabilization at 550 to 650 ppmv. Stabilization in this range may significantly lessen and possibly avoid some of the impacts associated with warming greater than 3 °C for average to low climate sensitivity. However, it would not prevent the substantial impacts associated with the warming up to 3 °C, such as the loss of some unique vegetation systems, extensive coastal wetland loss, decreases in crop yields, and other adverse impacts. There is a risk of warming by more than 3 °C for high climate sensitivity.

Stabilization at 450 ppmv. Stabilization at this level is likely to limit global warming to less than 3 °C, even for high climate sensitivity. The impacts for 3 °C warming may be significantly reduced and some may be avoided. However, there would still be risks for impacts associated with warming less than 3 °C. The benefits of stabilization at 450 ppmv are clearly greater than those for stabilization at higher concentrations.

However, it should be emphasized that the earth's carrying capacity to absorb human-induced GHG emissions may have been exceeded already because it is believed that climate change has already taken place. It is assumed for this research that global warming is the sole effect from human-induced GHG emissions and the global warming will not encounter any further damage if CO_2 concentrations are limited to the pre-industrial level of 280 ppmv[1] (Houghton et al. 2001). Therefore, the pre-industrial CO_2 concentrations can be considered as the stabilization level or the threshold level for estimating the carrying capacity for global warming impact. The carrying capacity can be adopted from the GHG emissions in the mitigation scenario that lowers the CO_2 concentrations to 280 ppmv and allows the climate to recover to the pre-industrial situation. Should this stabilization level not be viable, a more viable one may be used.

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[1] The average atmospheric CO_2 concentration in 1998 was 365 ppmv.

5.4 Carrying Capacity Estimate

A mitigation scenario from the literature for the atmospheric CO_2 stabilization level of 280 ppmv is not currently available. However, a mitigation scenario can be simulated and its consequent climate sensitivity can be interpolated using the Java Climate Model (JCM)[1]. In this research, mitigation scenarios in this model are estimated based on the mid-range IS92a reference emission scenario. The results from the simulation using the JCM shows that the stabilization at 280 ppmv is not a viable option because the natural sink sources do not have sufficient capability to reduce the atmospheric CO_2 to reach the pre-industrial level of 280 ppmv. This simulation assumes that the technologies for carbon capture and sequestration will not be available in the future. By using the JCM, it is found that the most conservative yet viable mitigation scenario is the stabilization level at 380 ppmv. Even though this is a viable option, an immediate cut of global GHG emissions must be made to make it practical. This outcome seems to be unrealistic. However, it illustrates a possible pathway of achieving the most conservative option that considers the natural absorption capacity to slow and minimize future climate change.

It is assumed in this research that the CO_2 stabilization level at 380 ppmv is the sustainable level considering that it is the most conservative yet also feasible mitigation scenario. The carrying capacity is defined as the annual allowable human-induced GHG emissions following the mitigation scenario with the stabilization level at 380 ppmv. According to this scenario, the GHG emissions must be cut immediately from the current level to about 3 Gt C/yr in 2010 and 2 Gt C/yr in 2020. At this emission projection, the temperature will be stabilized at +0.9 °C (2060) relative to 1990 level and the

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[1] JCM (Matthews 2003a; 2003b) is a simple model for simulating climate situations forced by GHG emissions. The JCM is developed following the principles described in Harvey et al. (1997).

atmospheric CO_2 concentrations will reach the stabilization level within a few years from now. Figure 5.1 and Figure 5.2 depict the profiles for GHG emissions, average atmospheric CO_2 concentration, temperature change, and sea level rise read from the results obtained from the JCM. In this figures, dashed lines are for zero emissions.

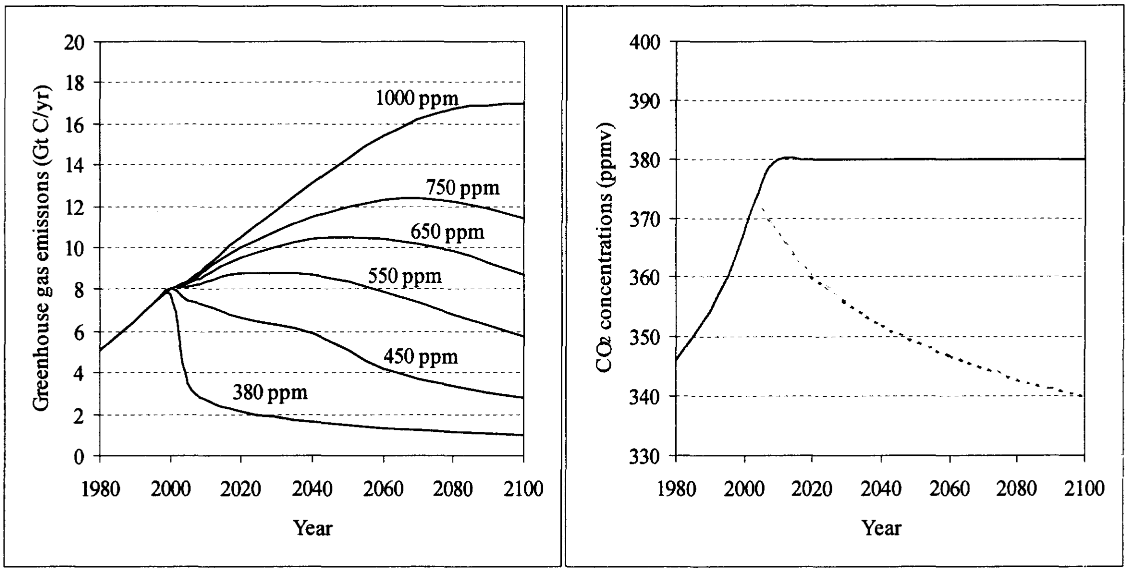

Figure 5.1 Greenhouse gas emission profiles at different mitigation scenarios (left panel) and CO_2 concentration profiles for stabilization level at 380 ppmv (right panel).

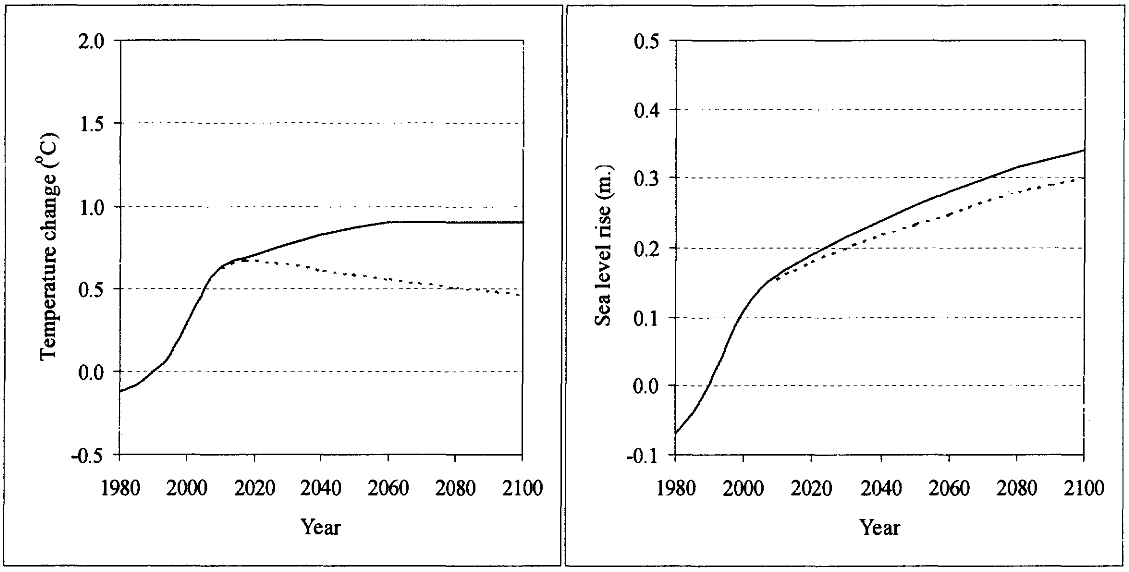

Figure 5.2 Temperature change profile (left panel) and sea level rise profile (right panel) for stabilization level at 380 ppmv.

In LCIA, the quantification of greenhouse gas emissions is usually expressed in terms of CO_2 equivalents (CO_2-eq) (Goodkoep 1998; Goedkoop and Spriensma 2001; Guinee 2001; Hauschild and Wenzel 1998). The conversion of GHG emissions reported in Gt C to kg CO_2-eq can be made simply by using the following equation:

$$kg\ CO_2\text{-}eq \quad = \quad Gt\ C \times 44/12 \times 10^{12} \qquad (5.1)$$

The emission of other GHG emissions can be characterized relative to CO_2-eq using the Global Warming Potentials or GWPs, which are used to compare the ability of each greenhouse gas to trap heat in the atmosphere relative to CO_2 (Houghton et al. 2001; Pennington et al. 2000; WMO 1994; 1998a; 2003; Sygna et al. 2002). GWP values are summarized in Table C.1 Appendix C. GWPs have units of kg CO_2/kg GHG_i, where i represents a species of GHG. The time horizon of 100 years is often used as the reference for GWPs (Houghton et al. 2001; Fearnside 2002). A simple equation is used to characterize the amount reported in kg of GHG relative to kg of CO_2-eq:

$$kg\ CO_2\text{-}eq \quad = \quad GWP_i \times kg\ GHG_i \qquad (5.2)$$

5.5 Summary

A time-dependent carrying capacity for GHG emission is estimated from the mitigation scenario that stabilizes the atmospheric carbon dioxide concentration at 380 ppmv. The emission levels in this mitigation scenario may not be realistic. However, it illustrates a possible pathway of achieving the most conservative option that considers the natural absorption capacity to minimize the future climate change. At this emission projection, the temperature will be stabilized at +0.9 °C (2060) relative to the 1990 level and the atmospheric CO_2 concentrations will reach the stabilization level within a few years from now. The carrying capacity is estimated annually up to the year 2100 as shown in Table 5.3 but the annual carrying capacity from 2010 to 2100 can be estimated from Figure 5.1.

Table 5.3 Carrying Capacity for Greenhouse Gas Emissions

Year	Gt C/yr	10^{12} kg CO_2-eq/yr	Year	Gt C/yr	10^{12} kg CO_2-eq/yr
2000	8.0	29.3	2010	2.7	9.9
2001	7.0	25.7	2020	2.1	7.7
2002	6.0	22.0	2030	1.8	6.6
2003	5.0	18.3	2040	1.6	5.9
2004	4.0	14.7	2050	1.4	5.3
2005	3.5	12.8	2060	1.3	4.8
2006	3.3	12.1	2070	1.2	4.4
2007	3.1	11.4	2080	1.1	4.2
2008	2.9	10.6	2090	1.0	3.9
2009	2.8	10.3	2100	1.0	3.7

Because of the importance of the global warming risk potential, a large number of investigations are expected to be conducted in the near future. These new findings will obviously influence the view of the IPCC and other scientists. For example, the new findings may lead to a more stringent upcoming greenhouse gas limitation and mitigation policy, which is currently debatable due to the scientific uncertainties. More refined climate models may also lead to the revision of greenhouse gas emission and mitigation scenarios. The damage assessment may be refined and a consensus on threshold level may be identified. As a result of these new findings, the carrying capacity may need to be refined accordingly. Future IPCC assessment reports are a source of information that will be useful for the refinement of the carrying capacity.

One factor that may be considered for this carrying capacity estimate is any future international protocol on GHG limitation. The emission level established by any such international agreement may not account for the natural absorption capacity, but it would reflect the political, social, and practical basis that is acceptable to the LCA community.

CHAPTER 6

STRATOSPHERIC OZONE DEPLETION IMPACT

6.1 Overview

Stratospheric ozone depletion is the thinning of the ozone layer in the stratosphere, the region between 10 to 50 kilometers above the earth's surface. About 90% of the atmospheric ozone is in the stratosphere while the remaining 10% is in the troposphere, residing in the surface to 10 kilometers in the atmosphere (WMO 2003). A thin ozone layer allows more radiation, especially ultraviolet-B, to reach the earth and cause adverse impacts. For humans, overexposure to ultraviolet can lead to skin cancer, eye damage, and a weakened immune system (Slaper et al. 1996; 1998; Caldwell et al. 2003; De Gruijl et al. 2003). For ecosystems, ultraviolet exposure can lead to the reduction of crop yield and damage to both terrestrial and aquatic ecosystems (Hansen et al. 2001; Velders et al. 2001; Hader et al. 2003). Ultraviolet also causes damage to air quality and outdoors materials such as woods and plastics (Andrady et al. 2003; Solomon et al. 2003; Zepp et al. 2003). In contrast to global warming caused by GHG emissions, Shindell (2001) pointed out that ozone depletion had cooled the global temperature by about 0.1 °C.

Stratospheric ozone depletion is caused by the presence of chlorofluorocarbons (CFCs) and other ozone-depleting substances (ODSs) in the stratosphere. Chlorine and bromine are the primary atmospheric halogens responsible for the depletion of stratospheric ozone, with bromine nearly 50 times more potent than chlorine for destroying ozone (Daniel et al. 1999; WMO 2003). The only known natural sources of ODSs are the emissions of methyl bromide (CH_3Br) from the oceans and biomass burning. However, if the natural methyl bromide emissions were the sole source of ODS, the ozone would not be depleted (WMO 1998b). The formation of ozone and its destruction by chlorines and bromines in the stratosphere can be expressed by the following elementary reactions, in which Z denotes chlorine or bromine and *hv* denotes ultraviolet light (Chapman 1930; Molina and Rowland 1974; Wofsy et al. 1975):

$$O + O \rightarrow O_2 \tag{6.1}$$

$$O + O_2 \rightarrow O_3 \tag{6.2}$$

$$O + O_3 \rightarrow 2O_2 \tag{6.3}$$

$$O_3 \rightarrow O_2 + O \tag{6.4}$$

$$2O_3 \rightarrow 3O_2 \tag{6.5}$$

$$ZO + O \rightarrow Z + O_2 \tag{6.6a}$$

$$Z + O_3 \rightarrow ZO + O_2 \tag{6.6b}$$

$$O_3 + O \rightarrow 2O_2 \tag{6.6c}$$

$$HO_2 + ZO \rightarrow HOZ + O_2 \tag{6.7a}$$

$$HOZ + hv \rightarrow OH + Z \tag{6.7b}$$

$$Z + O_3 \rightarrow ZO + O_2 \tag{6.7c}$$

$$OH + O_3 \rightarrow H_2O + O_2 \tag{6.7d}$$

$$2O_3 \rightarrow 3O_2 \tag{6.7e}$$

Recently, stratospheric chlorine from ODSs has been declining. On the other hand, bromine is increasing because of its continued production as allowed under the Montreal Protocol (WMO 2003), an international agreement designed to protect the ozone layer. Montzka et al. (2003) conclude that the trends for currently observed total halon concentrations are increasing, which are consistent with the halons scenarios developed according to the Montreal Protocol. Despite this, the projections suggest that the accumulation of stratospheric halons will be decreasing in the coming years. In the tropics, ozone levels are typically between 250 and 300 DU[1] year round. An ozone hole, which is defined as a section of the atmosphere with an ozone layer less than 220 DU, has been observed in the Antarctic. A recent study by Shindell and Faluvegi (2002) shows that ozone depletion in this century may have been 50% more than previously acknowledged.

Hauschild and Wenzel (1998) estimated the environmental carrying capacity using a sustainability emission level of ODSs. They assumed that the sustainability emission level was the atmospheric concentration of chlorine that would not cause ozone holes over the Antarctic. They used the atmospheric chlorine concentration levels for the year 1970, which was 1.5-2.0 ppbv, as the critical level. The year 1970 was a pre-CFC period and the ozone holes had not been observed at that time. However, since the emission and consumption inventory of ODS was not available for 1970, they estimated the ODS emissions by assuming a mutual ratio between the atmospheric concentrations and the emission rates for the years 1970 and 1990. Using this mutual correlation, the uniform sustainable level of ODS emission was determined from the estimated 1970 global emission rate which was 281×10^6 kg CFC11-eq/yr.

For EI95, Goedkoop (1998) derived the critical emission level of ODSs according to the Montreal Protocol. A reduction factor of 100 was used to estimate the critical level from the 1990 emission level. This factor was used to ensure that the probability of death

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[1] 100 DU (Dobson Unit) is equal to 1 mm. of ozone layer at the standard temperature and pressure.

would be less than one case per million people per year. At this impairment level, a European target value of 4.6×0^6 kg ODP/yr[2] was calculated from the European 1990 emission of 4.6×10^8 kg ODP/yr.

The carrying capacity for stratospheric ozone depletion impact can be estimated following the threshold-oriented approach in a similar manner to the one noted by Hauschild and Wenzel (1998), based on the emissions agreed upon in the Montreal Protocol. This chapter presents the background of the Montreal Protocol and its amendments and adjustments and the ODS production scenario which can be adopted as the time-dependent carrying capacity. Like the global warming impact, the carrying capacity for stratospheric ozone depletion impact may have been exceeded already. As a result, the threshold level can be determined from a level that allows recovery from the damage due to past ODS emissions. Also in this chapter, the possible endpoint indicators are reviewed, a threshold level is specified, and the carrying capacity is determined.

6.2 The Montreal Protocol

Ozone depletion was first noticed by the NOAA and NASA in the 1970's. Since then, federal agencies have become increasingly aware of the effect. The first mitigation measure was issued in 1975 by the state of Oregon to ban CFCs in aerosol sprays (NSC 2000). The Ozone Protection Amendment was passed as part of the Clean Air Act in 1977. Also in 1977, UNEP hosted the first international meeting on ozone depletion. In 1987, the Montreal Protocol called for a freeze on the production and use of halocarbons at 1986 levels by mid-1989, and over the next 10 years a reduction in CFC production by half (UNEP 2000a). However, there were still concerns that the measures called for in

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[2] ODP denotes ozone depleting potential which is often referred to as CFC11-equivalent.

the protocol were insufficient. The following amendments and adjustments have been made (London-1990, Copenhagen-1992, Vienna-1995, Montreal-1997, and Beijing-1999) to shorten the time span for the ozone layer to recover. Basically, the Montreal Protocol and its amendments and adjustments have projected the ODS emission scenario based on the remaining stock and the necessity of use. Figure 6.1 presents the emissions of major ODSs projected by the Montreal Protocol and its amendments and adjustments.

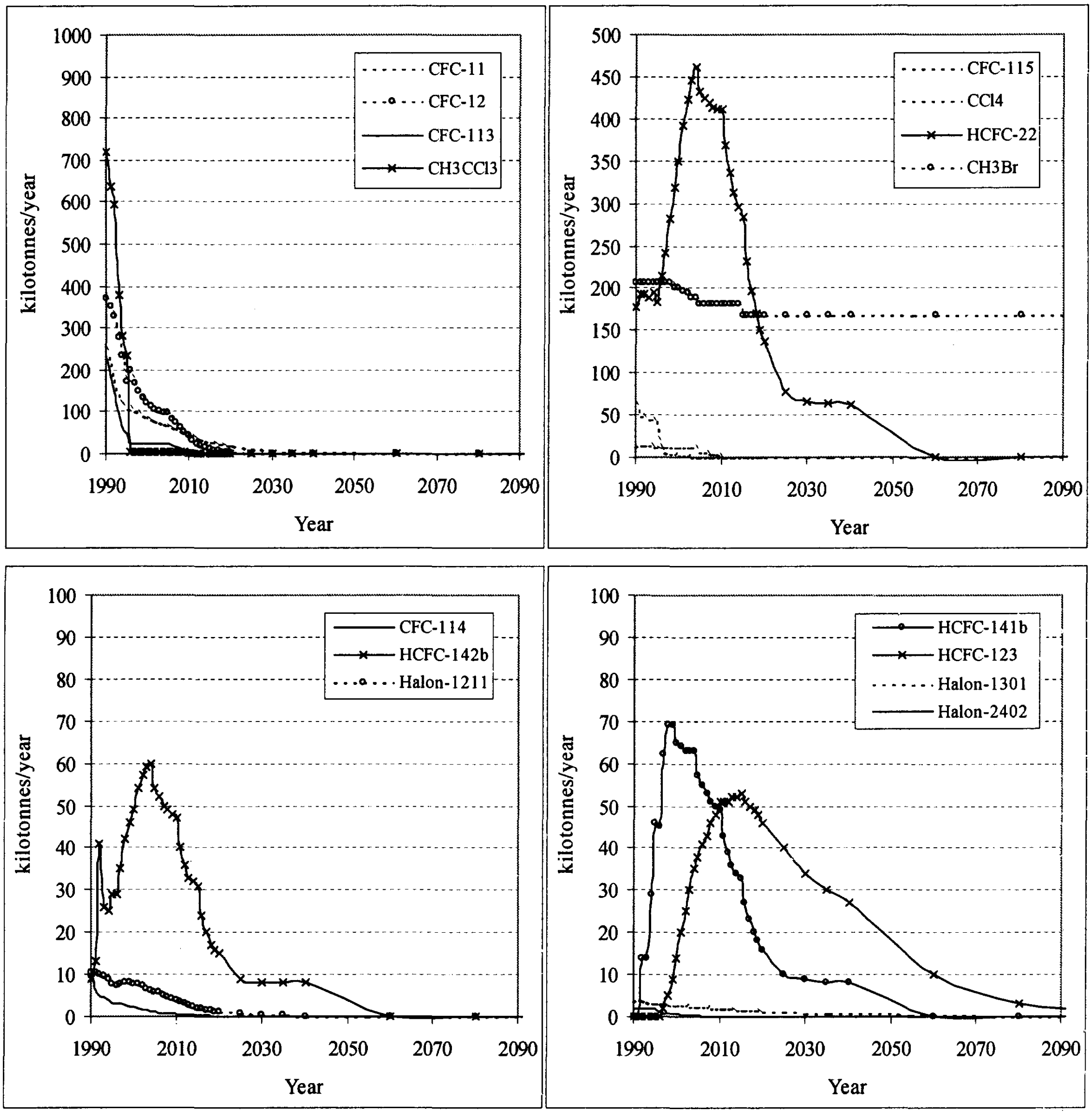

Figure 6.1 The emissions of major ODSs projected by the Montreal Protocol and its following amendments and adjustments.
(Source: WMO 1998a)

The WMO has defined the meaning of "recovery" by indicating that full recovery occurs when chlorine loading returns to the pre-1980 level of 2 ppbv, which is considered to have been the loading when the Antarctic ozone hole was first apparent (WMO 1998a). Recently, it has been found that there has been a response (recovery), from the peak of ozone depletion in 1994, as a result of the ODS cuts following the ratification of the Montreal Protocol (WMO 2003). According to the emission projection agreed to in the Montreal Protocol, the use of ODSs will be phased out with the expectation that the ozone layer can recover to the pre-CFC period by 2050 (Rosenfield et al. 2002; WMO 2003). If ODSs continued to be emitted without the control of the Montreal Protocol, the ozone depletion would be 10 times worse than the current level (WMO 1998a).

However, it should be noted that many factors, in addition to halogen loading, influence ozone distribution in stratosphere. For example, the atmospheric concentrations of many gases are changing, with consequent impacts on ozone transport, temperature, and chemistry (Shindell 2001; Randeniya et al. 2002). As a result of a variety of influences, different time frames for ozone recovery are estimated. Some studies indicated that the ozone layer would recover fully after 2050 (Dvortsov and Solomon 1998; Montzka et al. 1999; Shindell et el. 1998; Shindell 2001) or that the recovery would even be delayed until after 2100 (Randeniya et al. 2002). On the other hand, some studies suggested that recovery may be reached before 2050 (Nagashima et al. 2002; Schnadt et al. 2002).

6.3 Threshold Level for Carrying Capacity Estimate

It should be noted that the earth's carrying capacity to absorb the emissions of ODSs may have been exceeded already since the start of the CFC-period. Therefore, environmental conditions in the pre-1980 levels can be considered as the stabilization level or the threshold level for estimating the carrying capacity.

An issue for measuring the impacts associated with ozone depletion is the selection of an endpoint indicator. A common measure allows consistent discussion about the relationship between ozone depletion and impacts. Some possible indicators include: the ultraviolet index, the thickness of the ozone layer, the size of the ozone hole, and the tropospheric and stratospheric chlorine/bromine loadings. Among these indicators, ultraviolet seems to be less used because ultraviolet changes are not caused by ozone depletion alone (McKenzie et al. 1999; 2003). The thickness of the ozone layer is typically expressed in terms of Dobson Unit (DU) (Shindell et al. 1998). The size of an ozone hole is the area mainly in the Antarctic where the ozone layer is thinner than 220 DU (Austin et al. 2003). Tropospheric chlorine/bromine loading is the approximation of ODS abundance in the troposphere, which relates to the future loading for the stratosphere (CMDL 2001). Stratospheric chorine/bromine loading is basically the measure of the ODS in the stratosphere as is described more fully below (Daniel et al. 1995; Montzka et al. 1996; Wuebbles and Kinnison 1996).

The chlorine/bromine loading is an indicator that is widely used in ozone depletion studies (WMO 2003). This indicator is often expressed in terms of equivalent effective stratospheric chlorine, denoted EESC (Daniel et al. 1995; Montzka et al. 1996; Wuebbles 1996). EESC is an index developed to represent the potential damage caused by a given mixture of ODS to stratospheric ozone. EESC can also be thought of as the mechanism by which the effect of increasing chlorine/bromine loading results in decreasing ozone concentrations.

6.4 Carrying Capacity Estimate

The economic carrying capacity for ODS emissions can be estimated by assuming that ozone depletion is the sole effect from ODS emissions. So, if the human-induced ODS emissions are limited to the pre-1980 level, the ozone layer will be allowed to recover to the pre-1980 level as well. The ultimate goal of the ozone depletion issue is to pose no further damage, which means zero emission of human-induced ODS should be achieved in the future. In such a case, the carrying capacity of ODS emissions should be assumed to be zero emission and therefore any facility that releases an ODS can be considered to be non-sustainable. Practically, there are still some ODSs remaining in storage and in use for essential purposes such as medical and agricultural applications.

In an effort to achieve the zero emission goal for human-induced ODS emission, international agreements must be maintained and regulations must promulgated to ensure that these ODSs will be replaced by non-ODSs eventually. According to the Montreal Protocol and its amendments and adjustments, ODSs are scheduled for production cuts, emission cuts, freezes, and eventual phasing out sequentially by specific chemicals on the part of all signatory countries. The carrying capacity for human-induced ODS emissions is therefore adapted from the amounts projected by the Montreal Protocol and its amendments and adjustments. This projection will allow the damages to recover within a desirable time frame, which is 2050 in this case.

From the LCIA perspective, ozone depletion impact is considered to be a homogeneous impact category in which several burdens can be aggregated due to their additive effect to ozone depletion impact (Assies 1998). For the carrying capacity estimate, the emissions reported in the Montreal Protocol (Figure 6.1 shown before) must be aggregated to become a single value relative to a reference substance. This can be carried out using ozone depletion potential (ODP), which is a relative measure that compares the expected impact on ozone per unit of mass emission of an ODS to the impact of the same unit mass of CFC-11, integrated over time (Wuebbles 1983; Daniel et al. 1995). OPDs are summarized in Table C.2 Appendix C (UNEP 2000b). ODPs have units of kg CFC-11/kg ODS_i, where i represents a species of ODS. A simple equation is used to characterize the amount reported in kg of ODS relative to kg of CFC11-eq, the reference ODS widely used in LCIA (Goodkoep 1998; Goedkoop and Spriensma 2001; Guinee 2001; Hauschild and Wenzel 1998; Pennington et al. 2000):

$$kg\ CFC11\text{-}eq = ODP_i \times kg\ ODS_i \tag{6.8}$$

Figure 6.2 depicts the time-dependent carrying capacity calculated from the emissions reported in the Montreal Protocol. The sensitivity of endpoint indicators according to this emission scenario is presented in Figure 6.3 and Figure 6.4.

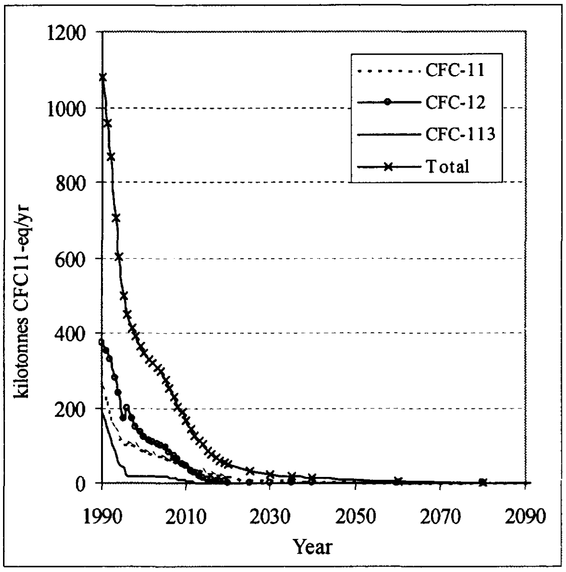

Figure 6.2 Carrying capacity for ODS emissions estimated from the Montreal Protocol.

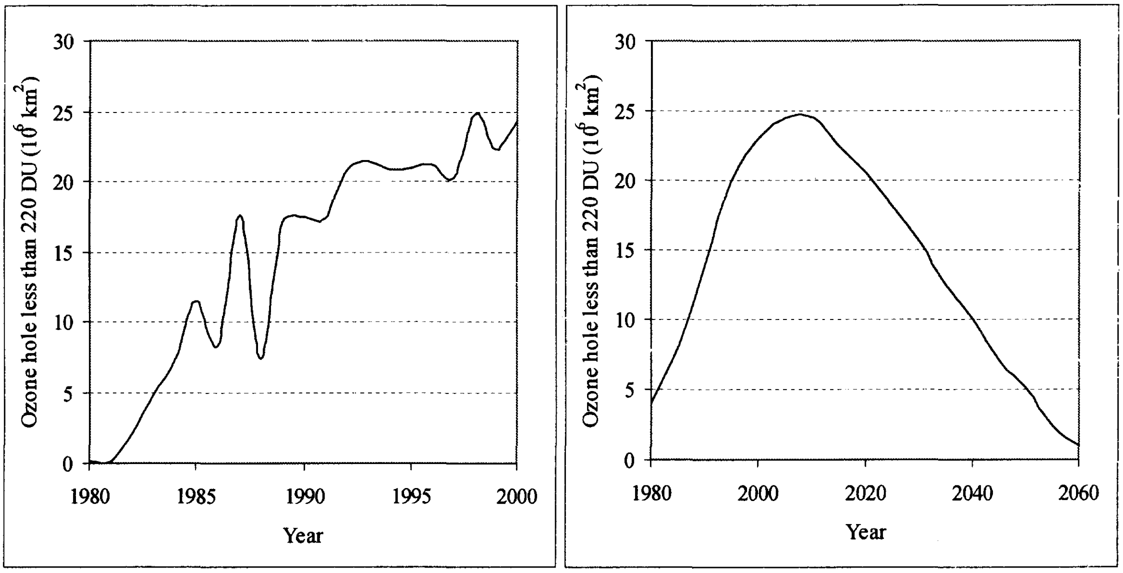

Figure 6.3 Ozone hole area at the Antarctic. Left panel is the observed data and right panel is the projection according to the emissions delineated in the Montreal Protocol. (Source: WMO 2003; Austin et al. 2003)

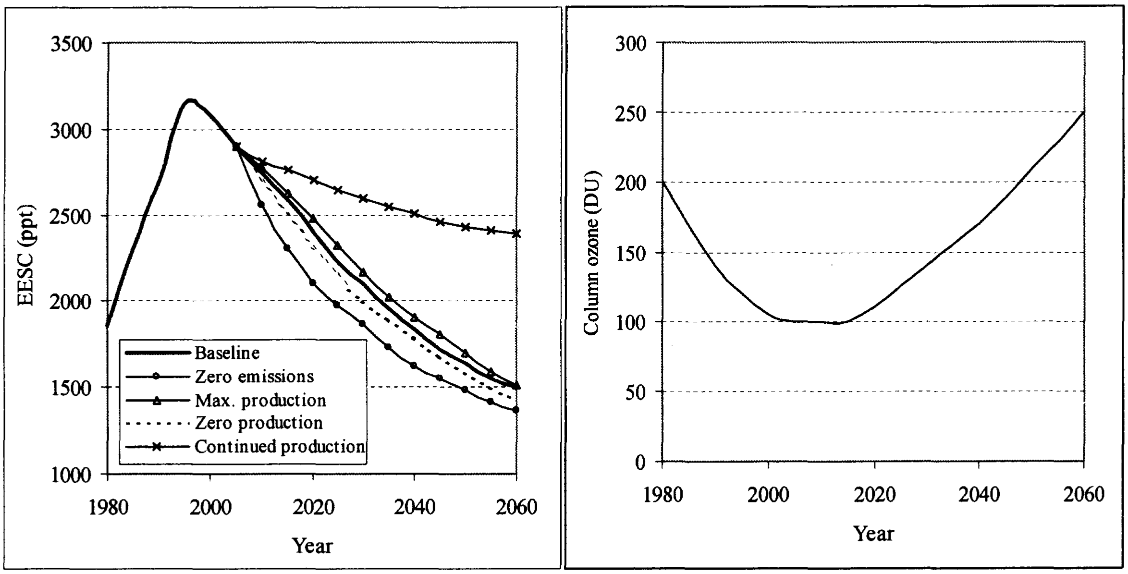

Figure 6.4 Projection of equivalent effective stratospheric chlorine, EESC, (left panel) and column ozone at the Antarctic (right panel) according to the scenarios following the Montreal Protocol.
(Source: WMO 2003; Austin et al. 2003)

6.5 Summary

The carrying capacity is estimated by assuming that ozone depletion is solely an effect of ODS emissions. If the ODS emissions are limited to the pre-1980 level, the ozone layer will be allowed to recover to the pre-1980 level as well. The Montreal Protocol and its amendments and adjustments stipulate that ODSs be scheduled for production cuts, emission cuts, freezes, and eventual phasing out. Analogous to the carrying capacity for global warming impact, the time-dependent carrying capacity for stratospheric ozone depletion impact is adapted from the amounts projected by the Montreal Protocol. In this research, the carrying capacity is estimated for up to the year 2100 (Table 6.1).

Table 6.1 Carrying Capacity for Ozone-Depleting Substance Emissions

Year	kilotonnes CFC-11/yr (10^6 kg CFC-11/yr)	Year	kilotonnes CFC-11/yr (10^6 kg CFC-11/yr)
2000	348	2010	169
2001	330	2020	48
2002	319	2030	23
2003	307	2040	15
2004	298	2060	3
2005	276	2080	1
2006	251	2100	1
2007	228		
2008	204		
2009	187		

The projected emission of ODS may not be the theoretical "sustainable level" because it takes into account several practical factors. The carrying capacity for the emission of ODS should be revised accordingly in line with other future findings. Recently, UNEP and WMO have been cooperating in addressing the ozone depletion issue. As a result, scientific assessment reports have been published and updated periodically (by NOAA, NASA, UNEP, WMO, and EC). Future research supported by these organizations and agencies as well as the political issues related to ozone emission limitation are sources that will be useful for the refinement of the carrying capacity.

CHAPTER 7

ACIDIFICATION IMPACT

7.1 Overview

Acidification is a decline in the ability of ecosystems to neutralize acid deposition, which in turn lowers the pH of lakes and soils. Susceptibility to acidification is affected by the level of acid deposition, and by the natural buffering capacity of the systems in question. Acidification in lakes causes the dissolution of soil minerals, lowered dissolved organic matter due to coagulation and sedimentation, deposition of sulfate ions along with ammonia and nitrate ions, which provide a certain degree of nitrogen enrichment in lakes (Gorham 1998). Acidification can diminish the ability of lakes and streams to sustain the survival of fish and aquatic species. Calcium and magnesium are depleted from forest soil; saturated nitrate is leached out to streams and lakes (Gorham 1998; Mill 2001). Acidification also impairs visibility and causes the erosion of rocks and manmade materials. Acidification is a serious problem in industrialized regions of the eastern U.S., Europe, the former Soviet Union, Southeast Asia, and China (Kuylenstierna, et al. 2001; Bouwman et al. 2002). Galloway (1995) has predicted that in the future Asian emissions of acidifying compounds will equal or exceed those of Europe and North America combined.

The main causes of acidification are airborne sulfur and nitrogen compounds emitted by anthropogenic sources (Galloway 2001). The largest source of anthropogenic sulfur emissions is sulfur dioxide (SO_2) emission caused by combustion of fossil fuels. Natural sources of sulfur emissions include volcanoes and oceans. The most important anthropogenic nitrogen emissions are nitrogen oxide (NO) and nitrogen dioxide (NO_2), together denoted as NO_X (Bouwman et al. 2002). The major sources of nitrogen are fossil fuel combustion, biomass burning, and microbiological gas generation in soils (Oliver et al. 1998). In the atmosphere, these pollutants undergo interaction with atmospheric moisture to become acids and impact the terrestrial and aquatic ecosystems through wet deposition (rain, snow, fog, and mist) as well as dry deposition (acidic dust and particles). Acid deposition is the transfer of strong acids and acidifying compounds from the atmosphere to the earth's surface (Driscoll et al. 2001).

From the LCIA perspective, Hauschild and Wenzel (1998) determined the environmental carrying capacity from the pre-1890 emission levels of 6.2×10^7 kg SO_2/yr for Denmark and 2.42×10^9 kg SO_2/yr for Europe. These values were obtained using a model to determine the deposition scenarios for acidification of forest systems in Scandinavia. The results showed that the area that did not exceed the critical load (discussed further) for any forest system during the pre-industrial scenario from around 1840 (Sverdrub et al. 1990 as cited in Hauschild and Wenzel 1998).

The target load for EI95 was calculated according to its impact on the ecosystem. In EI95, it was assumed that the European ecosystem impairment was estimated to be less than 5 % if the acidifying substance emissions were reduced by a factor of 10 (Goedkoop 1998). As a result, the European target value of 5.6×10^9 AP kg/yr[1] was used in EI95. Dickinson et al. (2001) subsequently estimated a value of U.S. carrying capacity for acidification impact by multiplying the normalized EI95's target value by the U.S. land area resulting in a carrying capacity value of 1.2×10^{10} AP kg/yr.

Potting et al. (1998) and Huijbregts et al. (2001) attempted to take into account the spatial variation in the characterization of acidification potential. A regional air

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[1] AP = Acidification potential (equivalent to SO_2)

transport model was used to determine the source-receptor relationship. The site factors were assigned according to the contribution of locally specific emissions to the sensitivity of the European ecosystems. A similar method has been used for the characterization of acidification potential of U.S. emissions (Bare et al. 2002; Norris 2002; EPA 2003a). However, the U.S. characterization factors do not take into account the sensitivity of the ecosystems because the information is not as well documented as in Europe.

This chapter presents background on the international and U.S. policies on acid deposition control, as well as a feasible method for determining carrying capacity estimates. Estimating the carrying capacity for acidification impact from the emission target levels is also elaborated. A method to estimate a region-specific carrying capacity is developed as well as a method to include site factors in the characterization of acidification potential.

7.2 Acid Deposition Control Policies

7.2.1 International Policies and the Critical Load Approach

European countries have attempted to address acidification problems since the 1960s. The original concept of using a uniform maximum allowable mass deposition rate of 20 kg wet SO_4^{2-}/ha/yr was first recognized as an interim target load (Nilsson and Grennfelt 1988). Subsequently, site-specific critical loads were increasingly used. During the 1970s, the hypothesis was confirmed that air pollutants could travel several thousands of kilometers before deposition and damage occurred. This necessitated cooperation at the international level to solve the acidification problem. As a response, a Convention on Long-Range Transboundary Air Pollution (CLRTAP) was first held in 1984 supported by

European governments, the European Community (EC), and the United Nations Economic Commission for Europe (UNECE). In 1985, member countries implemented the First Sulfur Protocol calling for a 30% reduction from 1980 emission levels. Through this convention, member countries also agreed that the critical load approach discussed below provided an effective scientific approach for devising strategies for the abatement of acid deposition. Moreover, these countries set long-term goals for meeting the critical load for SO_2 and NO_x.

A critical load was defined by the convention as the maximum amount of acidifying deposition an ecosystem can tolerate in the long-term without being damaged (UN 1998). The concept behind critical loads is based on a dose-response relationship where damages to an ecosystem are caused by a certain load of pollutant. The systems with the highest sensitivity can only be protected at the critical load level. Steps involved in defining and implementing critical loads usually include 1) resource identification and characterization, 2) identification of regions or functional subregions, 3) characterization of deposition within subregions, 4) definition of assessment endpoints, 5) selection and application of models, and 6) mapping projected environmental responses (Strickland et al. 1993). A target load may be less rigorous than the critical load because it incorporates social, policy, economic, and related considerations along with scientific findings (EPA 1995a; Gorham 1998). Therefore, target loads can be either higher or lower than the critical load values. Target loads are used in order that emissions can be reduced accordingly to meet the targets and limit the amount of damage.

The critical load approach has been applied to strategies for emission reduction under two sulfur protocols of the CLRTAP: the 1994 Oslo Protocol on Further Reductions of Sulfur Emissions and the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (UNECE 2003). The aim of the protocols is to control and reduce sulfur emissions in order to protect human health and the environment from adverse effects, in particular, acidifying effects. The participating states agreed that the deposition of oxidized sulfur compounds in the long term would not exceed critical loads for sulfur. The sulfur protocols set emission reduction targets and timeframes based on the country's specific SO_2 emission baseline and contribution to the regional impact on the sulfur load. European critical load maps for sulfur provided the analytical tool for establishing emission thresholds and reductions goals in the ECE region (Gregor et al. 2001; Grennfelt 2001 et al.). Figure 7.1 depicts the use of critical load maps in the abatement strategies.

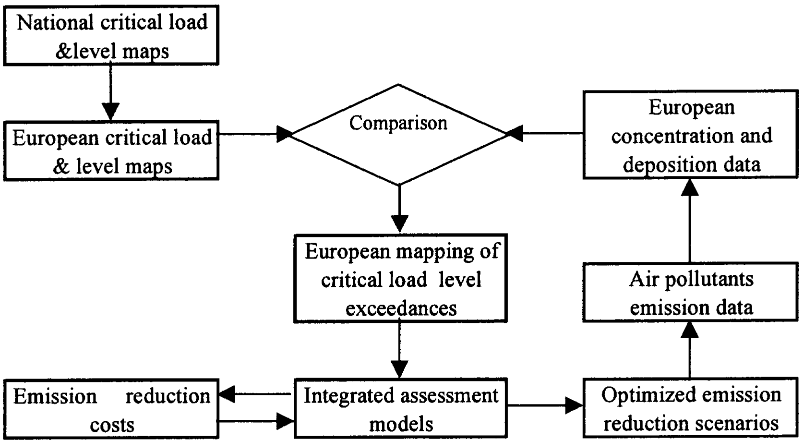

Figure 7.1 Use of critical load maps in abatement strategies.
(Source: Gregor et al. 2001)

Recently, 20 countries of the UNECE region, including Canada, the U.S., and EU have ratified the sulfur protocol. European policy makers have proposed that critical loads should not be exceeded anywhere in Europe by 2015 (Skeffington1999). Other than Europe and Canada, the critical loads have been studied and mapped across Asian countries such as China (Duan et al. 2000; Shah et al. 2000; Hao et al. 2001a; 2001b; Tao and Feng 2001; Ye et al. 2002), India (Satsangi et al. 2003), Japan (Fumoto et al. 2001; Hayashi and Okazaki 2001), South Korea (Park and Lee 2001), Russia (Semenov et al. 2001), and Thailand (Milindalekha et al. 2001). There is also a plan to map the critical load for developing countries in Asia, southern Africa and Latin America (Hicks et al. 2001). There are numerous methods that are available for estimating critical loads; the steady-state mass balance approach (SSMB) (Posch et al. 1999: Henriksen and Posch 2001) is the most widely used one (Skeffington 1999; Aherne and Farrell 2002).

Basically, scientists run models that take into account soil chemistry, rainfall, and topography to calculate the amount of acid substance that can be absorbed before the system reaches critical levels. In order to obtain values for the critical loads, an ecosystem has to be chosen and then a suitable indicator is selected to represent the ecosystem. A chemical limit is subsequently defined as the concentration at which the critical level of indicator is exceeded. In forests the indicators may be plants, and in freshwaters they may be aquatic biota. The magnitude of a critical load depends on the characteristics and conditions of the target ecosystem and receptor, e.g., the buffer capacity at a given location. Thus critical loads differ locally reflecting specific conditions that exist there.

The critical loads can be very low for a very sensitive ecosystem and higher for a more tolerant one. There are also some uncertainties associated with indicators, observed data, and approaches used for the critical load estimate (An et al. 2001; Hall et al. 2001). In Europe, the maximum critical loads for sulfur vary from less than 200 eq/ha/yr to more than 1,500 eq/ha/yr (3.2-24.0 kg S/ha/yr)[1]. For nitrogen, it varies from less than 200

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[1] 16 kg S/ha/yr = 1,000 eq/ha/yr

eq/ha/yr to more than 1,000 eq/ha/yr (2.8-14.0 kg N/ha/yr)[1] (Gregor et al. 2001). There has been less than 5% exceedance of the acidity critical load across Europe recently; decreasing from about 30% in 1980, due to effective measures in place. In the eastern Canada, the critical loads of different watersheds vary from less than 8 kg SO_4^{2-}/ha/yr to more than 20 kg SO_4^{2-}/ha/yr (2.7-6.7 kg S/ha/yr)[2] (Fenech 1998; Ouimet et al. 2001).

In the U.S., a survey was conducted to investigate the acidity in some eastern lakes in the late 1980s. Critical loads for sulfate deposition were calculated for some of these lakes based on the criteria of maintaining pH above 5.3 (Table 7.1). There was also a study to estimate critical loads for Maryland streams (Janicki et al. 1991). In this study, the critical load is defined as the rate of sulfur and nitrogen deposition that results in stream pH less than the critical pH values of 5.3 to 6.5. Critical load for the Appalachian Plateau, Coastal Plain, and some portions of the Blue Ridge Mountains is 0.5 keq/ha/yr (8 kg S/ha/yr). In contrast, the Valley and Ridge, Piedmont, and portions of the Blue Ridge regions have critical loads of more than 2.0 keq/ha/year (32 kg S/ha/yr). For two hardwood forests in New Hampshire and the Adirondack Mountains, critical loads for nitrogen deposition with respect to acidity ranged from 0 to 630 ea/ha/yr or 0 to 8.82 kg N/ha/yr (Pardo and Driscoll 1996).

Table 7.1 Estimated Critical Loads for Some U.S. Eastern Lakes
(Source: Henriksen and Brakke 1988)

Subregion	Critical loads (kg SO_4^{2-}/ha/yr)	Subregion	Critical loads (kg SO_4^{2-}/ha/yr)
Adirondacks	19-34	Maine	36-38
Catskills/Poconos	46-60	Northeastern Minnesota	62-63
Southern New England	27-35	Upper peninsula of Michigan	20-25
Central New England	46-57	Upper Great Lakes	31

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[1] 14 kg N/ha/yr = 1,000 eq/ha/yr
[2] 1 kg S/ha/yr = 96/32 kg SO_4^{2-}/ha/yr

Studies made in the early 1990s focused on the feasibility of adopting the critical load approach for U.S. atmospheric deposition calculations (Hicks et al. 1993; Holdren et al. 1993; Hunsaker et al. 1993; Strickland et al. 1993). A conceptual framework for adopting the critical load approach was laid down. However, in 1995, a report was prepared for Congress on the feasibility of adopting acid deposition standards (EPA 1995a). The report concluded that it was feasible to establish the standard, however, there were two critical areas of uncertainty in standard setting at that time, namely, policy decisions or goals for protecting sensitive systems and scientific uncertainties. Therefore, the effects of acid deposition have been the focus of the subsequent research rather than the calculation of the critical loads (Sullivan 2000). The study of critical loads in the U.S. is in its infancy stage because the current policy aims at reducing emissions rather than focusing on receptor-oriented considerations.

7.2.2 U.S. Policy

In 1980, Congress passed the Acid Precipitation Act. This act allowed a period of ten years to study and examine the relationships among pollutants, their sources and effects on the environment and human health. The National Acid Precipitation and Assessment Program (NAPAP) was established to coordinate and administer the study. Significant conclusions from the 1990 NAPAP Reports and Integrated Assessment are: 1) effects of acid deposition and its precursor emissions are broad, 2) acid deposition is a regional scale effect, and 3) the inventories show that two-thirds of SO_2 emissions and one-third of NO_x came from electric power generation (NAPAP 1991; 1998).

Following the report of the NAPAP, Congress passed the Federal Acid Deposition Control Program as Title IV of the Clean Air Act Amendments of 1990 (CAAA 1990).

The objectives of Title IV were threefold: 1) focusing on reducing the national SO_2 and NO_x emissions rather than relying on regionally variable deposition standards and state-by-state implementation plans 2) translating a 10 million short tons of SO_2 reduction goal into a nationwide cap on emissions from electric generating sources and allowing industry 20 years to achieve it, and 3) providing a new tool to achieve the reduction-an innovative market-based allowance trading program, where one allowance is a limited authorization to emit one ton of SO_2 (NAPAP 1998).

To comply with Title IV, reductions of 10 million short tons of SO_2 and 2 million short tons of NO_x, compared to 1980 levels were mandated (EPA 1997a). In contrast to the typical command-and-control approach to regulation, Congress adopted a market-based "Cap and Trade" approach (NAPAP 1998). The "Cap and Trade" approach in Title IV allowed industry some flexibility in using compliance methods. They could install pollution control equipment, switch fuel, conserve energy, rely more on renewable resources, trade SO_2 allowances, or any combination of these approaches. Implementation of Title IV was divided into two phases. Phase I, which lasted from 1995 to 2000, aimed to cut the emissions of the 110 major energy utilities in the Midwest region. Phase II, 2000-2010, was designed to cap the emissions of the other 2,000 utilities across the nation. The largest emission reductions will be in the highest emitting regions (Ohio, Indiana, West Virginia, and western Pennsylvania) (NAPAP 1998).

Recently, researchers have reported trends in pollutant emission data in an effort to evaluate the effectiveness of Title IV in solving the acid deposition problem. Shannon (1999) reported that the emission of SO_2 in the U.S. and Canada fell by about 28% from 1980 to 1995. Lynch et al. (2000) and Butler et al. (2001; 2003) concluded that a significant reduction of SO_2 emissions had been observed for most states in the eastern

U.S. However, only a few states showed a significant decline in NO_x levels. Kelly et al. (2002) reported that S and N concentrations and deposition declined for a period of 12 years (1988-1999) in southeastern New York. A survey by EPA showed that the water resource conditions in some sensitive areas have been improving (EPA 2003b). Table 7.2 presents the improvement of water resource conditions in some acid-sensitive regions subsequent to the implementation of Title V. These results support the conclusion that Phase I of Title IV has effectively reduced acid rain in the eastern U.S. However, during the early years of such a reduction program, observable responses by the environment may be minimal, mostly due to the inherent time lags for the ecosystems to recover.

Table 7.2 Improvement of Water Resource Conditions in Acid-Sensitive Regions
(Source: EPA 2003b)

Acid Neutralization Class		Number of Sites	Change in Acid Neutralization Capacity (ANC),(µeq/L)
Acidic	ANC < 0 µeq/L	26	+ 1.29
Low ANC	0 < ANC < 25 µeq/L	51	+ 0.84
Moderate ANC	25 < ANC < 200 µeq/L	43	+ 0.32

Region	Population Size	Acidic in the past		2002 Status	
		% Acidic	Time	Change in ANC (µeq/L/year)	% Acidic
New England	6,834 Lakes	5.6 %	1991-94	+0.3	5.5 %
Adirondacks	1830 Lakes	13.0 %	1991-94	+0.8	8.1 %
North Appalachians	42,426 km	11.8 %	1993-94	+0.7	8.5 %
Ridge/Blue Ridge	32,687 km	5.0 %	1987	-0.0	5.0 %
Upper Midwest	8,574 Lakes	2.9 %	1984	+1.0	0.9 %

7.3 Threshold Level for Carrying Capacity Estimate

The transport of acidic species is considered to be long-range in the scale of regional to sub-continental and continental scales depending on the moderate atmospheric lifetime of the pollutants. Thus, whenever regional critical loads are available, they will be used as the threshold levels for estimating region-specific carrying capacity. However, the critical loads for all U.S. regions are not available. The development of the critical loads across the U.S. would require a massive multi-year effort due to incomplete data, especially for soil chemistry. In the future, the mapping of critical loads may be carried out using one of the broad range of methods available as discussed in Subsection 7.2.1. The regional-specific carrying capacity can then be estimated from the critical loads by applying them in a long-range atmospheric transport model to investigate the source-receptor relationship.

In this research, the emission target levels, which are less rigorous than using the critical load approach, are selected as the approach for carrying capacity estimate instead. It is assumed for the carrying capacity estimate that the emission target levels according to Title IV of the CAAA 1990 are the spatially uniform desirable emission levels. It is expected that the emission projection according to Title IV would result in the mitigation of impacts to various degrees for the eastern U.S. For instance, sulfur deposition in some sensitive areas would be reduced by 30-50% in 2010 compared to the 1985 levels, the life span of sensitive cultural materials would be expanded by 30-40%, and the number of lakes unsuitable for aquatic biota would be reduced by about 10% (NAPAP 1991).

7.4 Carrying Capacity Estimate

7.4.1 Carrying Capacity Estimate

Should the critical loads be used, a source-receptor relationship would be required to determine the carrying capacity in terms of allowable emissions. The use of a long-range transport atmospheric model coupled with a critical load map would provide a regional scale source-receptor relationship. The adoption of the emission target levels thus eliminates this modeling step. According to Title IV, the 2010 target emission levels of

SO_2 and NO_x are limited at 10 and 2 million tons less than the 1980 levels respectively. Assuming that the 1980 emission levels of SO_2 and NO_x are approximately 26 and 27 million short tons per year, the constant U.S. carrying capacities are therefore 16 and 25 million short tons per year for SO_2 and NO_x emissions respectively. However, EPA estimated that actual emissions, especially NO_x, were less than the designated target levels after the enforcement has started (Figure 7.2).

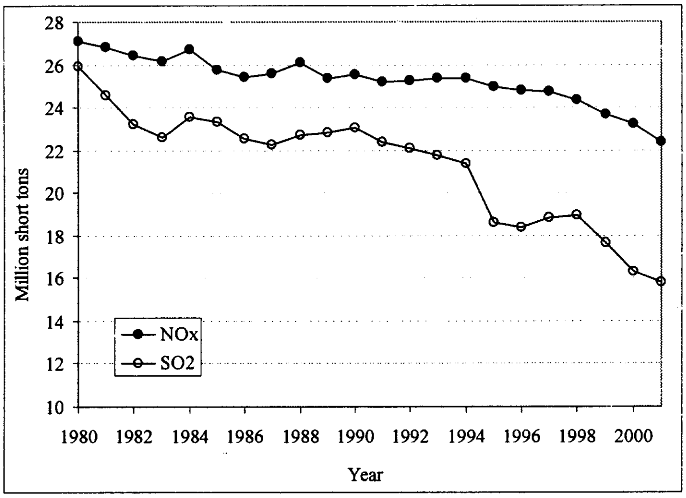

Figure 7.2 Actual national emissions of SO_2 and NO_x.
(Source of data: EPA 2003c)

From the LCA perspective, acidification impact is considered a homogenous impact category, in which several burdens can be aggregated due to their additive effect to the acidification impact (Assies 1998). Likewise, the emissions for both SO_2 and NO_x can be lumped together to become a single carrying capacity. This can be made using acidification potentials, denoted as AP (Hauschild and Wenzel 1998; Potting et al. 1998; Huijbregts et al. 2001a; Bare et al. 2002; Norris 2002). For the characterization, SO_2 is used as a basis for determination of acidification potentials. The method of establishing acidification potentials for acidifying substances is based on stoichiometric considerations and it is widely used in LCA studies (Hauschild and Wenzel 1998). AP values are summarized in Table C.3(a) Appendix C (Norris 2002).

Acidification potentials have units of kg SO_2/kg AS_i, where AS_i is the acidifying substance i. A simple equation is used to characterize the amount in kg of acidifying compound relative to kg of SO_2-eq, the reference acidifying compound widely used in LCIA (Goedkoop 1998; Hauschild and Wenzel 1998; Pennington et al. 2000; Goedkoop and Spriensma 2001; Guinee et al. 2001):

$$kg\,SO_2\text{-}eq \quad = \quad AP_i \times kg\,AS_i \qquad (7.1)$$

By using the above equation, a single carrying capacity for acidification impact can be determined (Table 7.3). Table 7.3 also presents the carrying capacity normalized by the U.S. land area (9.4×10^6 km^2). This figure is in the middle range of the critical loads for some U.S. eastern lakes, in the high end of the critical loads for Europe, and relatively high compared to the Canadian critical loads for some sensitive areas. This carrying capacity is equivalent to 9.34×10^{11} eq/year, which is about two times lower than the U.S. normalization value (2.08×10^{12} eq/year) derived from 1999 emission data for used with TRACI (Bare et al. 2006). This means that the current U.S. emissions already exceed the environmental carrying capacity.

Table 7.3 Summary of the Carrying Capacity for Acidification Impact

Target emission (year 2010) according to Title IV	U.S. carrying capacity			Critical load range (kg S/ha/yr)
	kg SO_2-eq/yr[1]	kg SO_2/ha/yr	kg S/ha/yr	
SO_2 16×10^6 short tons/year	3.04×10^{10}	32.3	16.2	U.S. eastern lakes 8.5 - 31.5
				Appalachian plateau 8.0
NO_2 25×10^6 short tons/year				Europe 3.2 - 24.0
				Canada 8 - 20

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[1] 1 short ton = 907.2 kg, AP_{NOx} = 0.7 kg SO_2/kg NO_x

7.4.2 Region-Specific Carrying Capacities

The spatial variation in terms of the receiving environment can be addressed by the assessment of region-specific carrying capacities. Generally, the causes of the acidification are twofold: the intrinsic low acid neutralization capability (i.e., the critical load) of an area and the high acid deposition to that area. Since not all of the regional critical loads for the U.S. are available, the current acid deposition rates are used as an indicator of the seriousness of the acidification problem to weight the region-specific carrying capacity. This approach assumes a uniform critical load across the U.S. Furthermore, it is also assumed that the acidification in the sensitive areas (e.g., the Adirondacks, the Southern Blue Ridge, Appalachian Mountains) is caused by the past cumulative acid load. This assumption implies that the load to the sensitive areas (where the acidification problem is serious) should be lowered to allow the ecosystems to recover. The field data on current acid load maps also supports this assumption. As can be seen from Figure 7.3, high acid loads are clustered along the northeast states, which are facing serious acidification problems.

U.S. states are divided into six regions for acidification classification (Figure 7.4): Northeast (EPA regions 1, 2, 3), Southeast (4), Midwest (5, 7), Southwest (6), Rocky Mountain (8), and Western (9, 10). The data on acid deposition were obtained from the National Atmospheric Deposition Program (NADP 2003). Total acid wet deposition is assumed to be the sum of sulfate and nitrate wet depositions. Region-specific factors can be determined from the distance-to-average multiplication/reduction factors using the equation:

$$RF_R = D_{avg}/D_R \qquad (7.2)$$

Where RF_R is a region-specific factor, D_{avg} is the U.S. average acid deposition, and D_R is the total acid deposition for region R. Table 7.4 summarizes the calculations of region-specific factors and region-specific carrying capacities for the U.S.

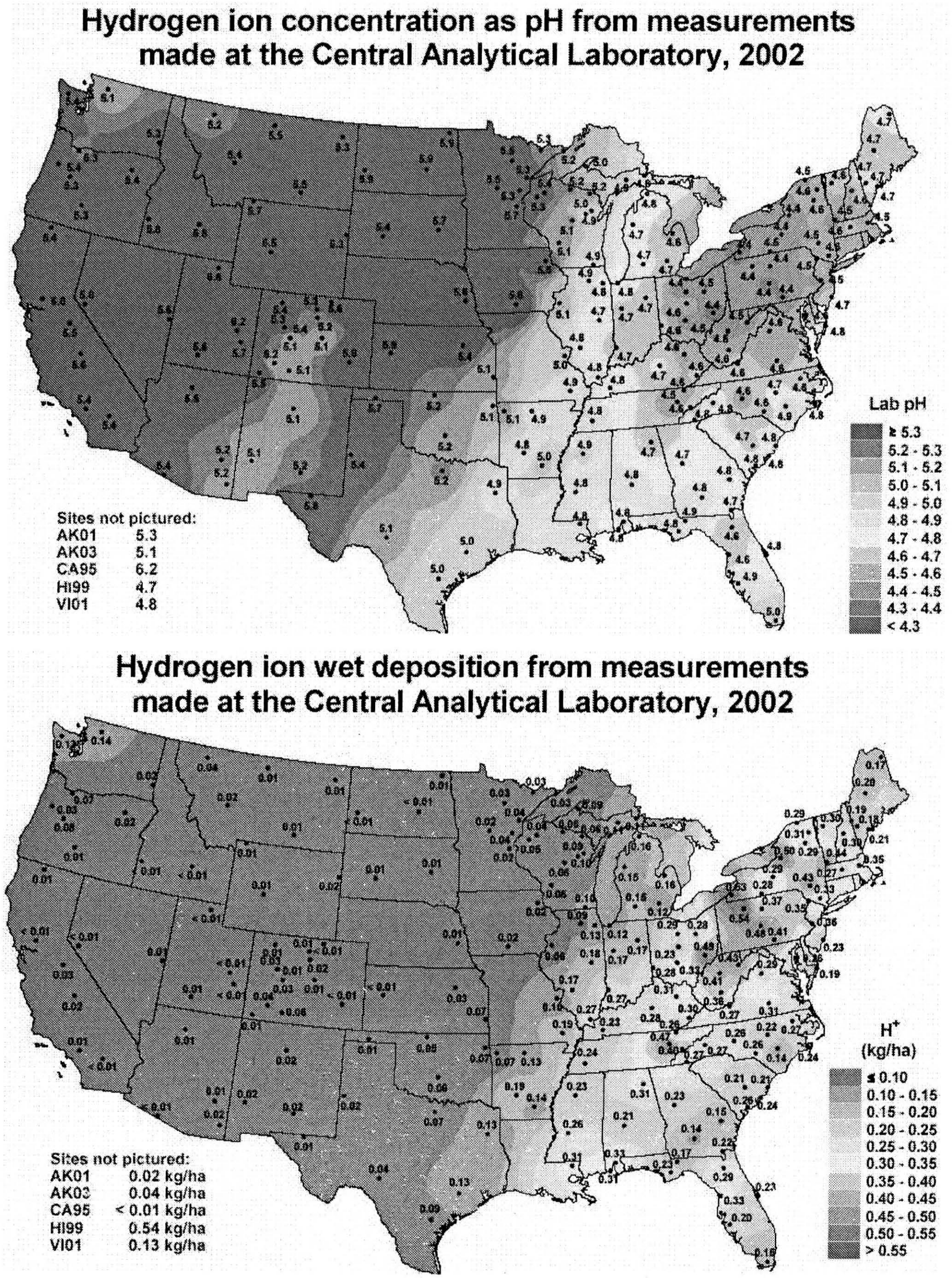

Figure 7.3 pH and acid loading maps.
(Source: NADP 2003)

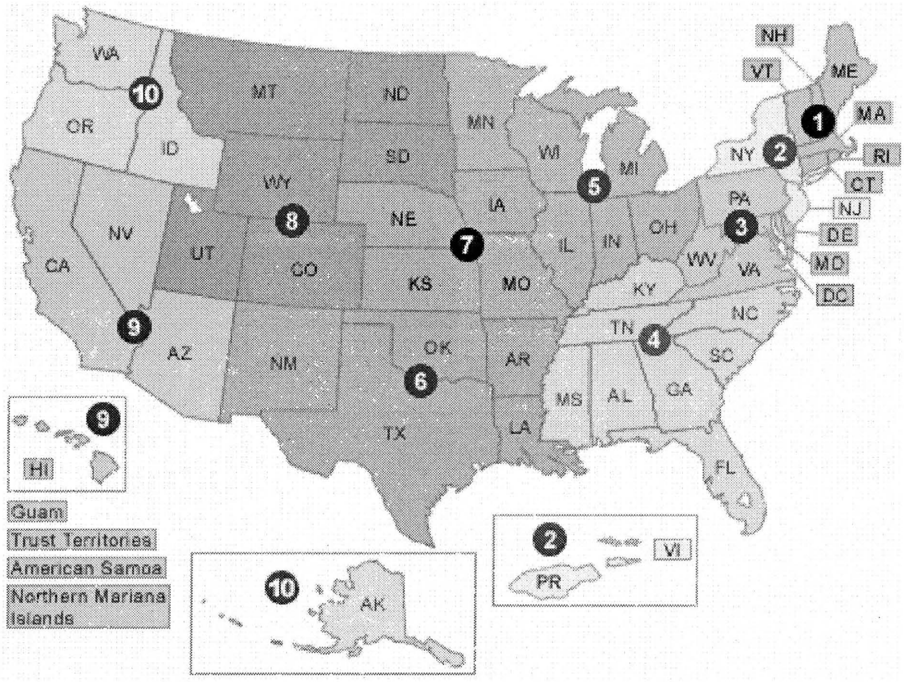

Figure 7.4 EPA regions.
(Source: EPA 2003d)

Table 7.4 Calculations of Region-Specific Factors and Region-Specific Carrying Capacities for the U.S.

Region	EPA Region	State
Northeast	1 2 3	CT DC DE MA MD ME NH NJ NY PA RI VA VT WV
Southeast	4	AL GA FL KY MS NC SC TN
Midwest	5 7	IA IL IN KS MI MN MO NE OH WI
Southwest	6	AR LA NM OK TX
Rocky Mountain	8	CO MT ND SD WY UT
Western	9 10	AK AZ CA HI ID NV OR WA

Region	Sulfate wet (SO_4^{2-}) kg/ha/yr	Nitrate wet (NO_3^-) kg/ha/yr	Ammonia wet (NH_3^-) kg/ha/yr	Total acid wet deposition (keq/ha/yr)[1]	Region-specific factor	Region-specific carrying capacity (kg SO_2/yr)
Northeast	16.9	13.9	2.7	0.734	0.61	1.86E+10
Southeast	14.4	10.0	2.2	0.590	0.76	2.31E+10
Midwest	12.3	11.7	3.8	0.668	0.67	2.04E+10
Southwest	8.1	8.1	2.2	0.428	1.05	3.19E+10
Rocky Mountain	2.0	3.6	1.2	0.170	2.64	8.02E+10
Western	1.7	2.1	0.6	0.105	4.30	1.31E+11
Average	9.2	8.2	2.1	0.449		

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[1] 1 keq = 48 kg SO_4^{2-} = 62 kg NO_3^- = 17 kg NH_3^-

7.4.3 Spatial Distributions of Emissions

Since acidifying substances (ASs) are transported and distributed on a regional scale, the spatial differentiation should be taken in account. Potting et al. (1998) and Huijbregts et al. (2001) have developed an approach for LCIA application to characterize AS emitted in Europe by using a long-range transport model coupled with the sensitivity of the receiving environment (e.g., the critical loads of ecosystems). For the U.S., a similar approach has been adopted for the Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts-TRACI (Bare et al. 2002; Norris 2002; EPA 2003a). However, the sensitivities of ecosystems cannot be considered because there is no regional database of receiving environments (Norris 2002). The site-dependent characterization factors (termed as site factors hereafter) in TRACI are basically the fate factors that take into account the transport of a pollutant under site-specific atmospheric pathways and chemistry processes. A higher site factor indicates more deposition of pollutant and more contribution to acidification impact on North America. The states along the eastern seaboard have lower site factors than the mid-continent states because some portions of pollutants are transported and deposited offshore (Norris 2002). By using the site factors, the overall site-dependent characterization for an emission of an AS can be modified from Equation (7.1) and is expressed as:

$$kg\ SO_2\text{-}eq = SF_{Acid} \times AP_i \times kg\ AS_i \qquad (7.3)$$

Where SF_{Acid} is a site factor for an emission of an AS from a U.S. site. Table C.3(b) in Appendix C presents the site factors for the state and regional levels. The average site factor for the U.S. is 1.0 and should be used where the actual site cannot be identified precisely.

7.5 Summary

The carrying capacity for acidification impact is estimated from the U.S. target levels according to Title IV of the Clean Air Act Amendments of 1990. Title IV requires the reduction of 10 million short tons per year of SO_2 and 2 million short tons per year of NO_x by 2010 compared to 1980 annual emission levels. Target levels for national SO_2 and NO_x emissions are estimated at 16 and 25 million short tons per year in 2010, respectively. It is expected that the emission projection according to Title IV of CAAA of 1990 would result in the mitigation of impacts to various degrees for the eastern U.S., where acidification is a serious environmental issue. For instance, sulfur deposition in some sensitive areas would be reduced by 30-50% in 2010 compared to the 1985 levels, the life span of sensitive cultural materials would be expanded by 30-40%, and the number of lakes unsuitable for aquatic biota would be reduced by approximately 10% (NAPAP 1991).

The emission levels according to Title IV are considered as the uniform carrying capacity and they can be aggregated to become a total allowable emission of 3.04×10^{10} kg SO_2/yr. This carrying capacity is about six times greater than the previous value used with the STM derived by Dickinson et al. (2001), which is 1.2×10^{10} SO_2 kg/yr and was based on the European calculation. Also this carrying capacity, when converted to 32.3 kg SO_2/ha/yr or 16.2 kg S/ha/yr, is in the middle range of the critical loads for some U.S. eastern lakes, in the high end of the critical loads for Europe, and is relatively high compared to the Canadian critical loads for some sensitive areas.

Acidification is considered as a regional impact. Region-specific carrying capacities are estimated by applying region-specific factors. The region-specific factors are weighted according to the seriousness of the acidification problem among regions using the current field data of acid wet deposition as the indicator. By applying the region-specific factors, the regions susceptible to acidification have lower carrying capacities than the more tolerant ones. The region-specific carrying capacities can then be calculated and summarized in Table 7.5.

Table 7.5 Carrying Capacities for Acidifying Substance Emissions

Region	Region-specific carrying capacity (kg SO_2/yr)
Northeast	1.86E+10
Southeast	2.31E+10
Midwest	2.04E+10
Southwest	3.19E+10
Rocky Mountain	8.02E+10
Western	1.31E+11
U.S. Average	3.04E+10

Site-dependent characterization factors (site factors) are also needed in order to characterize an emission of acidifying substances from a specific location in the U.S. Site factors are basically the fate factors that take into account the transport of a pollutant under the site-specific atmospheric pathways and chemistry processes. A higher site factor indicates more deposition of pollutant and more contribution to acidification impact on North America. By applying the site factors and chemical characterization factors, the overall characterization of acidifying substances can be calculated.

The carrying capacity for acidification impact can be refined to reflect changes in policy on acid deposition control and new scientific findings. Another means to refine the carrying capacity is the use of a long-range transport model coupled with a soil-water chemistry model. This approach is likely to be more accurate and sophisticated than simply using the target levels; but the use of this sophisticated approach is less feasible because it is very complex, costly, and time-consuming. However, it may be employed in the future should a more refined carrying capacity be needed.

CHAPTER 8

EUTROPHICATION IMPACT

8.1 Overview

Eutrophication is a process that can be defined as an increase in the rate of supply of organic matter to an ecosystem (Nixon 1995 as cited in Pinckney et al. 2001). Excessive fertility in surface waters results in heavy growth of undesirable weeds and phytoplankton, particularly of blue-green algae (Pitois et al. 2001). The endpoint of eutrophication, or nutrient enrichment, is depletion of oxygen associated with the decomposition of dead biomass. With the depletion of dissolved oxygen in water, there is a loss of biodiversity because some aquatic biota cannot survive (Pinckney et al. 2001) under very low or near anaerobic conditions. The crops of phytoplankton often clog the filters of water treatment plants and make the treatment of water more costly (Pitois et al. 2001). Some organic substances produced by the phytoplankton can pass through the filters at water treatment plants and cause unpleasant tastes and odors.

Some phytoplankton such as blue-green algae (*cyanobacteria*) can produce neurotoxins and hepatotoxins that are harmful to animals and pose serious health hazards to humans (Carpenter et al. 1998; Pitois et al. 2001). These toxins cannot be removed by conventional treatment and disinfection processes used in most public drinking water supplies. Eutrophication does not only pose a health hazard, but also affects the aesthetic quality and the supply of water, as well as the use of water for fisheries, industry, and recreation (McDowell et al. 2003). Degradation of water resources due to eutrophication can be considered as the loss of natural systems, their component species, and the amenities that they provide (Carpenter et al. 1998).

The essential elements needed by aquatic biota are C, H, O, some macronutrients, and trace elements. Carbon is the most important among these elements. However, carbon compounds are easily soluble in water and it is unlikely to be a limiting factor (Pitois et al. 2001). Hydrogen and oxygen are also present in waters at significant concentrations high enough to not be a limiting reactant. Phosphorus is often considered as the primary limiting factor in the growth of phytoplankton in surface waters (Carpenter et al. 1998; Mihalyfalvy et al. 1998; Drolc and Koncan 2002; Mainstone and Parr 2002; Turner et al. 2003). This is because P is typically in shortest supply among the essential elements in surface waters and generally has the greatest potential to limit plant growth (Finnveden and Potting 1999). An additional amount of P, if it is the limiting nutrient, will lead to increased growth, however, additional amounts of other nutrients will not since they are already in amounts in excess of the growth needs. An ideal stoichiometric atomic ratio of C:N:P at 106:16:1 (Redfield ratio), is widely accepted as the ratio for the growth of phytoplankton (Pinckney et al. 2001). According to this ratio, nitrogen is the secondary limiting nutrient for the eutrophication. Some studies, however, consider that N is the limiting nutrient for eutrophication in estuaries, coastal seas, and marine ecosystems (Carpenter et al. 1998; Tusseau-Vuillemin et al. 1998; Pahlow and Riebesell 2000; Pinckney et al. 2001; Tusseau-Vuillemin 2001; Mainstone and Parr 2002; Arhonditsis et al. 2003).

Eutrophication caused by excessive loadings of phosphorus and nitrogen nutrients is a major cause of water impairments in the U.S. (EPA 1996; 1999a; 2003c). Phosphorus and nitrogen inputs to surface waters come from both point and nonpoint sources. Nonpoint contributions are the dominant source of nutrient inputs causing eutrophication in the U.S. (Carpenter et al. 1998). Nonpoint P and N result primarily from agricultural and urban activities (Sharpley et al. 2001). Other nonpoint sources include runoff, septic tank leachate, atmospheric deposition, and other activities on land (Carpenter et al. 1998; Pitois et al. 2001). Point sources include wastewater effluent, storm sewer outfalls, and runoff from waste sites (Carpenter et al. 1998; Pitois et al. 2001). It has been observed that eutrophication is becoming widespread in coastal seas and estuaries around the world (Carpenter et al. 1998, Pinckney et al. 2001; Arhonditsis et al. 2003). Natural sources of P and N include the drainage of watershed areas, the direct atmospheric deposition onto water surfaces, and the internal recycling from lake sediments (Pitois et al. 2001). The distinction between natural sources and anthropogenic sources is not usually obvious (Pitois et al. 2001). However, the loadings from natural sources are often considered to be very low compared to those of anthropogenic origin (Ahl 1998 as cited in Pitois et al. 2001; Carpenter et al. 1998; Pinckney et al. 2001; Pitois et al. 2001; Sharply et al. 2001).

From an LCA perspective, eutrophication has been recommended as one of the impact categories for European practice (Fennveden and Potting 1999; Huijbregts and Seppala 2000; 2001; Huijbregts et al. 2001a). The impacts of eutrophication can involve scales as large as continents (Finnveden and Potting 1999; Huijbregts and Seppala 2000; 2001; Huijbregts et al. 2001a; Norris 2002). In EI95 (Goedkoop 1998), the target value for eutrophication impact was determined by considering the critical concentrations of phosphates and nitrates in surface water to be 0.15 and 2.2 mg/l, respectively. It was assumed that eutrophication would not occur at these critical concentrations. The reduction factor of 5 from the 1990 nitrogen nutrient emission was selected based on the fact that critical concentrations of both phosphates and nitrates in some important rivers in Europe were exceeded more than five times (Goedkoop 1998). As a result, the

European target value of 3.8×10^9 NP kg/yr (NP = Nutrification potential, equivalent to PO_4^{3-}) was used in EI95. Dickinson et al. (2001) estimated a value of the U.S. carrying capacity for eutrophication impact by multiplying the normalized EI95's target value by the U.S. land area translating to a calculated carrying capacity value of 8.0×10^9 NP kg/yr. Hauschild and Wenzel (1998) determined the environmental carrying capacity for Denmark using a method similar to the one in EI95 with reduction factors ranging from 5 to 20 for various types of nutrients. The Danish carrying capacities for total P and N were therefore estimated at 5.7×10^5 and 3.0×107^7 kg /yr, respectively.

For a local watershed area, the locally specific carrying capacity may be estimated using a model assessment. The carrying capacity for a regional scale is more generic, but it is difficult to assess because no such model covers all the variety of water resources. This is because the rate of eutrophication depends on a complex relation between several factors including water chemistry and depth, water inflow, mineral content or buffering capacity, and the biota of the water.

This chapter elaborates the selection of the threshold level and carrying capacity estimate using total maximum daily load, TMDL. A method that takes into account site factors in the characterization of eutrophication potential is also presented.

8.2 Threshold Level for Carrying Capacity Estimate

For some countries in Europe, the critical load mapping has been extended to nitrogen loads for eutrophication (see also Subsection 7.2.1). Typically, the critical load of N to protect forest soils (terrestrial eutrophication) is much greater than that for the protection of surface water (aquatic eutrophication) (NPS 2002). In Ireland, an estimate using an empirical approach showed that the critical loads for eutrophication were in the range of

8 to 25 kg N/ha/yr with an average of 10 kg N/ha/yr covering most of the country (Aherne and Farrell 2000). A similar range (2.8 to 28.0 kg N/ha/yr) was mapped for the UK (Hall et al. 1998). Agren (2001) suggested the critical load for Europe should be in the range of 2 to 5 kg N/ha/yr for natural forests. UNECE (1996) also suggested the critical loads for N affecting nutrient imbalance to semi-natural terrestrial and wetland ecosystems were in the range of 10-30 kg N/ha/yr. For 95% prevention of eutrophication, critical loads of 3-10 kg N/ha/yr have been established for Europe (Posch et al. 1999). In the U.S. where the critical load concept is not as widely studied as in Europe, Pardo and Driscoll (1996) suggested a range of 0 to 1,450 mol/ha/yr (20.3 kg N/ha/yr) for the critical loads for N with respect to eutrophication for study areas of forests in New Hampshire and New York.

Typically, the quality of waters in the U.S. is protected through development of individual states' pollution control strategies. These strategies are constructed to achieve the Water Quality Standards (WQS) established for rivers, lakes, estuaries and coastal waters (EPA 2003e). National drinking water standards for nitrate and nitrite, which have direct impacts to human health by causing blue-baby syndrome among other things, are established at 10 mg-N/l and 1 mg-N/l, respectively (EPA 2002a). The ambient water quality criterion of 10 mg-N/l is also adopted from the national drinking water standards (EPA 2003f). However, a drinking water standard has not been established for P because it is not considered to be directly toxic to animals and humans (at least in the phosphate form) (Carpenter et al. 1998; EPA 2002a).

For eutrophication impact, EPA has initiated development of chemical-specific (P and N) numeric water quality criteria designed to be the basis of control of excessive nutrients in surface waters (EPA 2002a; 2003e). Water quality criteria for P and N have been developed for 14 different ecoregions (EPA 1998a). Other criteria for excessive nutrient presence include chlorophyll-*a*, turbidity, and secchi depth. These criteria have been developed using a regional and waterbody-specific approach (EPA 1998a). Table 8.1 is a summary of the recent EPA recommended ecoregional nutrient criteria for lakes & reservoirs, and rivers & streams (Figure 8.1 is the ecoregion map). These criteria were derived from the conditions of the water resources that do not encounter eutrophication impairment. These criteria are used to establish state water quality standards for nutrient enrichment. The waters where the nutrient standards are exceeded are defined as impaired waters according to the Clean Water Act 303(d).

Table 8.1 Summary of the Ecoregional Nutrient Criteria
(Source: EPA 2003g)

Aggregate Ecoregion		Lakes & Reservoirs			
		Total P (μg/l)	Total N (mg/l)	Chlorophyll-a (mg/l)	Secchi depth (m)
II	Western Forested Mountains	8.75	0.10	1.90	4.50
III	Xeric West	17.00	0.40	3.40	2.70
IV	Great Plains Grass and Shrublands	20.00	0.44	2.00$^{(1)}$	2.00
V	South Central Cultivated Great Plains	33.00	0.56	2.30$^{(1)}$	1.30
VI	Corn Belt And Northern Great Plains	37.50	0.78	8.59$^{(1)}$	1.36
VII	Mostly Glaciated Dairy Region	14.75	0.66	2.63	3.33
VIII	Nutrient Poor Largely Glaciated Upper Midwest and Northeast	8.00	0.24	2.43	4.93
IX	Southeastern Temperate Forested Plains and Hills	20.00	0.36	4.93	1.53
XI	The Central and Eastern Forested Uplands	8.00	0.46	2.79$^{(1)}$	2.86
XII	Southeastern Coastal Plain	10.00	0.52	2.60	2.10
XIII	Southern Florida Coastal Plain	17.50	1.27	12.35$^{(2)}$	0.79
XIV	Eastern Coastal Plain	8.00	0.32	2.90	4.50
Aggregate Ecoregion		Rivers & Streams			
		Total P (μg/l)	Total N (mg/l)	Chlorophyll-a (mg/l)	Turbidity (FTU/NTU)
I	Willamette and Central Valleys	47.00	0.31	1.80	4.25
II	Western Forested Mountains	10.00	0.12	1.08	1.30$^{(3)}$
III	Xeric West	21.88	0.38	1.78	2.34
IV	Great Plains Grass and Shrublands	23.00	0.56	2.40	4.21
V	South Central Cultivated Great Plains	67.00	0.88	3.00	7.83
VI	Corn Belt And Northern Great Plains	76.25	2.18	2.70	6.36
VII	Mostly Glaciated Dairy Region	33.00	0.54	1.50	1.70$^{(3)}$
VIII	Nutrient Poor Largely Glaciated Upper Midwest and Northeast	10.00	0.38	0.63	1.30
IX	Southeastern Temperate Forested Plains and Hills	36.56	0.69	0.93$^{(1)}$	5.70
X	Texas-Louisiana Coastal and Mississippi and Alluvial Plains	128.00	0.76	2.10$^{(1)}$	17.50
XI	The Central and Eastern Forested Uplands	10.00	0.31	1.61$^{(1)}$	2.30$^{(3)}$
XII	Southeastern Coastal Plain	40.00	0.90	0.40$^{(1)}$	1.90$^{(3)}$
XIV	Eastern Coastal Plain	31.25	0.71	3.75$^{(1)}$	3.04

Note: (1) Spectrophotometric method, (2) trichromatic method, (3) NTU

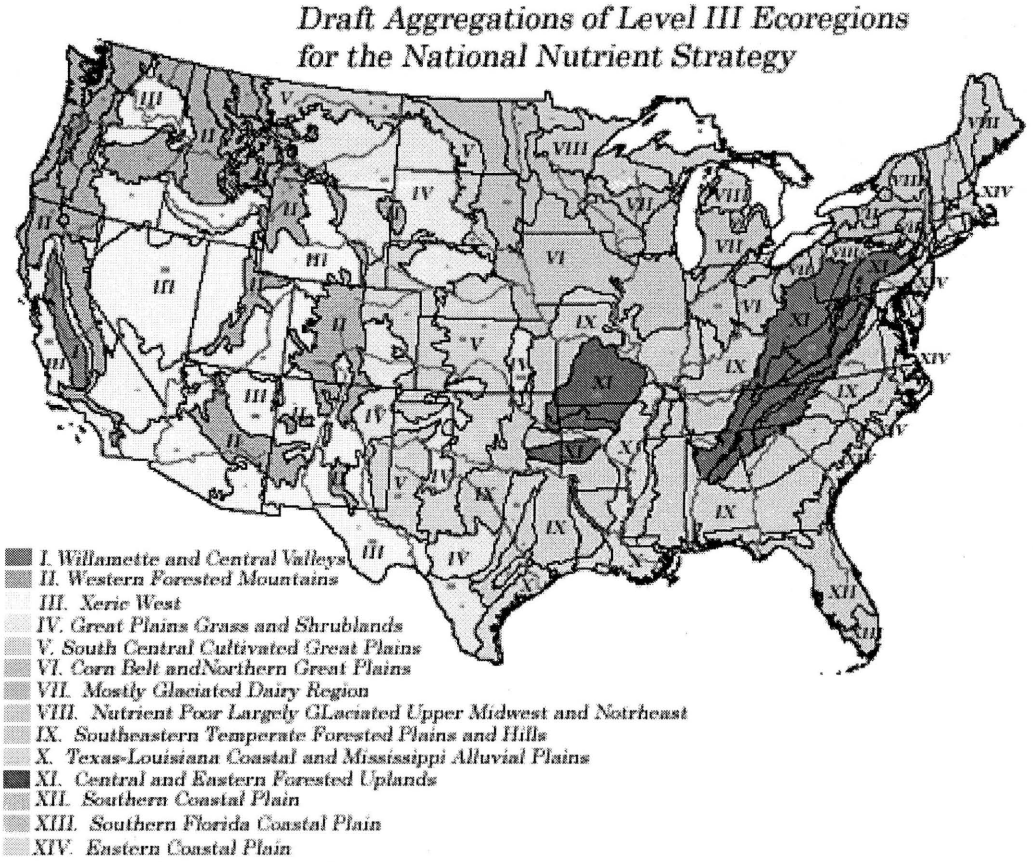

Figure 8.1 U.S. ecoregion map.
(Source: EPA 2003g)

It is assumed for this research that eutrophication is the effect of human-induced nutrient (P and N) inputs and that eutrophication will not occur if the concentrations of total P and N are limited to the recommended level. Therefore, the concentrations of P and N according to the recommendation criteria are selected as the desirable or threshold levels for estimating the carrying capacity for eutrophication impact. The carrying capacity can be adopted from the TMDLs developed as an attempt to maintain the water quality criteria, or other available allowable emissions of P and N, where TMDLs have been established.

8.3 Carrying Capacity Estimate

8.3.1 Carrying Capacity Estimate Using TMDL

Under requirements of Section 303(d) of the Clean Water Act, the states are directed to identify and list impaired waters and implement their total maximum daily load (TMDL) programs (NCR 2001). The objective of a TMDL program is to improve water quality through control of both point and nonpoint sources of pollution. Information in a TMDL specifies a pollutant budget that must be achieved to maintain water quality standards and to allocate pollutant loads. States must submit TMDLs for approval by EPA. Although TMDL development has been progressing since 1972, not many TMDLs have been assessed until recently. Recent legal actions have forced EPA and states to develop more TMDLs (EPA 2006). It is estimated that over 40,000 TMDLs for 25,000 waters must be developed over the next 10 to15 years to meet current guidelines (NRC 2001; Borsuk et al. 2002). Currently, the TMDL program is applied to several pollutants such as sediment, pathogens, nutrients, metals, temperature, pH, and pesticides.

There is no specific model or method to assess a TMDL. A state has to develop TMDLs using methodologies that fit its impaired waters. Typically, a model is used to determine the total daily load to maintain a state's water quality standard for a pollutant under consideration. Bringing forward approximate values for point and nonpoint sources of the pollutant, and a margin of safety, the TMDL can then be calculated (EPA 2003h). A margin of safety estimate is required to ensure that an impaired water will meet its designated uses. The margin of safety is based on uncertainties in the model and analytical assumptions, observed data, and natural variability (Lee and Jones-Lee 2002). The TMDL can be generically described by the following equation (EPA 1999a):

$$TMDL = LC = \Sigma WLA + \Sigma LA + MOS \tag{8.1}$$

LC is the loading capacity or the greatest loading a water can receive without violating water quality standards. WLA is the waste load allocation or the portion of the TMDL allocated to existing or future point sources. LA is the load allocation or the portion of the TMDL allocated to existing or future nonpoint sources and natural background. MOS is the margin of safety to compensate for uncertainty about the relationship between pollutant loads and receiving water. TMDL can be expressed in terms of mass per time, toxicity or other appropriate measures.

As mentioned, the ecoregional water quality criteria for nutrients have been mapped. These nutrient criteria are used to establish a state's water quality standards. Impaired waters due to nutrient enrichment can then be identified. Recently, over 5,000 waters are listed as impaired due to nutrient enrichment but only about 1,300 TMDLs have been approved (EPA 2003h). Phosphorus and nitrogen concentrations are the major indicators used in these TMDL assessments. Figure 8.2 shows the correlation plot between TMDL for nutrients (P and N) and watershed area of the impaired waters. The data were obtained from approximately 200 TMDL reports and assessments across the U.S. made available online (Appendix D). Although the rate of eutrophication and nutrient absorption capacity depends on complex characteristics among watersheds; the trends for both P and N in Figure 8.2 indicate good correlations between TMDL and watershed area. In other words, TMDL varies due to the size of the watershed.

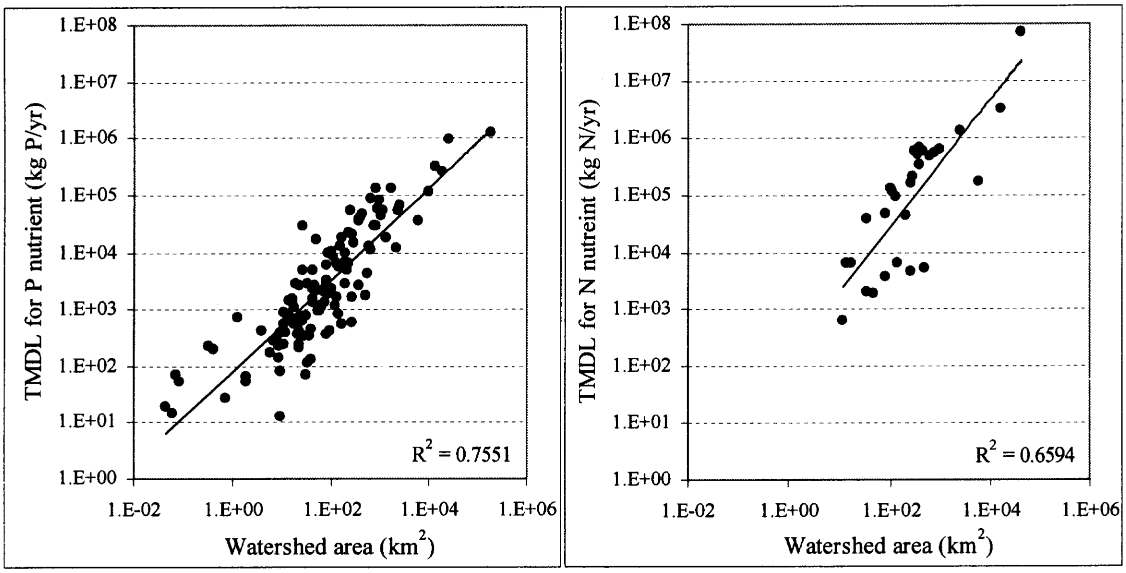

Figure 8.2 Correlations between TMDL and watershed area for phosphorus nutrient (left panel) and nitrogen nutrient (right panel).

The economic carrying capacity for eutrophication impact can be estimated using TMDL based on the assumptions that: 1) TMDL for nutrients of a watershed depends on its size (this correlation is illustrated in Figure 8.2); 2) TMDL data from the available 200 reports and assessments represent the eutrophication phenomena (in terms of dose-response relationship) surface waters across the U.S.; 3) the nutrient loadings from natural sources are very limited, and can be negligible, compared to those from human activities; and 4) the TMDL amount is totally due to human activities. The latter two assumptions may cause the carrying capacity to be overestimated if the nutrient loadings from natural sources be significant. From these assumptions, a simple means to estimate a TMDL or carrying capacity of a watershed is to use the average relationship between TMDL and watershed area, expressed in terms of a critical load (e.g., kg P/km^2/yr). This critical load is analogous to the term used internationally to determine allowable loadings of pollutants for acidification and eutrophication impacts. The U.S. carrying capacity for both P and N can then be calculated from the critical load using equation:

$$U.S.\ CC\ (kg/yr) = Critical\ Load\ (kg/km^2/yr) \times U.S.\ Area\ (km^2) \qquad (8.2)$$

Basically, an average critical load (arithmetic mean) can be used to calculate the carrying capacity. However, in this research the median critical load derived from probability analysis is used instead to minimize the bias due to outlier data. Furthermore, the probability function associated with the carrying capacity estimate in this research can also be determined. This probability function may be useful for further uncertainty assessment. Figure 8.3 presents the log normal probability plots for P and N critical loads and probability function (Z-score). The critical loads are determined for different probabilities: 50th percentile, 50% confidence interval (25th – 75th percentiles), and 95% confidence interval (2.5th -97.5th percentiles). The carrying capacities for P and N can then be calculated using Equation (8.2) as presented in Table 8.2. At the 50th percentile, the U.S. carrying capacities for P and N are 3.24×10^8 kg P/yr and 2.95×10^9 kg N/yr, respectively. These carrying capacities are calculated using the U.S. land area of 9.4×10^6 km^2. And these carrying capacities are much higher than the U.S. normalization value (5.02×10^6 kg N/year) derived from 1999 emission data for used with TRACI (Bare et al. 2006). This means that the current U.S. emissions do not exceed the environmental carrying capacity. Since eutrophication is considered to affect areas as large as regional and continental scales, only the national level carrying capacities are determined.

A locally specific carrying capacity (watershed level) may be estimated using the local critical load. However, it is not appropriate to evaluate state-specific carrying capacities because of insufficient data, i.e., TMDL, which are still available only for limited areas. State-specific carrying capacities derived from insufficient TMDL data may result in atypical values of carrying capacity.

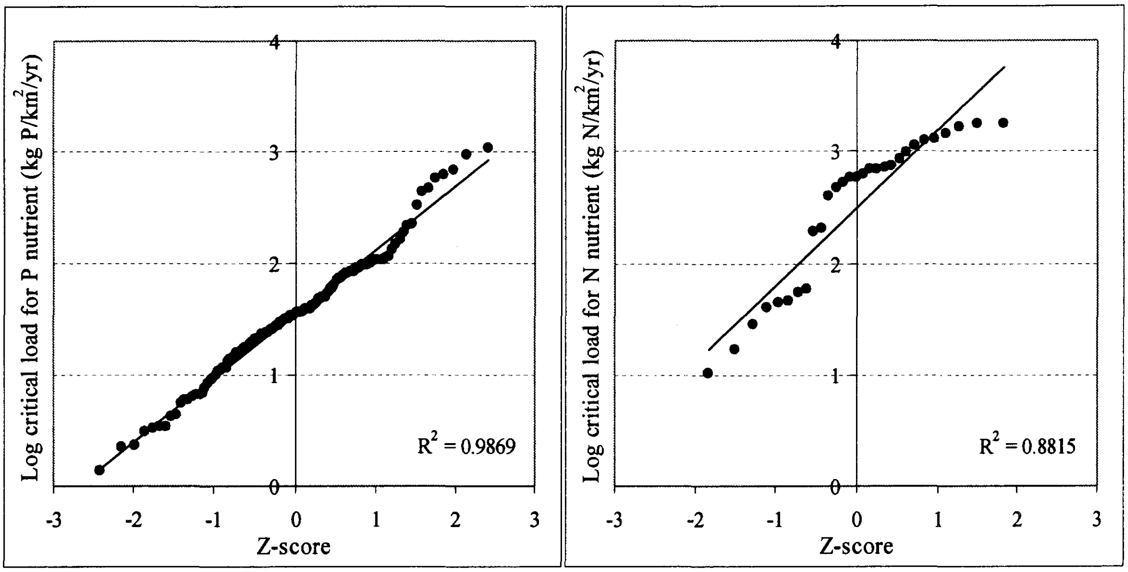

Figure 8.3 Log normal probability plots for critical load and probability function (Z-score) for phosphorus (left panel) and nitrogen (right panel).

Table 8.2 Carrying Capacity Estimates at Different Probabilities.

Probability percentile	Phosphorus		Nitrogen	
	Critical load (kg P/km^2/yr)	Carrying capacity (kg P/yr)	Critical load (kg N/km^2/yr)	Carrying capacity (kg N/yr)
2.5	2.55	2.40E+07	13.89	1.31E+08
25	14.09	1.32E+08	107.49	1.01E+09
50	**34.49**	**3.24E+08**	**314.12**	**2.95E+09**
75	84.41	7.93E+08	917.98	8.63E+09
97.5	465.62	4.38E+09	7103.30	6.68E+10

From the LCA perspective, eutrophication impact is considered a homogenous impact category, in which several burdens from nutrient emissions can be aggregated due to the assumption that P and N compounds have an additive effect to the eutrophication impact (Assies 1998). For LCIA, the Redfield ratio and the limiting nutrient concept are accepted as a method to derive the characterization factors (also known as nutrification potentials, nutrient enrichment potentials, and nutrient factors) to express the contribution of P and N emissions to biomass production in terms of the equivalent emission of a reference substance (Goedkoop 1998; Hauschild and Wenzel 1998; Huijbregts and

Seppala 2001; Goedkoop and Spriensma 2001; Norris 2002). This method implies that the characterization factors of the substance that is the limiting nutrient (which is P in this case) are greater than those of the other (which is N) by a number that is equal to N:P in the Redfield ratio (Norris 2002). For example, P has the potential to cause eutrophication as high as 16 times of that of N due to the Redfield ratio of C:N:P = 106:16:1.

In this research, however, the cross-species aggregation between P and N is not recommended. This is due to the spatial and temporal variations of the N:P ratio in various water bodies (Pinckney et al. 2001; Norris 2002; Turner et al. 2003). Furthermore, the additive effect due to the P and N loadings seems to be less logical, especially for the STM, where the impact assessment is based on a damage-oriented approach rather than an emission-oriented approach used in other LCIA methods. As a result, the carrying capacity estimates for P and N are not aggregated and they are counted as impact subcategories.

The amount of an emission must be expressed in the form of its contribution to the effect, which in this case is in the form of P or N compound. Hauschild and Wenzel (1998) suggested a simple approach for estimating eutrophication potentials using the number of moles of P or N that can be released into the environment from one mole of the P or N compounds in question. Hence, a eutrophication potential (EP) can be calculated for P and N compounds as (Hauschild and Wenzel 1998):

$$EP(P) = \frac{p \times 30.97}{MW_i} \tag{8.3}$$

$$EP(N) = \frac{n \times 14.01}{MW_i} \tag{8.4}$$

Where p and n represent the number of moles of P and N compounds respectively while 30.97 and 14.01 are the atomic masses of P and N respectively. MW_i is the molecular weight of the substance in question, i. Norris (2002) and Karrman and Jonsson (2001) also suggested that eutrophication potentials for BOD and COD, which are often present in LCI, should be made available because they contain the essential elements (and may be nutrients) that contribute to eutrophication impact. Table C.4(a) in Appendix C summarizes eutrophication potentials for P and N compounds as well as BOD and COD. These characterization factors are widely used in LCIA (Goedkoop 1998; Goedkoop and Spriensma 2001; Huijbregts and Seppala 2001; Norris 2002). Eutrophication potentials have units of kg P/kg $Nutrient_i$ or kg N/kg $Nutrient_i$. Simple equations are used to characterize the amount in kg of nutrient i relative to kg of P-eq or N-eq:

$$kgP\text{-}eq = EP(P)_i \times kg\,Nutrient_i \tag{8.5}$$

$$kgN\text{-}eq = EP(N)_i \times kg\,Nutrient_i \tag{8.6}$$

8.3.2 Spatial Distributions of Emissions

The transport of nutrients is complex and difficult to assess (Finnveden and Potting 1999; Norris 2002). Nutrients can be emitted to both air and water. Nutrients in water can be transported half-way across a continent while nutrients in air can be transported across a region (Norris 2002). Impacts can occur in many different types of terrestrial and aquatic systems over scales as large as continents (Finnveden and Potting 1999; Huijbregts and Seppala 2000; 2001; Huijbregts et al. 2001a; Norris 2002). The fate and transport processes depend on site-specific characteristics of the emitting source and environmental pathways. The impacts depend on background loads and concentrations and sensitivities of different ecosystems. It is assumed for this research that 100% of nutrient releases to water have reached aquatic environments. However, not all of air emissions reach water resources. A factor that takes into account the probability of an air emission reaching aquatic environments is required for site-dependent characterization.

Huijbregts and Seppala (2000) have developed an approach for LCIA application to characterize nutrients emitted in Europe by using a long-range transport model integrated with the sensitivity of the receiving environment (e.g., the critical loads of ecosystems). For the U.S., a similar approach has been adopted for the Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts-TRACI (Bare et al. 2002; Norris 2002; EPA 2003a). However, the sensitivities of ecosystems are not considered in this approach (Norris 2002). The site-dependent transport factors in TRACI take into account the transport of nutrients by means of air and water pathways. This tool assumes that 100% of nutrient releases to water have reached aquatic environments. The transport factors for air emission express the probability that a nutrient arrives in an aquatic environment initially by an air pathway (Bare et al. 2002). A higher transport factor indicates more deposition of the nutrient from air transport over water resources. The transport factors were developed based on water budget analysis and nutrient fate and transport modeling (Norris 2002). By using the transport factors for air emission, the overall site-dependent characterization for an emission of P nutrient and N nutrient can be modified from Equation (8.5) and (8.6) and is expressed as:

$$kgP\text{-}eq = EP(P)_i \times (SF_{Eutro,air} \times kg\ Nutrient_{i,air} + kg\ Nutrient_{i,water}) \qquad (8.8)$$

$$kgN\text{-}eq = EP(N)_i \times (SF_{Eutro,air} \times kg\ Nutrient_{i,air} + kg\ Nutrient_{i,water}) \qquad (8.9)$$

Where $SF_{Eutro,air}$ is a site-specific transport factor for an emission of nutrient from a U.S. site by means of air emission. Table C.4(b) in Appendix C presents the transport factors for air emission at the state and regional levels. The average transport factor for air emission for U.S. is approximately 0.07.

8.4 Summary

The carrying capacity for eutrophication impact is estimated using the critical load concept. The critical loads for eutrophication are adopted from TMDL reports from 27 states available online. Generally, TMDLs are reported for impaired waters only. The carrying capacity estimate in this research is derived from the available TMDL reports assuming that the impaired waters can be representative of all other waters. And the U.S. carrying capacity for phosphorus nutrient is 3.24×10^8 kg P/yr or 9.9×10^8 kg PO_4^{3-}/yr[1] while the carrying capacity for nitrogen nutrient is 2.95×10^9 kg N/yr.

From the LCA perspective, several burdens from nutrient emissions may be aggregated due to the assumption that P and N compounds have an additive effect to the eutrophication impact. In this research, however, cross-species aggregation between P and N is not recommended. This is due to the spatial and temporal variations of the N:P ratio in various water bodies (Pinckney et al. 2001; Norris 2002; Turner et al. 2003). Furthermore, the synergistic and additive effects due to the P and N loadings seem make aggregation less logical for the STM, where the impact assessment is based on a damage-oriented approach rather than an emission-oriented approach used in other LCIA methods. As a result, the carrying capacity estimates for P and N are not aggregated and they are counted as impact subcategories.

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[1] 1 kg P/yr = 95/31 kg PO_4^{3-} /yr

For comparison purposes, the carrying capacity values for both phosphorus and nitrogen may be aggregated using nutrient factor (Huijbregts and Seppala 2001) to become a single carrying capacity of 2.23×10^{9} kg PO_4^{3-}/yr[1]. This aggregated value can be thought of as the national allowable P and N emissions to prevent eutrophication. This value is about four times less than the previous carrying capacity value estimated by Dickinson et al. (2001) for STM application (8.0×10^{9} kg PO_4^{3-}/yr). The average critical load for nitrogen estimated from TMDL is 314.1 kg N/km^2/yr or 3.14 kg N/ha/yr. This value is around the low end of critical loads for eutrophication reported in Europe. It is also about six times less than the value reported for study areas of forests in New Hampshire and New York.

The carrying capacity for eutrophication impact can be refined when better methodologies and new scientific findings become available. Further improvement could come from the evaluation of locally-specific carrying capacities. A locally-specific carrying capacity can be determined from the source-receptor or dose-response relationship that is the result of water pollution modeling. Another means to refine the carrying capacity is to update the TMDL information. Although TMDL development has been progressing since 1972, not many TMDLs have been assessed until the past few years. Recently, legal actions have forced EPA and states to assess more TMDLs. A large number of TMDLs is thus expected to be developed in the next 10 to15 years.

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[1] Nutrient factor for Total N = 0.42

CHAPTER 9

PHOTOCHEMICAL OZONE FORMATION IMPACT

9.1 Overview

Ozone (O_3) is a photochemical oxidant and the major component of photochemical ozone formation (also known as photochemical smog, Los Angeles smog, and summer smog) (Lu and Turco 1996). Summer smog contrasts with winter smog, or London smog, which is caused by a mixture of high levels of particles, carbon monoxide, nitrogen oxides and sulfur dioxide (Bower et al 1994). Ozone in photochemical smog is a major environmental concern because of its adverse impacts on human and ecological health. Ozone causes human and animal health problems by impeding lung function and damaging the respiratory system and causing problems to plants by damaging their leaves (Sillman 1999; Madden and Hogsett 2001; Laurence and Andersen 2003; Manning 2003). High levels of tropospheric or ambient ozone are responsible for most violations of the National Ambient Air Quality Standards (NAAQS) in the U.S. (EPA 2000a; Cynthia-Lin et al. 2001). Tropospheric ozone is also a major urban air pollutant in other regions such as Europe and Asia (Monks 2000; Huang et al.2001; Derwent et al. 2003; Fuhrer and Booker 2003).

Tropospheric ozone is produced by chemical reactions of emitted primary pollutants. Volatile organic compounds (VOCs) and nitrogen oxides (NO_x) are the two primary precursors for ozone formation, which also requires the presence of sunlight. The chemical reaction mechanisms of ozone formation are complex and are considered to lack certainty (Dodge 2000; Sadanaga et al. 2003). However, basic photochemical reactions involving the formation of tropospheric ozone are (Atkinson 2000; Dodge 2000; Guicherit and Roemer 2000; Jenkin and Clemitshaw 2000; Sadanaga et al.2003):

Photolysis of O_3

$$O_3 + hv \rightarrow O(^1D) + O_2 \quad (\lambda < 330\ nm) \tag{9.1}$$

$$O(^1D) + H_2O \rightarrow 2OH \tag{9.2}$$

Photolysis of NO_2

$$NO_2 + hv \rightarrow NO + O(^3P) \quad (\lambda \leq 420\ nm) \tag{9.3}$$

$$O(^3P) + O_2 + M \rightarrow O_3 + M \quad (M = air) \tag{9.4}$$

$$O_3 + NO \rightarrow NO_2 + O_2 \tag{9.5}$$

Degradation of non-methane VOC

$$RH + OH \rightarrow R + H_2O \tag{9.6}$$

$$R + O_2 + (M) \rightarrow RO_2 + (M) \tag{9.7}$$

$$RO_2 + NO \rightarrow RO + NO_2 \tag{9.8}$$

$$RO + O_2 \rightarrow HO_2 + RCHO \tag{9.9}$$

$$HO_2 + NO \rightarrow OH + NO_2 \tag{9.10}$$

$$2NO_2 + hv \rightarrow 2NO + 2O(^3P) \tag{9.3}$$

$$O(^3P) + O_2 + (M) \rightarrow O_3 + (M) \tag{9.4}$$

$$RH + 4O_2 + hv \rightarrow RCHO + H_2O + 2O_3 \tag{9.11}$$

Degradation of methane

$$OH + CH_4\ (+O_2) \rightarrow CH_3O_2 + H_2O \tag{9.12}$$

In the presence of CO

$$CO + OH\ (+O_2) \rightarrow CO_2 + HO_2 \tag{9.13}$$

Large quantities of VOCs are emitted into the troposphere from anthropogenic and biogenic or natural sources (Atkinson 2000; Sawyer et al. 2000). The estimated global emissions of methane, which is less active in photochemical ozone formation than most VOCs, are approximately 155-240 million tones/year from natural sources such as wetlands and 350-375 million tones/year from anthropogenic sources such as ruminants, rice paddies, landfills, and the combustion of fossil fuels (Atkinson 2000). Non-methane volatile organic compounds (NMVOC) are also emitted into the troposphere from a variety of anthropogenic sources, including combustion of fossil fuels, fuel storage and transport, solvent usage, emissions from industrial operations, landfills, and waste facilities (Sawyer et al., 2000). The estimated U.S. and global emissions of NMVOC are approximately 20 million tonnes of carbon/year and 60-140 million tonnes of carbon/year, respectively, from anthropogenic sources and 30-45 million tonnes of carbon/year and 1,150 million tonnes of carbon/year, respectively, from natural sources (Atkinson 2000; Guenther et al. 2000).

Natural sources of tropospheric NO_x are the emissions from soil and the formation in situ from lightning (Atkinson 2000). Major anthropogenic sources of tropospheric NO_x are the emissions from fossil fuel combustion (Atkinson 2000; Sawyer et al. 2000). The estimated U.S. and global emissions of NO_x are approximately 1 million tonnes/year and 10 million tonnes/year (as N), respectively, from natural sources, and 6 million tonnes/year and 40 million tonnes/year (as N), respectively, from anthropogenic sources (Atkinson 2000; Guenther et al. 2000). In urban areas, NMVOC and NO_x from anthropogenic sources dominate over NMVOC and NO_x from natural sources and vice versa in rural areas (Atkinson 2000).

From the LCIA perspective, methodologies for assessing the photochemical ozone formation impact have neglected NO_x entirely. There are a few LCIA studies focusing on the value that is similar to the carrying capacity estimate for VOC emission. The VOC emission target value for EI95 was calculated according to the impact on crops (Goedkoop 1998). The acceptable level for ozone concentration was 0.03 ppm under which crop damage would not occur. It was estimated that the level of the summer smog ozone should be reduced by 90% from the existing level of 0.3 ppm to the target level of 0.03 ppm. With this ozone reduction rate, VOCs and NO_x, which are the primary photochemical smog precursors, must be reduced by 60-70%. Hence, the reduction factor of 2.5 from the current emission of VOCs was applied and used in EI95 (Goedkoop 1998). As a result, the European target value of 3.56×10^9 POCP kg/yr[1] was used. Dickinson et al. (2001) estimated a subsequent value of U.S. carrying capacity for photochemical ozone formation impact by multiplying the normalized EI95's target value by the U.S. land area resulting in a carrying capacity value of 7.52×10^9 POCP kg/yr.

Two different possible methods to determine carrying capacity for photochemical ozone formation were mentioned by Hauschild and Wenzel (1998), namely the use of a threshold value and pre-industrial emissions. For the threshold value approach, the types of assumptions required for VOCs and NO_x emissions are: 1) a threshold value of ozone in ambient air, 2) the total volume of air in which ozone is diluted, 3) the average lifetime of ozone, 4) the average annual transport of ozone down from the stratosphere, 5) a fixed background concentration of NO_x, 6) the ozone formation efficiency of ethylene, 7) natural emissions of VOCs, and 8) import and export of air pollution in the area. Hauschild and Wenzel (1998) also pointed out that the carrying capacity could be

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[1] POCP = Photochemical ozone creation potential (equivalent to C_2H_4)

considered for both VOCs and NO_x. For the estimate on a pre-industrial basis, the carrying capacity could be defined from the natural state, where the carrying capacity would be a pre-industrial emission scenario.

This chapter presents an approach for carrying capacity estimation. This approach uses OZIPR (Gery and Crouse 1990; Shodor 1997), a photochemical model, to calculate the critical emissions of the major ozone formation precursors (VOCs and NO_x). The carrying capacity estimate for photochemical ozone formation impact using the OZIPR results coupled with the threshold levels is also elaborated. Methods for the characterization of VOC's are discussed and the factor for characterizing a non-specific VOC to the reference species (C_2H_4) is estimated. Lastly, a method to include the site factor in the characterization of photochemical ozone formation potential for VOC emissions is also discussed.

9.2 Approach

The carrying capacity for photochemical ozone formation can be estimated using the threshold-oriented technique discussed in Section 4.2, Chapter 4. The development of the methodology to evaluate the carrying capacity in this chapter is an original contribution of this research.

A general step for evaluating the carrying capacity for photochemical ozone formation impact using the threshold-oriented technique is as follows.

1. Selection of a desirable threshold level.
2. Selection of an appropriate photochemical model and gathering data.
3. Simulation of emission-concentration relationships using the photochemical model. The emission is of the precursor pollutants to be evaluated for the carrying capacity,

i.e., VOC and NO_X. The concentration is of the pollutant selected as the indicator for photochemical ozone formation impact, e.g., ozone.

4. Determination of the critical emission rates for the precursor pollutants that result in the concentration of the indicator not exceeding the desirable threshold level. The emission rates may be for the entire national scale or per unit area that can be scaled up to the national scale.
5. Determination of the national scale carrying capacity for the precursor pollutants from the critical emission rates.

The algorithm for the carrying capacity estimation using photochemical modeling (OZIPR) in the threshold-oriented method is presented in Figure 9.1.

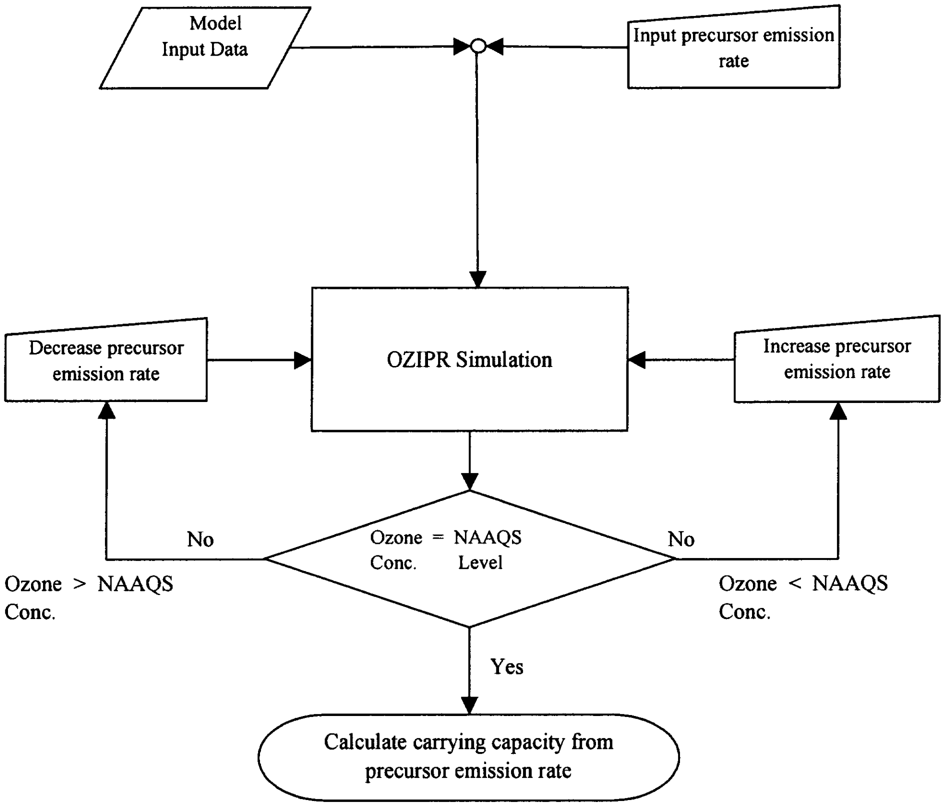

Figure 9.1 Algorithm for carrying capacity estimation using threshold-oriented technique for photochemical ozone formation impact.

9.3 Threshold Levels for Carrying Capacity Estimate

An issue for measuring the impacts associated with photochemical ozone formation is the selection of an endpoint indicator. A common measure allows consistent discussion about the relationship between photochemical ozone formation and its impacts. The name of the impact itself is instructive of the indicator that should be used. Ozone, a photochemical oxidant and the most prominent constituent of photochemical smog, is used as the endpoint indicator for threshold level selection. Ground level ozone is regulated through the National Ambient Air Quality Standard (NAAQS).

The original NAAQS for ozone, set in 1979, was a 1-hr average of 0.12 ppm not to be exceeded more than three times in three consecutive years. Based on review of scientific evidence linking ozone exposure and adverse effects on human health and welfare, in 1997 EPA revised the standard to an 8-hr average of 0.08 ppm based on the three-year average of the annual fourth-highest daily maximum 8-hr average ozone concentrations measured at each monitor within an area (Federal Register 1997). As of 1998, 51 million persons lived in areas violating the old standard while approximately 130 millions persons lived in areas violating the new standard (EPA 2000a).

Following a lawsuit by environmental groups, EPA and environmental groups agreed to a schedule for EPA to promulgate air quality designations for the 8-hr ozone standard by April 2004. In this research, the carrying capacity is evaluated based on both old and new ozone standards.

9.4 Photochemical Ozone Modeling

9.4.1 Photochemical Model

An important step in the threshold-oriented technique is the selection of an appropriate photochemical model. Similar to other pollutant transport phenomena in environmental media, photochemical ozone formation is a complex process; it consists of a large number of parameters which vary in properties and characteristics both spatially and temporally. However, the estimates for carrying capacity should rely on simple but applicable models yet should be transparent to users, especially LCA practitioners who come from a variety of disciplines, so that they can justify the methodology or even make further modifications should they be required. The factors that must be considered in the selection of an environmental model are: the level of detail and accuracy needed for the analysis, the technical validity of the simulation modeling, the resources available, and the detail and accuracy of the data used in the modeling. Furthermore, the model should be a generic one that can be applied on a larger scale, i.e., the national scale. Appropriate data should also be available before a model is chosen. A model that requires detailed, precise, input data should not be used if these data are not available.

There are a variety of mathematical models used to describe the relationship between the precursor pollutants and the formation of photochemical ozone. Simple box or zero-dimension models (Schere and Demerijian 1978 as cited in Jin and Demerjian 1993; Gery et al. 1989) have been used to predict pollutant concentrations in an area where pollutants are emitted and undergo chemical reactions. Transport into and out of the box by meteorological processes and dilution is taken into account. Three-dimensional Eulerian grid-based models, which account for emissions, chemistry, and dispersion simultaneously, are used to assess more realistic pollutant transport and concentrations. The Eulerian grid-based models can be separated into the urban-scale models, where the horizontal size of each grid is of the order of a few kilometers (UAM-Urban Airshed Model: Reynolds et al. 1973; SAI 1999, CIT-California Institute of Technology model: McRae et al. 1982; Russell et al. 1988, SMOG-Surface Meteorology and Ozone Generation model: Lu et al. 1997a; 1997b) and the regional-scale models, where the scale is of the order of 15-130 kilometers (RADM-Regional Acid Deposition Model: Chang et al. 1987, CMAQ-Models-3/Community Multiscale Air Quality modeling system: EPA 1999b). The Eulerian grid-based models are considered the most accurate means of predicting concentrations of pollutants. They have been primarily used by states and federal agencies for developing emission control strategies to reduce ambient ozone concentrations to a level below the NAAQS. However, these models are burdensome to run because they require considerable expertise, high costs, and an extensive computing process. Based on the criteria discussed above and the availability of the database for modeling, the EPA's OZIPR model (Gery and Crouse 1990) is used to estimate the carrying capacity for photochemical ozone formation precursors in this research.

9.4.2 Description of Model and Database Used in Carrying Capacity Estimate

The OZIPR (Ozone Isopleth Plotting model for Research) is a modified version of the OZIP (Ozone Isopleth Plotting model: EPA 1989a), which is a city-specific model which is used to fill the gap between more sophisticated photochemical dispersion models and proportional (rollback) modeling techniques. The OZIP can be used to simulate ozone formation in urban atmospheres. OZIP is a one-dimensional photochemical box model with a time-varying box height (the mixing layer). This model calculates the maximum 1-hr average ozone concentrations given a set of input assumptions about initial precursor concentrations, light intensity, dilution, diurnal and spatial emission patterns, transported pollutant concentrations, and the reactivity of the precursor mix. The results of multiple simulations are used to produce an ozone isopleth diagram for particular cities. This isopleth diagram relates maximum ozone concentrations to concentrations of NMVOCs and NO_x. The diagram can be used in the Empirical Kinetic Modeling Approach (EKMA: EPA 1989a) to calculate the emission reductions necessary to achieve air quality standards.

The OZIPR (Gery and Crouse 1990; Shodor 1997) is an alternative version of the OZIP model with more modifications for research purposes. OZIPR employs a trajectory-based air quality simulation model, which, in conjunction with the EKMA, relates ozone concentrations to levels of VOCs and NO_x emissions. It is specifically enhanced to provide an input of more parameters, but in a less rigid format. This model was designed to predict surface ozone and other HAP concentrations. The model, which is generally run for daylight hours on a single day basis, calculates solar radiation from the zenith angle of the sun based on date, location, and time of day. Inputs include initial concentrations and hourly emission rates for the relevant chemical species and the hourly meteorological parameters for temperature, humidity, mixing heights, and pressure. The hourly emissions input data typically can be obtained from emission inventories. The model calculates ozone, total VOC, NO_x, and other secondary pollutant concentrations. An important option of the OZIPR is the varied initial concentrations for both VOC and NO_x that are required in order to plot the isopleth of the ozone concentration levels.

To estimate the carrying capacity for VOC and NO_x emissions, all input data including the city-specific VOC emissions, existing pollutant concentrations, and meteorological data are adopted from an EPA study to estimate the production of HAPs (aldehydes in this case) using OZIPR (EPA 1999c). This EPA study focused on the use of OZIPR to estimate some species of HAPs that are secondary pollutants from the photochemical process. Ten cities across the U.S. were selected as the study areas: Atlanta, Boston, Chicago, Denver, Houston, Los Angeles, Phoenix, Pittsburgh, Seattle, and Washington DC (Figure 9.2). The input data are in the daytime basis. For each study area, there are twelve sets of input data made for three typical days in various seasons: winter (December-February), spring (March-May), summer (June-August), and autumn (September-November). The methods for preparing the input data were summarized in the EPA study (EPA 1999c).

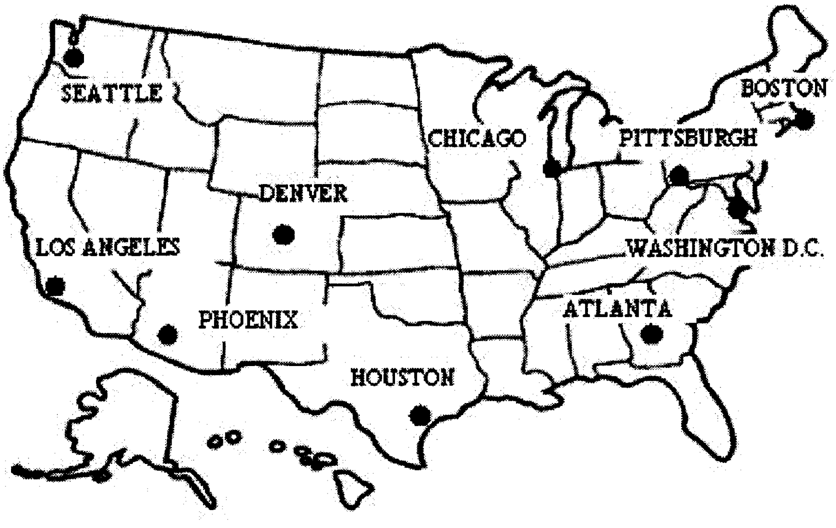

Figure 9.2 Study areas for an EPA study to estimate the production of HAPs (aldehydes) using OZIPR model.

9.5 Carrying Capacity Estimate

9.5.1 Emission-Concentration Relationship

Isopleth Plots

The simulation results are the numerical data for isopleth plots of the 1-hr average ozone concentrations, VOCs, and NO_X for 12 typical days (three typical days in a season) for those ten cities. For each data set, VOC and NO_X concentrations are varied (0 to 5 ppm for VOC and 0 to 1 ppm for NO_X) to provide the ozone concentrations in each mixture of both primary pollutants. Figure 9.3 shows an example of an isopleth plot using the data from the OZIPR simulation for a typical summer day in Chicago. The isopleth plots demonstrate that the dependence of ozone formation on the changes in precursor concentrations is complex and highly non-linear. The key results read from the isopleth plots are the critical VOC concentration and critical NO_X concentration, which are defined as the lowest VOC and NO_X concentrations (ppm), respectively, that cause ozone concentration at a level as high as the selected threshold level. For example (Figure 9.3), the critical VOC concentration reads 2.0 ppmC and the critical NO_X concentration reads 0.1 ppm when the threshold level of 1-hr average ozone concentration is 0.12 ppm.

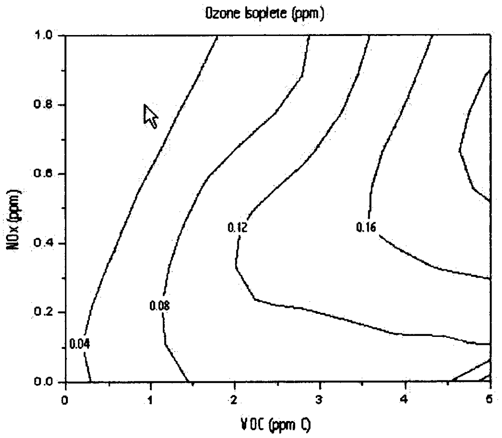

Figure 9.3 Example of ozone isopleth plot

The critical VOC and NO_x concentrations are obtained for two regimes: the VOC-sensitive regime and the NO_x-sensitive regime. A critical VOC concentration for the VOC-sensitive regime is obtained regardless of the available NO_x concentration and vice versa for a critical NO_x concentration. This assumption provides the conditions where the emission of a precursor has the greatest effect on ozone formation. By this assumption, the critical concentration of a precursor is determined based on the most conservative scenario, where ozone concentrations will not exceed the threshold level in any case. The results from the simulation show that the ratios between VOC and NO_x at the critical VOC or NO_x concentrations are in the range of 5 to 20 (ppmv/ppmv), which is in the typical range of observable atmospheric VOC/NO_x (Shodor 1997). The assumption that critical VOC or NO_x concentration can be achieved regardless of availability of the other precursor results in a conservative bias and produces a relatively low estimate of carrying capacity. Practically, NO_x emission sources are likely to be the same as VOC emission sources; this makes atmospheric NO_x concentration relatively high in the presence of high VOC concentration (discussed in Section 9.1). However, higher concentrations of VOC or NO_x do not necessarily imply higher concentrations of ozone because, as stated, the influence on ozone formation caused by the changes in precursor concentrations is complex and highly non-linear (as illustrated in Figure 9.3).

Equivalence of 1-hour and 8-hour Ozone Threshold Levels

The OZIPR provides the results only for maximum ozone formation in terms of 1-hr average concentration. Therefore, the critical VOC and NO_x concentrations with regard to the new ozone standard, 8-hr 0.08 ppm, cannot be determined unless the new standard is compared to the concentration in terms of 1-hr average. An EPA study (EPA 2003i) determined the equivalent design values of 8-hour ozone for three key design values of 1-hour ozone: 0.12 ppm (the old 1-hour standard level), 0.121 ppm (used in the 1-hour classifications), and 0.125 ppm (the unrounded, lowest measurable value above the 1-hour standard). Three types of equivalence were determined using the U.S. monitoring data from 1998-2002: mathematical equivalence (regression analysis), stringency equivalence (county count), and health protection equivalence (population count). Table 9.1 summarizes the results of the equivalence value of the 1-hr design values to the 8-hr values. The interpolation of the equivalent values in Table 9.1 reveals that the 8-hr 0.08 ppm is equivalent to 1-hr average of approximately 0.11 ppm. Therefore, the 8-hr 0.08 ppm of ozone can be substituted by the 1-hr 0.11 ppm as the threshold level for the evaluations of critical VOC and NO_X concentrations, critical VOC and NO_X emissions, and VOC and NO_X carrying capacity for photochemical ozone formation impact in this research.

Table 9.1 Equivalence of 8-hour and 1-hour Ozone Design Values
(Source: EPA 2003i)

Mathematic Equivalence

1-hour value design value	Years	95% confidence interval lower bound	Equivalent 8-hour value	95% confidence interval upper bound
0.12	1998-2000	0.089	0.090	0.090
	1999-2001	0.089	0.089	0.089
	2000-2002	0.088	0.089	0.089
0.121	1998-2000	0.090	0.090	0.091
	1999-2001	0.089	0.090	0.090
	2000-2002	0.089	0.089	0.090
0.125	1998-2000	0.092	0.092	0.093
	1999-2001	0.091	0.092	0.092
	2000-2002	0.091	0.092	0.092

Table 9.1 Equivalence of 8-hour and 1-hour Ozone Design Values (Continued)

Stringency Equivalence

1-hour value design value	Years	County count for 1-hour value	Equivalent 8-hour value	County count for 8-hour value
0.12	1998-2000	435	0.092	437
	1999-2001	474	0.091	479
	2000-2002	504	0.091	499
0.121	1998-2000	447	0.093	456
	1999-2001	479	0.091	479
	2000-2002	511	0.092	519
0.125	1998-2000	475	0.094	472
	1999-2001	505	0.093	508
	2000-2002	546	0.094	548

Health Protection Equivalence

1-hour value design value	Years	Cumulative population count for 1-hour value	Equivalent 8-hour value	Cumulative population count for 8-hour value
0.12	1998-2000	111,442,378	0.090	112,801,437
	1999-2001	121,598,978	0.090	123,186,873
	2000-2002	123,309,306	0.090	122,934,040
0.121	1998-2000	114,264,183	0.090	112,801,437
	1999-2001	123,169,845	0.090	123,186,873
	2000-2002	124,225,447	0.090	122,934,040
0.125	1998-2000	121,074,515	0.092	121,378,399
	1999-2001	130,955,867	0.092	133,285,040
	2000-2002	136,549,839	0.093	137,470,512

Critical VOC and NO_x concentrations and Critical VOC and NO_x emissions

Table 9.2 and Table 9.3 present the calculations of the critical VOC and NO_x emission rates, respectively. Note that the old ozone standard is 1-hr 0.12 ppm and the new standard is 8-hr 0.08 ppm (calculated to be equivalent to 1-hr 0.11 ppm). Figure 9.4 and Figure 9.5 are the plots of the critical VOC and NO_x emission rates, respectively, for ten cities. The critical emission of a precursor is the emission that results in the critical atmospheric concentration of that precursor. The critical VOC and NO_x concentrations for both threshold levels (old standard and new standard) are read from the isopleth plots for ten cities at various seasons. A critical emission rate is then calculated using the

Empirical Kinetic Modeling Approach (EKMA-EPA 1989a). The EKMA is a procedure that has been used to estimate emission reductions that are needed to achieve the design level (i.e., the NAAQS) from the peak level (critical case) of ozone, and it is commonly used in the control strategy options in state implementation plans (Finlayson-Pitts and Pitts 2000). The standard level of ozone can be achieved by a percentage reduction of a precursor emission from the existing rate (base case). This approach is based on the assumption that the atmospheric concentration of a primary pollutant is directly proportional to its emission. The calculation of the percentage reduction of VOC in EKMA is expressed as (EPA 1989a):

$$\textit{Percentage Reduction of VOC Emission} = \frac{VOC_B - VOC_C \times 100}{VOC_B} \tag{9.14}$$

Hence, the VOC emission for the critical case scenario can be calculated using the equation modified from the above equation:

$$VOCE_C = \frac{VOC_C \times VOCE_B}{VOC_B} \tag{9.15}$$

Where $VOCE_C$ is the critical VOC emission rate (kg/km^2/hr), $VOCE_B$ is the base case VOC emission rate (kg/km^2/hr), VOC_C is the critical VOC concentration (ppmC), and VOC_B is the base case VOC concentration (ppmC).

Likewise, the NO_X emission for the critical case scenario can be calculated from:

$$NOxE_C = \frac{NOx_C \times NOxE_B}{NOx_B} \tag{9.16}$$

Where $NOxE_C$ is the critical NO_X emission rate (kg/km^2/hr), $NOxE_B$ is the base case NO_X emission rate (kg/km^2/hr), NOx_C is the critical NO_X concentration (ppm), and NOx_B is the base case NO_X concentration (ppm).

Table 9.2 Calculations of Critical VOC Emission

Study Area	Season	Base Case			Critical Case			
					1-hr 0.12 ppm ozone		8-hr 0.08 ppm ozone	
		VOC Conc. (ppmC)	VOC Emission (kg/km^2/hr)	Maximum 1-hr Ozone Conc. (ppm)	VOC Conc. (ppmC)	VOC Emission (kg/km^2/hr)	VOC Conc. (ppmC)	VOC Emission (kg/km^2/hr)
Atlanta	Autumn	0.20	9.2	0.066	1.02	46.9	0.86	39.5
	Spring	0.20	9.5	0.091	0.51	24.2	0.40	19.1
	Summer	0.20	11.9	0.129	0.13	7.7	0.05	3.2
	Winter	0.20	7.7	0.034	1.44	55.4	1.28	49.3
	Average		9.6			33.6		27.8
Boston	Autumn	0.06	6.9	0.027	1.45	166.8	1.28	147.1
	Spring	0.06	7.0	0.041	0.81	94.5	0.71	82.9
	Summer	0.06	10.1	0.075	0.37	62.3	0.30	50.9
	Winter	0.06	5.9	0.005	1.63	160.3	1.48	145.6
	Average		7.5			121.0		106.6
Chicago	Autumn	0.12	19.1	0.028	1.85	294.5	1.63	260.0
	Spring	0.12	18.8	0.045	1.25	195.8	0.66	102.9
	Summer	0.12	20.3	0.062	1.09	184.4	0.91	153.7
	Winter	0.12	18.3	0.013	1.97	300.4	1.76	269.1
	Average		19.1			243.8		196.4
Denver	Autumn	0.20	16.5	0.047	1.83	151.0	1.58	130.5
	Spring	0.20	16.3	0.066	0.94	76.6	0.79	64.6
	Summer	0.20	17.5	0.101	0.41	35.9	0.30	25.9
	Winter	0.20	15.6	0.024	4.26	332.3	3.75	292.3
	Average		16.5			148.9		128.3
Houston	Autumn	0.30	9.6	0.073	0.97	31.0	0.82	26.3
	Spring	0.30	9.8	0.087	0.57	18.6	0.45	14.6
	Summer	0.30	12.0	0.137	0.18	7.2	0.11	4.5
	Winter	0.30	7.9	0.043	2.06	54.2	1.80	47.4
	Average		9.8			27.8		23.2
Los Angeles	Autumn	0.50	7.5	0.075	1.40	21.0	1.19	17.8
	Spring	0.50	7.5	0.134	0.41	6.2	0.35	5.3
	Summer	0.50	8.1	0.160	0.27	4.4	0.22	3.5
	Winter	0.50	7.1	0.049	2.04	29.0	1.80	25.5
	Average		7.6			15.1		13.0
Phoenix	Autumn	0.20	1.1	0.059	0.84	4.6	0.73	4.0
	Spring	0.20	1.1	0.090	0.48	2.6	0.38	2.1
	Summer	0.20	1.5	0.124	0.17	1.3	0.11	0.8
	Winter	0.20	0.9	0.034	1.42	6.4	1.27	5.7
	Average		1.2			3.7		3.2
Pittsburgh	Autumn	0.20	7.2	0.044	1.28	46.1	1.13	40.5
	Spring	0.20	6.9	0.067	0.79	27.3	0.68	23.3
	Summer	0.20	8.2	0.105	0.37	15.2	0.26	10.6
	Winter	0.20	6.7	0.023	2.11	70.7	1.88	63.0
	Average		7.3			39.8		34.4
Seattle	Autumn	0.06	4.7	0.023	1.42	111.2	1.26	98.8
	Spring	0.06	4.6	0.046	1.12	85.9	0.97	74.1
	Summer	0.06	5.7	0.070	0.42	39.9	0.34	32.5
	Winter	0.06	4.3	0.001	2.58	184.9	2.33	167.1
	Average		4.8			105.5		93.1
Washington DC	Autumn	0.15	24.2	0.043	0.86	138.7	0.76	122.9
	Spring	0.15	24.0	0.062	0.65	104.0	0.56	89.2
	Summer	0.15	26.5	0.103	0.26	45.9	0.19	33.8
	Winter	0.15	22.3	0.016	1.37	203.7	1.24	184.5
	Average		24.3			123.1		107.6

Table 9.3 Calculations of Critical NO_x Emission

Study Area	Season	Base Case			Critical Case			
					1-hr 0.12 ppm ozone		8-hr 0.08 ppm ozone	
		NOx Conc. (ppm)	NOx Emission (kg/km^2/hr)	Maximum 1-hr Ozone Conc. (ppm)	NOx Conc. (ppm)	NOx Emission (kg/km^2/hr)	NOx Conc. (ppm)	NOx Emission (kg/km^2/hr)
Atlanta	Autumn	0.04	3.9	0.039	0.13	13.0	0.12	11.8
	Spring	0.04	3.9	0.049	0.10	10.0	0.09	9.1
	Summer	0.04	4.2	0.058	0.09	9.1	0.08	8.3
	Winter	0.04	4.0	0.037	0.15	14.5	0.13	13.1
	Average		4.0			11.6		10.6
Boston	Autumn	0.02	3.7	0.019	0.15	26.9	0.13	24.1
	Spring	0.02	3.8	0.029	0.09	16.7	0.08	15.3
	Summer	0.02	4.2	0.034	0.07	15.4	0.07	14.2
	Winter	0.02	3.8	0.016	0.19	35.8	0.17	31.7
	Average		3.9			23.7		21.3
Chicago	Autumn	0.04	8.0	0.031	0.18	35.8	0.16	32.3
	Spring	0.04	8.1	0.039	0.14	27.6	0.12	24.9
	Summer	0.04	7.9	0.076	0.12	24.3	0.10	20.3
	Winter	0.04	11.5	0.026	0.24	67.9	0.21	60.0
	Average		8.9			38.9		34.4
Denver	Autumn	0.03	11.2	0.020	0.23	84.3	0.20	74.5
	Spring	0.03	11.3	0.032	0.13	47.0	0.11	42.6
	Summer	0.03	12.4	0.041	0.10	39.2	0.09	35.5
	Winter	0.03	11.8	0.020	0.30	118.2	0.28	110.1
	Average		11.7			72.2		65.7
Houston	Autumn	0.03	4.0	0.031	0.13	17.2	0.11	15.4
	Spring	0.03	4.1	0.036	0.11	14.5	0.10	13.3
	Summer	0.03	4.7	0.056	0.07	10.5	0.06	9.5
	Winter	0.03	5.3	0.020	0.23	40.1	0.20	35.4
	Average		4.5			20.6		18.4
Los Angeles	Autumn	0.06	3.0	0.048	0.17	8.2	0.15	7.5
	Spring	0.06	3.0	0.120	0.06	3.0	0.05	2.7
	Summer	0.06	3.3	0.134	0.05	2.9	0.05	2.7
	Winter	0.06	2.9	0.038	0.24	11.5	0.20	10.0
	Average		3.1			6.4		5.7
Phoenix	Autumn	0.03	0.7	0.037	0.11	2.4	0.10	2.2
	Spring	0.03	0.7	0.045	0.09	2.0	0.08	1.8
	Summer	0.03	0.8	0.056	0.07	1.8	0.06	1.7
	Winter	0.03	0.6	0.026	0.16	3.3	0.14	2.9
	Average		0.7			2.4		2.2
Pittsburgh	Autumn	0.03	2.2	0.030	0.14	9.8	0.13	9.0
	Spring	0.03	2.2	0.038	0.10	7.5	0.09	6.8
	Summer	0.03	2.3	0.035	0.11	8.6	0.10	7.7
	Winter	0.03	3.0	0.019	0.25	24.7	0.22	22.0
	Average		2.4			12.6		11.4
Seattle	Autumn	0.02	1.5	0.019	0.14	10.6	0.13	9.7
	Spring	0.02	1.5	0.021	0.13	9.7	0.12	8.9
	Summer	0.02	1.6	0.034	0.08	6.1	0.07	5.5
	Winter	0.02	1.4	0.010	0.45	30.8	0.33	22.7
	Average		1.5			14.3		11.7
Washington DC	Autumn	0.03	12.7	0.041	0.09	39.8	0.09	36.0
	Spring	0.03	12.8	0.046	0.08	35.0	0.07	31.9
	Summer	0.03	13.6	0.053	0.07	31.8	0.06	28.9
	Winter	0.03	14.7	0.027	0.16	77.1	0.14	69.1
	Average		13.5			45.9		41.5

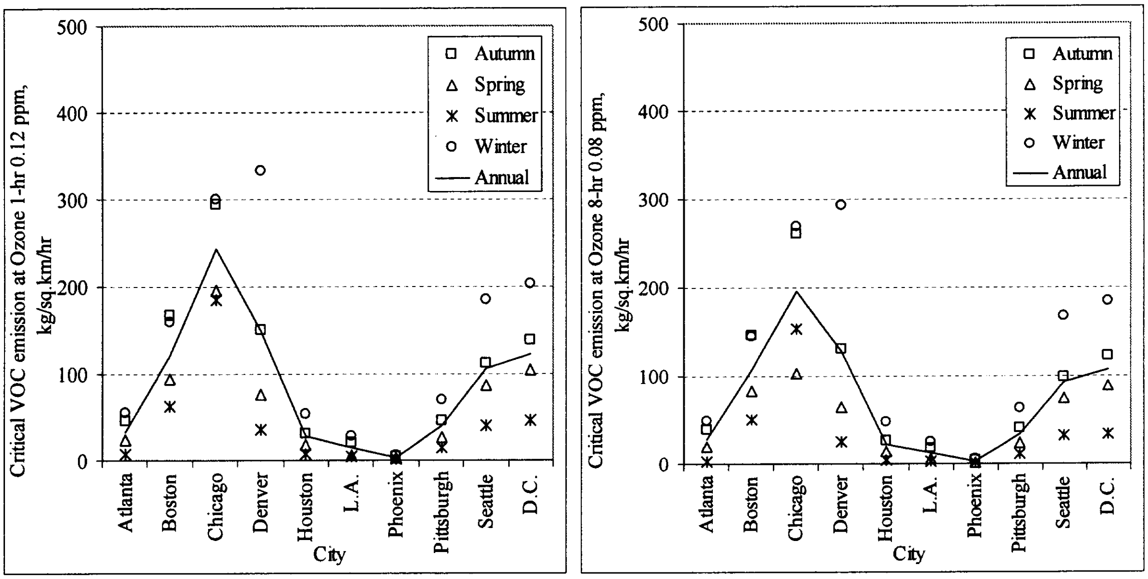

Figure 9.4 Seasonal and average critical VOC emission for ten cities. Left panel is for the old ozone standard (1-hr 0.12 ppm) while the right panel is for the new ozone standard (8-hr 0.08 ppm).

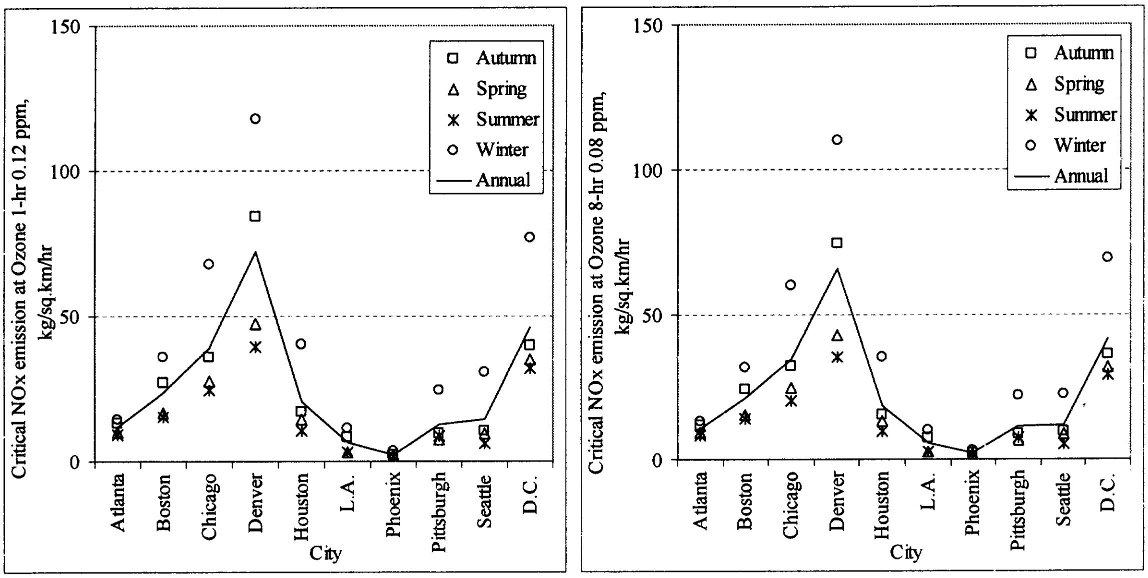

Figure 9.5 Seasonal and average critical NO_x emission for ten cities. Left panel is for the old ozone standard (1-hr 0.12 ppm) while the right panel is for the new ozone standard (8-hr 0.08 ppm).

9.5.2 Carrying Capacity Estimate

The simulation using the OZIPR for the base case scenario reveals that ozone concentrations have already exceeded the standard under existing emission conditions in some areas. In such cases, the critical VOC or NO_X emission is lower than the VOC or NO_X emission of the base case (the existing emission rate). These areas are Atlanta, Houston, and Phoenix in the summer, and Los Angeles in the spring and summer (see also Table 9.2). For the NO_X-sensitive regime, Los Angeles is the only city that ozone concentrations have already exceeded the standard under existing emission conditions. It should be noted that the cities in lower latitudes (Atlanta, Houston, Los Angeles, and Phoenix) tend to have low critical VOC and NO_X emissions due to their higher average temperature and sunlight intensity.

If the assumption is made that the atmospheres of these ten cities represent the atmospheric conditions across the U.S., a national scale carrying capacity for VOC and NO_X emissions can be estimated using critical VOC and NO_X emission rates obtained from OZIPR simulations. The seasonal critical VOC emission rates obtained from the OZIPR simulations range from 1.3 $kg/km^2/hr$ (Phoenix-summer) to 332.3 $kg/km^2/hr$ (Denver-winter) with the arithmetic mean of 83.1 $kg/km^2/hr$ when the threshold level is the old ozone standard. These values are 0.8, 292.3, and 70.4 $kg/km^2/hr$, respectively, when the threshold level is the new ozone standard. The seasonal critical NO_X emission rates range from 1.8 $kg/km^2/hr$ (Phoenix-summer) to 118.2 $kg/km^2/hr$ (Denver-winter) with the arithmetic mean of 24.0 $kg/km^2/hr$ when the threshold level is the old ozone standard. These values are 1.7, 110.1, and 21.6 $kg/km^2/hr$, respectively, when the threshold level is the new ozone standard.

The deviation of the critical emission rates among U.S. cities is due to the spatial variability of multiple parameters (temperature, humidity, pressure, sunlight intensity, mixing height, existing VOC and NO_X emissions and concentrations), and it is difficult to relate the critical VOC emission to every parameter or even some key parameters such as the average temperature, light intensity, and mixing height. However, the national scale carrying capacity can be estimated from the median critical VOC and NO_X emission rates derived from probability analysis for those ten cities. The median critical emission rates derived from probability analysis is used, instead of the arithmetic means, so the bias due to outlier data is minimized. Furthermore, the probability function associated with the carrying capacity estimate in this research can also be determined. This probability function may be useful for further uncertainty assessment. Upon the availability of the median critical VOC and NO_X emission rates, the carrying capacities for VOC and NO_X can then be calculated using the following equations:

$$U.S.\ CC_{VOC} = Critical\ VOC\ Emission\ Rate\ (kg/km^2/hr) \times U.S.\ Area\ (km^2) \qquad (9.17)$$

$$U.S.\ CC_{NOx} = Critical\ NO_x\ Emission\ Rate\ (kg/km^2/hr) \times U.S.\ Area\ (km^2) \qquad (9.18)$$

Figure 9.6 and Figure 9.7 present the log normal probability plots for VOC and NO_X critical concentrations and probability function (Z-score). The critical VOC and NO_X emission rates are determined for different probabilities: 50th percentile, 50% confidence interval (25th – 75th percentiles), and 95% confidence interval (2.5th -97.5th percentiles).

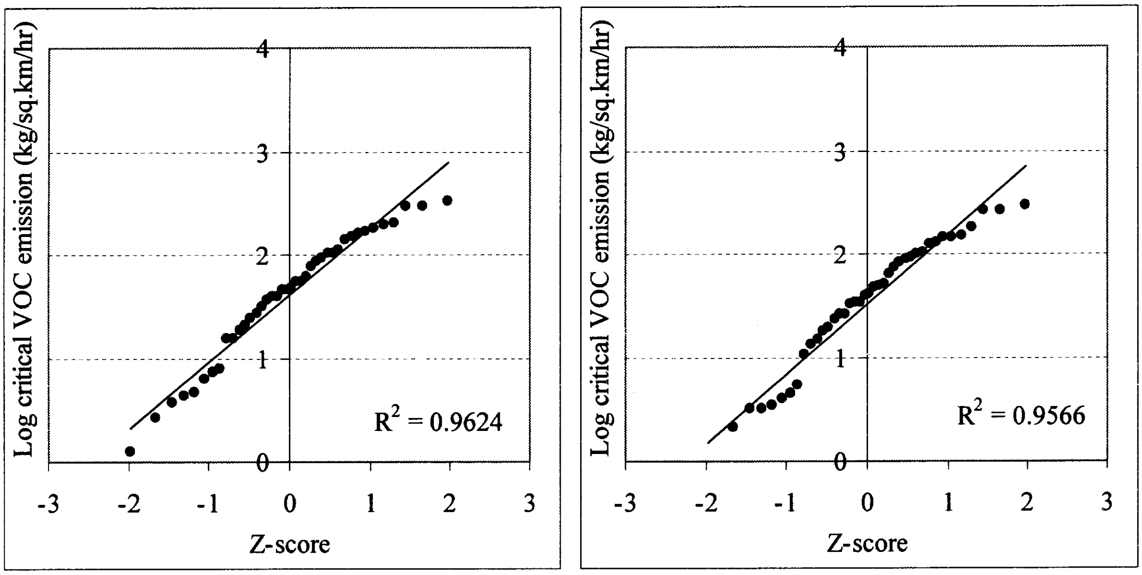

Figure 9.6 Log normal probability plots for critical VOC emission and probability function (Z-score) when threshold levels are 1-hr 0.12 ppm ozone (left panel) and 8-hr 0.08 ppm ozone (right panel).

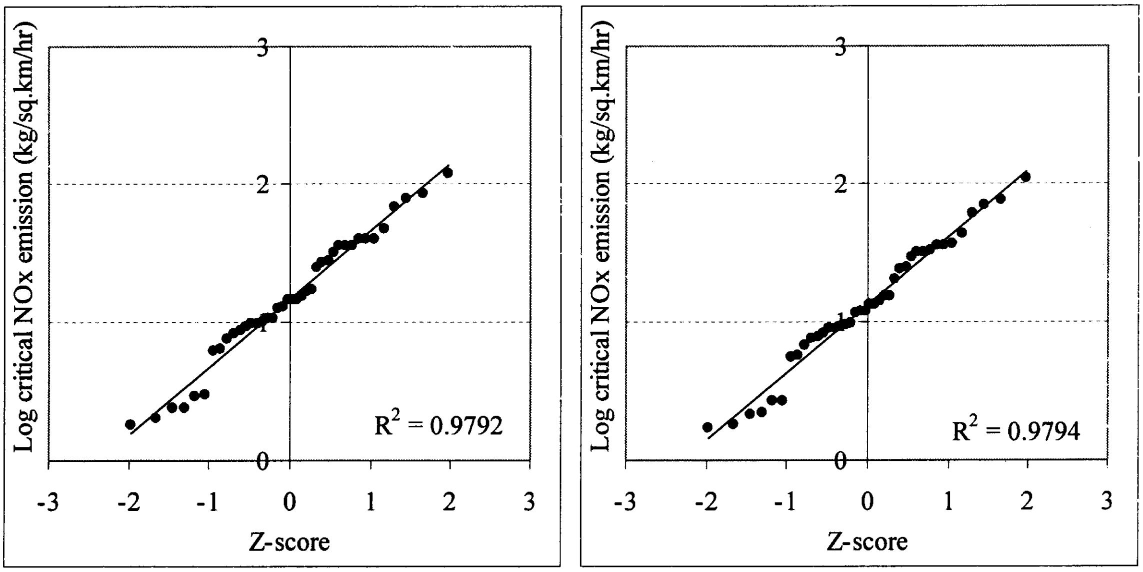

Figure 9.7 Log normal probability plots for critical NO_X emission and probability function (Z-score) when threshold levels are 1-hr 0.12 ppm ozone (left panel) and 8-hr 0.08 ppm ozone (right panel).

The carrying capacities for VOC and NOx can be calculated using Equation (9.17) and Equation (9.18), respectively. These are presented in Table 9.4 (for VOC) and Table 9.5 (for NO_X). At the 50th percentile, the U.S. carrying capacities for VOC and NO_X are 1.66×10^{12} kg/yr and 5.98×10^{11} kg/yr, respectively, for the threshold level of 1-hr 0.12 ppm ozone and 1.34×10^{12} kg/yr and 5.40×10^{11} kg/yr, respectively, for the threshold level of 8-hr 0.08 ppm ozone. It should be noted that the emissions of VOC and NO_X can be in reality from both anthropogenic and natural sources. As a result, the carrying capacity values estimated in Table 9.4 and Table 9.5 are the natural carrying capacities that account for both sources. The calculation of the economic carrying capacity, which is used in the STM context, is made in Subsection 9.5.4.

Table 9.4 U.S. Carrying Capacity Estimates for VOC Emission at Different Probabilities

Probability percentile	Old ozone standard as the threshold (1-hr 0.12 ppm)		New ozone standard as the threshold (8-hr 0.08 ppm)	
	Critical emission rate ($kg/km^2/yr$)	Natural carrying capacity (kg/yr)	Critical emission rate ($kg/km^2/yr$)	Natural carrying capacity (kg/yr)
2.5	2.12	1.75E+11	1.51	1.25E+11
25	14.64	1.21E+12	11.31	9.32E+11
50	**40.25**	**3.31E+12**	**32.49**	**2.68E+12**
75	110.71	9.12E+12	93.27	7.68E+12
97.5	763.00	6.28E+13	697.78	5.75E+13

Table 9.5 U.S. Carrying Capacity Estimates for NO_X Emission at Different Probabilities

Probability percentile	Old ozone standard as the threshold (1-hr 0.12 ppm)		New ozone standard as the threshold (8-hr 0.08 ppm)	
	Critical emission rate ($kg/km^2/yr$)	Natural carrying capacity (kg/yr)	Critical emission rate ($kg/km^2/yr$)	Natural carrying capacity (kg/yr)
2.5	1.58	1.30E+11	1.44	1.19E+11
25	6.78	5.58E+11	6.13	5.05E+11
50	**14.53**	**1.20E+12**	**13.11**	**1.08E+12**
75	31.16	2.57E+12	28.01	2.31E+12
97.5	133.52	1.10E+13	119.32	9.83E+12

9.5.3 Characterization of Volatile Organic Compounds

The procedure to estimate the carrying capacity for VOC emission has been presented in the previous sections. However, the unit of the estimated carrying capacity is in kilograms of "typical" VOC per year. It is necessary to characterize the carrying capacity by a common unit or an equivalent unit before applying it to LCA practice. Similarly, a VOC emission from a source needs to be characterized by a common or equivalent species as well.

The sources of VOC can be automobile exhaust, vapors from cleaning solvents, and other industrial or household emissions. Individual VOCs can differ significantly in their effects on ozone formation due to differences in their reactivity and associated reaction rates. Therefore, a reactivity scale, in which the ranking is based on the amount of ozone formed from each VOC, is required for LCA. Two widely-used methods for the characterization of VOCs in terms of ozone formation are the MIR (Maximum Incremental Reactivity) and the POCP (Photochemical Ozone Creation Potential).

Maximum Incremental Reactivity (MIR)

MIR has been developed as a reactivity scale that accounts approximately for all factors and it is based on photochemical model calculations using typical U.S. atmospheric conditions (Carter 1994; 1995; 2000; Carter et al. 1995). MIR approximates the potential of different VOCs to form ozone under conditions where the availability of NO_x does not limit ozone formation. MIR is basically defined as the change in the amount of ozone caused by a species of VOC being added, divided by the amount of added VOC (Equation (9.19)). This can also be thought of as the partial derivative of ozone with respect to the emission of the VOC.

$$MIR = \Delta O_3/\Delta VOC \qquad (9.19)$$

The incremental reactivities depend significantly on the environmental conditions, particularly on the availability of NO_x. In general, VOCs tend to have the largest incremental reactivities under relatively high NO_x conditions, and have much lower reactivities under conditions of limited NO_x. Other environmental aspects, such as the nature of other organics, dilution, and humidity, can also be important in affecting VOC reactivities. The fact that incremental reactivities depend on environmental conditions means that no single scale can predict incremental reactivities. Thus the concept "reactivity scale" has been used to simplify the complexities of the effects of VOC emissions on ozone formation (Carter 1994). The incremental reactivities were obtained from both computer modeling and experimental results in several scenarios. Carter et al. (Carter 1994; 1995; 2000; Carter et al. 1995) developed the maximum incremental reactivity (MIR) scenarios by adjusting the NO_x inputs so that the highest incremental reactivity was achieved. These MIR scenarios represent NO_x conditions where organic emissions have the greatest effect on ozone formation. MIR has been used extensively for LCA purposes, for example, in an LCA project of EPA that employs MIR to calculate the photochemical ozone formation indicator (Bare et al. 2003; EPA 2000b; 2003g).

Photochemical ozone-forming potential for a VOC can be converted to a common species, ethylene (C_2H_4). Ethylene is one of the most common VOCs emitted from automobile exhaust and it is widely used in LCA as a common reference VOC (Hauschild and Wenzel 1998). A VOC species *i* can be converted to an "ethylene equivalent" using the equation:

$$kg\ C_2H_4\text{-}eq = (MIR_i/MIR_{Ethylene}) \times kg\ VOC_i \qquad (9.20)$$

MIR has a unit of g O_3/g VOC. MIR value for ethylene is 9.078 kg. Therefore, Equation (9.20) can be expressed as:

$$kg\ C_2H_4\text{-}eq \quad = \quad (MIR_i/9.078) \times kg\ VOC_i \qquad (9.21)$$

This equation can be used for the characterization of a VOC in terms of C_2H_4-equivalent. MIR values for VOCs have been updated and revised periodically using the results from contributing research (Carter 2000). The most recent MIR list for approximately 700 VOCs has been available since November 2000 (Table C.5(a), Appendix C). The average MIR for these 700 VOCs is 3.02 g O_3/g VOC.

Photochemical Ozone Creation Potential (POCP)

POCP was originally developed to assess various scenarios for VOC emissions causing regional scale ozone formation over periods of up to five days in northwest Europe (Derwent and Jenkin 1991). POCP is very complicated and requires many reaction mechanisms in the simulation using a photochemical trajectory model. The POCP value for a particular VOC was calculated from the results of a separate model experiment, each variant of which provided base case scenarios. In each separate model, a certain amount of VOC emission was added and the resulting ozone increment was observed. The ozone increment from the addition of the VOC was then compared to the reference VOC, ethylene. The POCP for a particular VOC is defined as (Derwent et al. 1998):

$$POCP_i \quad = \frac{\text{Ozone increment with the i species}}{\text{Ozone increment with ethylene}} \times 100\ \% \qquad (9.22)$$

POCP is widely used in Europe, including the establishment of air quality standards for ozone by UNECE (Jenkin and Hayman 1999) and several LCA studies (Goedkoop 1998; Hauschild and Wenzel 1998). Table C.5(b) (Appendix C) provides POCP values for 120 VOCs estimated by Derwent et al. (1998). Similar to MIR, POCP can also be used to convert a VOC emission to an "ethylene equivalent" in terms of photochemical ozone-forming potential. The equation for this calculation is:

$$kg\ C_2H_4\text{-}eq \quad = \quad POCP_i \times kg\ VOC_i \qquad (9.23)$$

The major differences between MIR and POCP are threefold. First, POCP was developed using European scenarios, while MIR was developed using North American scenarios. Second, POCP is based on a photochemical trajectory model of VOC transport over Europe, while MIR is based on a single box model for U.S. conditions. Third, POCP was simulated for a time span of up to five days, while MIR was simulated over a time span of one day. However, there is a reasonable correlation between POCP and MIR values for VOCs in general (Jenkin and Hayman 1999). Both approaches tend to give similar predictions with regard to the relative importance of different classes of VOC. However, MIR is used in this research for the following reasons: it was developed using U.S. atmospheric conditions, its scenarios represent NO_x conditions where organic emissions have the greatest effect on ozone formation, and its values are available for approximately 700 VOCs, which is greater than the available number for POCP. The characterization of a VOC in terms of C_2H_4-eq, the reference VOC widely used in LCIA (Goedkoop 1998; Goedkoop and Spriensma 2001; Guinee et al. 2001; Hauschild and Wenzel 1998), can be made using Equation (9.21).

Characterization of Carrying Capacity In Terms of Ethylene-Equivalent

In LCA, inventory data are sometimes mentioned only as “total VOC emissions” or “CxHy” instead of being differentiated according to species. One way of addressing this issue is to use the average POCP or MIR values as the POCP or MIR for an unidentified VOC (Hauschild and Wenzel 1998; Derwent et al. 1996). If this approach is applied, the MIR for a “typical VOC” or “typical CxHy” will be 3.02 kg O_3/kg VOC (the average MIR value for 700 VOCs). By dividing this value by the MIR for ethylene (3.02/9.078 = 0.333), the average MIR value in terms of ethylene equivalent (kg C_2H_4/kg VOC) is similar to the average POCP for a “typical VOC” or a“typical CxHy” which is 0.398 kg C_2H_4/kg VOC as used in EI95 (Goedkoop 1998).

The critical VOC emission rates in the carrying capacity evaluation (Subsection 9.5.2) are also presented in the unit of “kg VOC/km^2/hr” as obtained from the OZIPR results. It is more appropriate to express the carrying capacity in terms of a reference VOC; i.e., kg C_2H_4-eq/yr. However the lumped or unidentified VOC cannot be converted to the unit of ethylene equivalent without using an appropriate MIR or POCP as the equivalency factor. The average MIR or POCP values, as mentioned in the previous paragraph, can be used to convert the carrying capacity in terms of kg VOC/yr to kg C_2H_4-eq/yr as well. However, since the chemical composition of the typical VOC emissions as the input for the OZIPR is identified, it is better to calculate the equivalency factor upon the availability of the information.

In the input data used in the OZIPR simulation (Subsection 9.5.1), the composition of the total (or typical) VOC emissions emitted from those ten cities was classified into ten different VOC groups and species (EPA 1999c). In the calculation of the equivalency factor, specific hydrocarbons with known MIRs are selected to represent each VOC group. Equivalency factors for those ten cities can then be calculated as presented in Table 9.6. The average equivalency factor for those ten cities is 0.481 kg C_2H_4/kg VOC, which is higher than the average MIR from 700 VOCs (0.333 kg C_2H_4/kg VOC) and the average POCP (0.398 kg C_2H_4/kg VOC). A U.S. carrying capacity for VOC emission can then be converted to the ethylene-equivalent using Equation (9.24). The natural carrying capacity for VOC emission (Table 9.4) can also be expressed in terms of C_2H_4-equivalent as presented in Table 9.7.

$$CC\ (kg\ C_2H_4\text{-}eq/yr) \quad = \quad CC\ (kg\ VOC/yr) \times 0.481\ \frac{kg\ C_2H_4}{kg\ VOC} \qquad (9.24)$$

9.5.4 Natural Burden and Economic Carrying Capacity

Some VOCs and NO_x are emitted into the atmosphere from natural sources such as wetlands and vegetation for VOCs and soil and lightning for NO_x. The estimated U.S. emissions of NMVOC, which is active in photochemical ozone formation, from natural sources are 4.5×10^{10} kg of carbon/yr (5.2×10^{10} kg C_2H_4-eq/yr) (Guenther et al. 2000). For NO_x, the estimated U.S. emissions from natural sources are 1.1×10^{9} kg N/yr (3.5×10^{9} kg NO_x/yr) (Guenther et al. 2000). These emissions are considered as the natural burden in natural carrying capacity. The economic carrying capacity, which will be used in the STM, can be calculated from Equation (9.25)(Dickinson et al. 2001). The economic carrying capacity that is left after subtraction of the natural burden. This is summarized in Table 9.8 for VOC emission and Table 9.9 for NO_x emission.

$$Carrying\ capacity \quad = \quad Natural\ carrying\ capacity - Natural\ burden \qquad (9.25)$$

Table 9.6 Calculation of Equivalency Factor for Typical VOC and Ethylene Equivalent

Hydrocarbon group/species[1]	Composition of hydrocarbon species in emission from 10 cities (%)[2]										MIR[3]		
	Atlanta	Boston	Chicago	Denver	Houston	Los Angeles	Phoenix	Pittsburgh	Seattle	Washington DC	Representative hydrocarbon species	Molecular weight	MIR (kg O_3/kgVOC)
ALK4	21.7	19.7	17.1	24.6	27.9	16.7	17.4	18.5	19.7	19.1	Butane (C4H10)	58	1.329
ALK7	21.3	20.9	27.2	24.7	20.7	32.0	28.7	21.7	20.9	21.6	Branched C12 Alkanes (C12H26)	170	0.796
ETHE	3.3	4.4	2.3	4.9	3.6	2.7	3.5	3.8	4.4	4.2	Ethylene (C2H4)	28	9.078
PRPE	1.1	1.3	1.5	1.1	1.9	4.9	1.4	1.1	1.3	1.7	Propylene (C3H6)	42	11.576
TBUT	9.5	10.6	12.0	6.3	7.2	5.8	7.5	11.7	10.6	8.3	Trans-2-butene (C4H8)	56	13.912
TOLU	8.3	9.0	8.5	7.9	5.1	5.3	8.0	5.7	9.0	10.3	Toluene (C7H8)	92	3.973
XYLE	5.2	5.4	5.3	4.8	3.3	3.9	5.6	4.4	5.4	10.0	Xylene (C8H10)	106	7.489
TMBZ	14.2	11.9	14.1	9.5	10.2	8.9	9.3	11.1	11.9	9.0	1,2,4 Trimethylbenzene ((CH3)3C6H3)	120	7.179
RCHO	0.9	0.9	0.9	0.9	0.9	2.5	0.9	0.9	0.9	0.9	C5 Aldehydes (C5H10O)	86	5.763
NRHC	14.5	15.9	11.1	15.5	19.1	17.4	17.8	21.0	15.9	14.9	Methane (CH4)	16	0.014
Total	100	100	100	100	100	100	100	100	100	100			
C_2H_4-equivalent	50.6	52.2	54.3	42.6	42.3	43.2	43.4	49.8	52.2	50.6			

Table 9.7 U.S. Carrying Capacity for VOC Emission in Terms of Ethylene-Equivalent

Probability percentile	Old ozone standard as the threshold (1-hr 0.12 ppm)		New ozone standard as the threshold (8-hr 0.08 ppm)	
	Natural carrying capacity (kg VOC/yr)	Natural carrying capacity (kg C_2H_4-eq/yr)	Natural carrying capacity (kg VOC/yr)	Natural carrying capacity (kg C_2H_4-eq/yr)
2.5	1.75E+11	8.42E+10	1.25E+11	6.01E+10
25	1.21E+12	5.82E+11	9.32E+11	4.48E+11
50	**3.31E+12**	**1.59E+12**	**2.68E+12**	**1.29E+12**
75	9.12E+12	4.39E+12	7.68E+12	3.69E+12
97.5	6.28E+13	3.02E+13	5.75E+13	2.77E+13

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[1] ALK4 = Alkanes with 3, 4, and 5 carbons (primary butane and isopentane); ALK7 = Heavier alkanes (6 or more carbons); ETHE = Ethylene; PRPE = Propene (propylene); TBUT = Heavier alkenes (4 or more carons-primarily trans-2-butene; TOLU = Toluene (also ethylbenzene); XYLE = Xylene; TMBZ = Trimethylbenzene (and heavier aromatics); RCHO = Heavier aldehydes; NRHC = Non-reactive hydrocarbons.

[2] From EPA (1999c).

[3] See Table C.5(a), Appendix C.

Table 9.8 U.S. Economic Carrying Capacity for VOC Emission
(Natural burden = 5.2×10^{10} kg C_2H_4-eq/yr)

Probability percentile	Old ozone standard as the threshold (1-hr 0.12 ppm)		New ozone standard as the threshold (8-hr 0.08 ppm)	
	Natural carrying capacity (kg C_2H_4-eq/yr)	Economic carrying capacity (kg C_2H_4-eq/yr)	Natural carrying capacity (kg C_2H_4-eq/yr)	Economic carrying capacity (kg C_2H_4-eq/yr)
2.5	8.42E+10	3.22E+10	6.01E+10	8.10E+09
25	5.82E+11	5.30E+11	4.48E+11	3.96E+11
50	**1.59E+12**	**1.54E+12**	**1.29E+12**	**1.24E+12**
75	4.39E+12	4.34E+12	3.69E+12	3.64E+12
97.5	3.02E+13	3.01E+13	2.77E+13	2.76E+13

Table 9.9 U.S. Economic Carrying Capacity for NO_x Emission
(Natural burden = 3.5×10^9 kg NO_x/yr)

Probability percentile	Old ozone standard as the threshold (1-hr 0.12 ppm)		New ozone standard as the threshold (8-hr 0.08 ppm)	
	Natural carrying capacity (kg/yr)	Economic carrying capacity (kg/yr)	Natural carrying capacity (kg/yr)	Economic carrying capacity (kg/yr)
2.5	1.30E+11	1.27E+11	1.19E+11	1.16E+11
25	5.58E+11	5.55E+11	5.05E+11	5.02E+11
50	**1.20E+12**	**1.20E+12**	**1.08E+12**	**1.08E+12**
75	2.57E+12	2.57E+12	2.31E+12	2.31E+12
97.5	1.10E+13	1.10E+13	9.83E+12	9.83E+12

9.5.5 Spatial Distributions of Emissions

The spatial variation in terms of the receiving environment is not addressed in this carrying capacity estimate because it is assumed that the same ozone standard has been developed to protect human health and welfare for the entire nation as the primary purpose. Therefore, the local or region-specific carrying capacity is not considered in the analysis. However, the site-dependent characterization addresses the spatial variation of the emission of photochemical precursors.

In the TRACI (Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts: Bare et al. 2002; EPA 2003a), an approach has been developed to address the spatial variation of the VOC and NO_x emission for photochemical ozone formation impact in the U.S. The site-dependent characterization factors (termed as site factors hereafter) in TRACI are basically the fate factors which take into account the transport and distribution of VOC or NO_x under site-specific atmospheric pathways and chemistry processes on a regional scale. A higher site factor indicates more contribution to photochemical ozone formation impact on North America. By using the site factors, the overall site-dependent characterization for an emission of VOC can be modified from Equation (9.26) and is expressed as:

$$kg\ C_2H_4\text{-}eq(site\text{-}dependent) = SF_{Smog} \times (MIR_i/9.078) \times kg\ VOC_i \qquad (9.26)$$

And for NO_x emission:

$$kg\ NO_x\ (site\text{-}dependent) = SF_{Smog} \times kg\ NO_x \qquad (9.27)$$

Where SF_{Smog} is a site factor for an emission of VOC or NO_x from a U.S. site. Table C.5(c) in Appendix C presents the site factors for the state and regional levels. The average site factor for the U.S. is 1.0 and should be used where the actual site cannot be identified precisely.

9.6 Summary

The carrying capacities for VOC and NO_x emissions are estimated, as summarized in Table 9.10, for the U.S. following the threshold-oriented method developed in this research. The carrying capacity is calculated from the average critical VOC and NO_x emission rates of ten U.S. metro cities. The evaluation of carrying capacity for photochemical ozone formation impact employs the OZIPR model to simulate the critical VOC and NO_X emission rates, i.e., the emissions that cause the atmospheric ozone concentration that just reaches the threshold level. The NAAQS for ozone (old 1-hr 0.12 ppm and new 8-hr 0.08 ppm) is selected as the threshold level for photochemical ozone formation. The carrying capacities for NO_X emissions are much higher than the U.S. normalization value (3.38×10^7 kg NO_X/year) derived from 1999 emission data for used with TRACI (Bare et al. 2006). This means that the current U.S. emissions do not exceed the environmental carrying capacity.

Table 9.10 Carrying Capacities for Photochemical Ozone formation Impact

Primary pollutant	Carrying capacity estimate	
	Old ozone standard as the threshold (1-hr 0.12 ppm)	New ozone standard as the threshold (8-hr 0.08 ppm)
VOC emission (kg C_2H_4-eq/yr)	1.54E+12	1.24E+12
NO_x emission (kg NO_x/yr)	1.20E+12	1.08E+12

The carrying capacity estimated in this research is conservatively biased because it is assumed that NO_X and VOC concentrations are in excess of the level required to cause a critical (maximum) ozone concentration for both VOC-sensitive and NO_X-sensitive regimes, respectively. Practically, in some areas with high emission of VOC, NO_X may not be available at a level high enough to undergo a photochemical reaction with all of the VOC and vice versa. If a precursor is only available in small amounts, this will level up the critical emission of the other precursor so that the maximum ozone formation is reached (causing a higher carrying capacity, which will result in a less conservative estimate). Furthermore, critical emission of a precursor cannot be reached in the case of a very low concentration of the other precursor; therefore it is assumed that there is always the other precursor in excess for the greatest ozone formation. Another assumption that causes the carrying capacity to be conservatively biased is that the ozone formation is carried over to night-time in the same manner as in daytime. Without this assumption, the carrying capacity cannot be reached because the ozone formation will not occur in the absence of sunlight, which will imply unlimited VOC or NO_x emission at night-time for the photochemical formation impact.

The conversion of a VOC to ethylene equivalent can be carried out using either MIR or POCP values. However, the MIR is recommended by this research because: it was developed using U.S. atmospheric conditions, its scenarios represent NO_x conditions where organic emissions have the greatest effect on ozone formation, and its values are available for approximately 700 VOCs. The conversion factor of a "typical VOC" to C_2H_4-equivalent is estimated to be 0.481 kg VOC/kg C_2H_4 using MIR values. This conversion factor is higher than that used in EI95.

The proposed carrying capacity estimate using OZIPR should be used in the U.S. only since the meteorological and emission conditions vary spatially. To estimate the carrying capacity for other parts of the world, corresponding data may be needed as the OZIPR input. Otherwise, other appropriate photochemical models may be used. A more sophisticated model will certainly improve the accuracy of the carrying capacity estimate.

The carrying capacity estimate can be revised and modified when the input parameters for photochemical modeling are refined as well as the change in reference threshold ozone concentration when the new regulatory level is promulgated or a new threshold level is selected. The carrying capacity estimate may also be refined should the input data for the OZIPR be prepared for other cities. However, the revision of the carrying capacity following the method presented here may be difficult and time consuming because there are a number of documents and information that would need to be reviewed. In this case, a less difficult and simpler approach may be used.

CHAPTER 10

HUMAN TOXICITY IMPACT

10.1 Overview

After being released to the environment, some chemicals have the potential to migrate from medium to medium. A danger to human health is presented when contaminated media are consumed, inhaled, or brought into dermal contact. In conventional LCA, human health impacts of products are addressed by aggregating the toxic chemical releases. The aggregation of the toxic chemical releases can be made by using a method to characterize toxic chemicals in terms of relative health hazards toxicity. The aggregation can be made after toxic equivalency potentials (or toxicity scoring system) are applied to individual chemicals. The better product or alternative in terms of this impact is the one that releases the least amount of aggregated toxics.

Examples of the characterization methods to determine toxic equivalency potentials of chemicals are the human toxicity potential (Guinee and Heijungs 1993), the toxicity-based scoring (Horvath et al. 1995), the health hazards scoring (Srinivasan et al. 1995 as cited in Hertwich et al. 1997), the sustainable process index (Narodoslawsky and Krotscheck 1995), and the concentration/toxicity equivalency (Jia et al. 1996). The human toxicity potential (HTP) is considered the most sophisticated approach and is widely used in LCA practice (Hertwich et al. 1998; Ecobalance 2000). The HTP takes into account the fate, transfer, intake/exposure, and effect in the determination using multimedia fate modeling. There are a few sets of HTP values that have been proposed recently. Hertwich et al. (2001) presented a set of 330 HTP values for air and surface-water emissions of chemicals based on U.S. settings for environmental modeling.

Huijbregts et al. (2001b; 2001c; 2001d) calculated HTP values for 181 chemicals based on European and global settings. A new set of HTP values for several thousand chemicals is being proposed using Western Europe settings (Jolliet et al. 2003).

In EI95 (Goedkoop 1998), chemicals that pose impact to human health are classified into four impact categories: heavy metals, carcinogens, winter smog, and pesticides. An aggregation or trade-off can be made for chemicals within the same impact categories. For heavy metals, lead was used as the surrogate for heavy metals in air impact. Atmospheric lead concentration in Europe was considered to be less than the adverse effect level for the exposure to human. However, it was assumed that a reduction factor of five for lead (as well as other heavy metals) would be required to pose no effect on human health. The reduction factor of five was also used to determine an emission target level of cadmium, the reference substance for heavy metals, in water.

For carcinogens, the EI95 method assumed that the loss due to cancer would be one case per year per million inhabitants if the concentrations of polyaromatic hydrocarbons (PAHs) in Europe were ten times lower that the existing levels. A reduction factor of ten was therefore selected. For winter smog, it was assumed that a reduction in SO_2 emission of more than 80% was necessary to eliminate the smog periods. As a result, a reduction factor of five was selected. For pesticides, it was assumed that a reduction factor of 25 for pesticides was necessary to reduce the 65% contamination of groundwater across Europe to 10%. By using these reduction factors, the European target values of 5.4×10^6 Pb-eq kg/yr, 5.4×10^5 PAH-eq kg/yr, 9.4×10^9 SO_2-eq kg/yr, and 1.92×10^7 kg/yr for heavy metals, carcinogens, winter smog, and pesticides, respectively, were used in EI95.

Human toxicity is considered as a non-homogeneous impact category (Assies 1998), wherein the adverse impact to human health is caused by several different toxic substances and these substances are not interchangeable in terms of human exposure. This is because different toxic chemicals affect different organs and systems and it is not possible to calculate any synergistic effect on human health caused by all chemicals. In contrast with the other methods for conventional LCA, the carrying capacity for human toxicity impact for the STM must be estimated individually for specific chemicals and aggregation or trade-off in terms of toxicity potential among toxic chemicals is not allowed. In other words, the carrying capacity for human toxicity impact consists of multiple carrying capacity estimates for impact subcategories, representing the individual toxic chemicals. Similarly, eco-toxicity is also considered as a non-homogeneous impact category because toxic chemicals affect different sub-ecosystems.

This chapter presents an approach for carrying capacity estimation for human toxicity impact. This approach uses CalTOX, a multimedia fate and transport model, to calculate the long-term partitioning factors of toxic chemicals in air, surface water, and surface soil. The carrying capacity estimate for toxic chemicals by using the CalTOX results coupled with the threshold levels is elaborated. Lastly, an empirical approach for estimating the partitioning factors for carrying capacity estimate is also provided.

10.2 Approach

The carrying capacity for toxic chemicals can be estimated using the threshold-oriented technique discussed in Section 4.2, Chapter 4. The development of the methodology to estimate the carrying capacity in this chapter is an original contribution of this research.

A general step for evaluating the carrying capacity for human toxicity impact using the threshold-oriented technique is as follows:

1. Selection of suitable threshold levels for toxic chemicals.
2. Selection of an appropriate environmental model.
3. Simulation of emission-concentration relationship (source-receptor relationship) using the environmental model. The emission represents release of toxic chemicals through air and surface water. The concentration is that of the toxic chemicals in air, surface water, and surface soil. Environmental concentrations of toxic chemicals are determined by physical-chemical and receiving media properties that govern persistence and pollutant fate.
4. Determination of the partitioning factors. A partitioning factor is the factor that expresses the concentration of a toxic chemical in an environmental medium due to a unit of continuous air or surface-water emission of that toxic chemical. A partitioning factor can be determined based on the assumption that the concentration in the receiving environmental medium is a linear function of the quantity released to the environment.
5. Determination of the national scale carrying capacity for toxic chemicals from the partitioning factors and the threshold levels.

The algorithm for the carrying capacity estimation for human toxicity impact using environmental transport modeling following the threshold-oriented technique is presented in Figure 10.1.

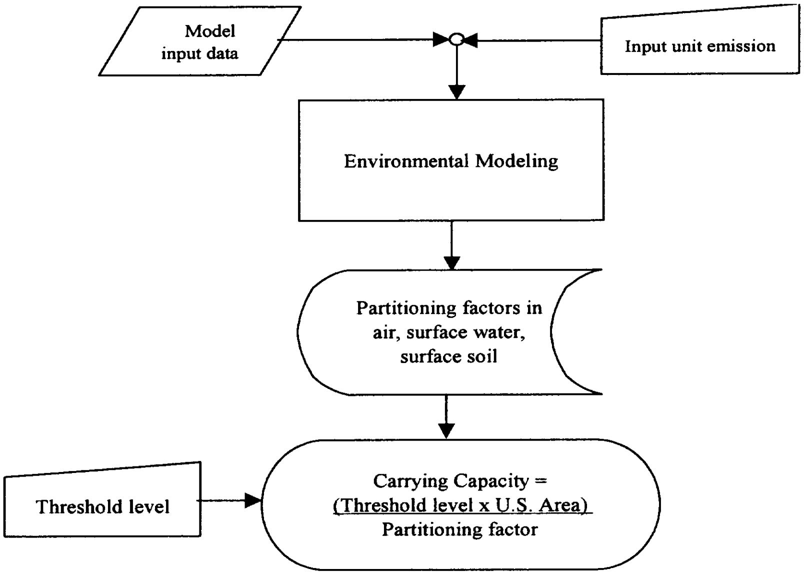

Figure 10.1 Algorithm for carrying capacity estimate using threshold-oriented technique for human toxicity impact.

10.3 Threshold Levels for Carrying Capacity Estimate

Several regulatory levels and associated databases are used as surrogates for the target threshold levels. The IRIS (EPA 2001) database was used extensively in this research. The IRIS provides inhalation reference concentrations (RfCs) and oral reference doses (RfDs) for chronic non-carcinogenic health effects and oral slope factors (Cancer Potency Factor, CPF) for carcinogenic health effects. The chronic RfCs and RfDs are generally used as the target threshold levels. Where available, the carcinogen risk level of 10^{-6} is used to determine the target threshold level[1]. This risk level is comparable to the criteria used by EPA that lead to an acceptable risk range of 10^{-4} to 10^{-6} (EPA 1989b). The IRIS database is updated on a monthly basis. Both RfC and RfD are equivalent to Acceptable Daily Intake (ADI), which is proposed by WHO and used by other environmental agencies.

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[1] Follows the risk-based preliminary remediation goals for Superfund (EPA 1991)

The other toxicity database that provides the target threshold levels is the chronic Minimum Risk Levels (MRLs) used by the ATSDR[1] (ATSDR 2002). For regulations and standards, the Drinking Water Standards and Health Advisories (EPA 2002b) and the National Ambient Air Quality Standards (NAAQS) (EPA 1990) are the major sources of reference. The Drinking Water Standards and Health Advisories provide the enforceable standard maximum contaminant levels (MCLs) for some common water pollutants and priority toxic pollutants that can be used as the target threshold levels. The target threshold levels for six criteria air pollutants are adopted from NAAQS.

It should be mentioned that using the regulatory levels and associated databases as the threshold level will give the results a conservative bias. This is because the regulatory levels and associated databases are defined as acceptable doses for safety and they take into consideration a variety of uncertainties. The ADIs, RfDs, RfCs, MRLs, MCLs, and CPFs, are non-toxicological parameters because they are adopted from experimental toxicological information, such as the no-observed-adverse effect level (NOAEL) and the lowest-observed-adverse effect level (LOAEL), by applying extrapolating factors (10X factors). When lumped together, the extrapolating factors can be as high as four orders of magnitude. As a result, the regulatory levels and associated databases are potentially much lower than the actual toxicological threshold. However, the regulatory levels and associated databases are still being utilized at this stage because they are widely perceived as levels that will not be harmful to human health. Furthermore, the regulatory levels and toxicological benchmarks are more accessible than the NOAEL and LOAEL.

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[1] Agency for Toxic Substances and Disease Registry

Typically, reference concentrations for air are expressed in terms of air concentration (mg/m^3 or ppm) and no further conversion is needed for adopting them as the target threshold levels. However, further calculations that take into account the exposure factors may be needed in order to adopt the ingestion reference dose (RfD) in terms of mg/kg/d as the target threshold levels for surface water and surface soil. In such cases, the equations to convert the reference dose to water and soil concentrations are adopted from EPA (1991):

Ingestion of surface water: carcinogens

$$C_w\,(mg/l) = \frac{Risk \times BW \times AT \times 365\ days/yr}{SF_o \times IR_w \times EF \times ED} \qquad (10.1)$$

Ingestion of surface water: non-carcinogens

$$C_w\,(mg/l) = \frac{HQ \times RfD_o \times BW \times AT \times 365\ days/yr}{IR_w \times EF \times ED} \qquad (10.2)$$

Where C_w = chemical concentration in water (mg/l), Risk = 10^{-6}, BW = average adult body weight (70 kg), AT = averaging time (70 years for carcinogens and 30 years for non-carcinogens), SF_o = chemical-specific oral cancer slope factor (mg/kg-day)$^{-1}$, IR_w = daily water consumption rate (2 liter/day), EF = exposure frequency (350 days/yr), ED = exposure duration (30 yr), HQ = hazard quotient (1.0), and RfD_o = chemical-specific oral chronic reference dose (mg/kg-day).

Ingestion of surface soil: carcinogens

$$C_s\,(mg/kg) = \frac{Risk \times AT \times 365\ days/yr}{SF_o \times 10^{-6}\ kg/mg \times EF \times IF_{soil/adj}} \qquad (10.3)$$

Ingestion of surface soil: non-carcinogens

$$C_s\,(mg/kg) = \frac{HQ \times RfD_o \times AT \times 365\ days/yr}{10^{-6}\ kg/mg \times EF \times IF_{soil/adj}} \qquad (10.4)$$

Where C_s = chemical concentration in soil (mg/kg), Risk = 10^{-6}, AT = averaging time (70 years for carcinogens and 30 years for non-carcinogens), SF_o = chemical-specific oral cancer slope factor $(mg/kg\text{-}day)^{-1}$, EF = exposure frequency (350 days/yr), $IF_{soil/adj}$ = age-adjusted soil ingestion factor (114 mg-yr/kg-day), HQ = hazard quotient (1.0), and RfD_o = chemical-specific oral chronic reference dose (mg/kg-day). Table 10.1 presents the target threshold levels for 78 organic chemicals, 10 metals, and particulate matter.

Table 10.1 Summary of the Threshold Levels for 89 Toxic Chemicals

Substance	CAS	Reference value						Threshold level		
		Inhalation			Ingestion			Air	Surface water	Surface soil
		Value	Unit	Note	Value	Unit	Note	mg/m^3	mg/l	mg/kg
Acenaphthene	83-32-9	6.0E-02	mg/kg/d	RfD	6.0E-02	mg/kg/d	RfD	2.9E-01	5.1E+00	3.8E+04
Acetone	67-64-1	1.0E-01	mg/kg/d	RfD	1.0E-01	mg/kg/d	RfD	4.9E-01	8.5E+00	6.4E+04
Aldrin	309-00-2	3.0E-05	mg/kg/d	RfD	3.0E-05	mg/kg/d	RfD	1.5E-04	2.6E-03	1.9E+01
Anthracene	120-12-7	3.0E-01	mg/kg/d	RfD	3.0E-01	mg/kg/d	RfD	1.5E+00	2.6E+01	1.9E+05
Benzene	71-43-2	1.0E-01	kg-d/mg	CPF	1.0E-01	kg-d/mg	CPF	4.9E+01	8.5E-04	6.4E+00
Benzo(a)anthracene	56-55-3	3.9E-01	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	1.2E+01	7.1E-05	5.3E-01
Benzo(a)pyrene	50-32-8	3.9E-01	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	1.2E+01	7.1E-05	5.3E-01
Benzo(b)fluoranthene	205-99-2	3.9E-01	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	1.2E+01	7.1E-05	5.3E-01
Bis(2-chloroethyl)ether	111-44-4	1.2E-01	mg/m3	MRL				1.2E-01		
Bis(2-ethylhexyl)phthalate	117-81-7	8.4E-03	kg-d/mg	CPF	8.4E-03	kg-d/mg	CPF	5.8E+02	1.0E-02	7.6E+01
Bromodichloromethane	75-27-4	2.0E-02	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	9.7E-02	1.7E+00	1.3E+04
Bromoform	75-25-2	2.0E-02	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	9.7E-02	1.7E+00	1.3E+04
Butanol	71-36-3	1.0E-01	mg/kg/d	RfD	1.0E-01	mg/kg/d	RfD	4.9E-01	8.5E+00	6.4E+04
Butyl benzyl phthalate	85-68-7	2.0E-01	mg/kg/d	RfD	2.0E-01	mg/kg/d	RfD	9.7E-01	1.7E+01	1.3E+05
Carbon disulfide	75-15-0	7.0E-01	mg/m3	RfC	1.0E-01	mg/kg/d	RfD	7.0E-01	8.5E+00	6.4E+04
Carbon tetrachloride	56-23-5	7.0E-04	mg/kg/d	RfD	7.0E-04	mg/kg/d	RfD	3.4E-03	6.0E-02	4.5E+02
Chlordane	57-74-9	7.0E-04	mg/m3	RfC	5.0E-04	mg/kg/d	RfD	7.0E-04	4.3E-02	3.2E+02
Chlorobenzene	108-90-7	6.0E-03	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	2.9E-02	1.7E+00	1.3E+04
Chlorodibromomethane	124-48-1	2.0E-02	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	9.7E-02	1.7E+00	1.3E+04
Chloroform	67-66-3	1.9E-02	kg-d/mg	CPF	3.1E-02	kg-d/mg	CPF	2.6E+02	2.7E-03	2.1E+01
Chrysene	218-01-9	3.9E-02	kg-d/mg	CPF	1.2E-01	kg-d/mg	CPF	1.2E+02	7.1E-04	5.3E+00
DDD	72-54-8	2.4E-01	kg-d/mg	CPF	2.4E-01	kg-d/mg	CPF	2.0E+01	3.5E-04	2.7E+00
DDE	72-55-9	3.4E-01	kg-d/mg	CPF	3.4E-01	kg-d/mg	CPF	1.4E+01	2.5E-04	1.9E+00
DDT	50-29-3	3.4E-01	kg-d/mg	CPF	3.4E-01	kg-d/mg	CPF	1.4E+01	2.5E-04	1.9E+00
Di-n-butyl phthalate	84-74-2	1.0E-04	mg/kg/d	RfD	1.0E-04	mg/kg/d	RfD	4.9E-04	8.5E-03	6.4E+01
Di-n-octyl phthalate	117-84-0	2.0E-02	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	9.7E-02	1.7E+00	1.3E+04
Dibenz(a,h)anthracene	53-70-3	6.0E-02	mg/kg/d	RfD	9.0E-02	mg/kg/d	RfD	2.9E-01	7.7E+00	5.8E+04
1,2-Dichlorobenzene (o)	95-50-1	6.0E-02	mg/kg/d	RfD	9.0E-02	mg/kg/d	RfD	2.9E-01	7.7E+00	5.8E+04
1,4-Dichlorobenzene (p)	106-46-7	4.0E-02	kg-d/mg	CPF	4.0E-02	kg-d/mg	CPF	1.2E+02	2.1E-03	1.6E+01
3,3-Dichlorobenzidine	91-94-1	1.2E+00	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	4.1E+00	7.1E-05	5.3E-01
1,1-Dichloroethane	75-34-3	5.7E-03	kg-d/mg	CPF	5.7E-03	kg-d/mg	CPF	8.5E+02	1.5E-02	1.1E+02
1,2-Dichloroethane	107-06-2	7.0E-02	kg-d/mg	CPF	7.0E-02	kg-d/mg	CPF	7.0E+01	1.2E-03	9.1E+00
1,1-Dichloroethylene	75-35-4	9.0E-03	mg/kg/d	RfD	9.0E-03	mg/kg/d	RfD	4.4E-02	7.7E-01	5.8E+03
cis-1,2-Dichloroethylene	540-59-0				3.0E-01	mg/kg/d	MRL		2.6E+01	1.9E+05
trans-1,2-Dichloroethylene	156-60-5	2.0E-02	mg/kg/d	RfD	2.0E-02	mg/kg/d	RfD	9.7E-02	1.7E+00	1.3E+04
1,2-Dichloropropane	78-87-5	6.3E-02	kg-d/mg	CPF	6.3E-02	kg-d/mg	CPF	7.7E+01	1.4E-03	1.0E+01

Table 10.1 Summary of the Threshold Levels for 89 Toxic Chemicals (Continued)

Substance	CAS	Reference value						Threshold level		
		Inhalation			Ingestion			Air	Surface water	Surface soil
		Value	Unit	Note	Value	Unit	Note	mg/m^3	mg/l	mg/kg
1,3-Dichloropropene	542-75-6	2.0E-02	mg/m3	RfC	3.0E-02	mg/kg/d	RfD	2.0E-02	2.6E+00	1.9E+04
Dieldrin	60-57-1	5.0E-05	mg/kg/d	RfD	5.0E-05	mg/kg/d	RfD	2.4E-04	4.3E-03	3.2E+01
Diethyl phthalate	84-66-2	8.0E-01	mg/kg/d	RfD	8.0E-01	mg/kg/d	RfD	3.9E+00	6.8E+01	5.1E+05
Dimethyl phthalate	131-11-3	1.0E+00	mg/kg/d	RfD	1.0E+00	mg/kg/d	RfD	4.9E+00	8.5E+01	6.4E+05
2,4-Dinitrotoluene	121-14-2	2.0E-03	mg/kg/d	RfD	2.0E-03	mg/kg/d	RfD	9.7E-03	1.7E-01	1.3E+03
2,6-Dinitrotoluene	606-20-2	1.0E-03	mg/kg/d	RfD	1.0E-03	mg/kg/d	RfD	4.9E-03	8.5E-02	6.4E+02
Endosulfan	115-29-7	6.0E-03	mg/kg/d	RfD	6.0E-03	mg/kg/d	RfD	2.9E-02	5.1E-01	3.8E+03
Endrin	72-20-8	3.0E-04	mg/kg/d	RfD	3.0E-04	mg/kg/d	RfD	1.5E-03	2.6E-02	1.9E+02
Ethylbenzene	100-41-4	1.0E+00	mg/m3	RfC	1.0E-01	mg/kg/d	RfD	1.0E+00	8.5E+00	6.4E+04
Fluoranthene	206-44-0	4.0E-02	mg/kg/d	RfD	4.0E-02	mg/kg/d	RfD	1.9E-01	3.4E+00	2.6E+04
Fluorene	86-73-7	4.0E-02	mg/kg/d	RfD	4.0E-02	mg/kg/d	RfD	1.9E-01	3.4E+00	2.6E+04
alpha-HCH (alpha-BHC)	319-84-6	6.3E+00	kg-d/mg	CPF	6.3E+00	kg-d/mg	CPF	7.7E-01	1.4E-05	1.0E-01
beta-HCH (beta-BHC)	319-85-7	1.9E+00	kg-d/mg	CPF	1.9E+00	kg-d/mg	CPF	2.6E+00	4.5E-05	3.4E-01
gamma-HCH (lindane)	58-89-9	3.0E-04	mg/kg/d	RfD	3.0E-04	mg/kg/d	RfD	1.5E-03	2.6E-02	1.9E+02
Heptachlor	76-44-8	5.0E-04	mg/kg/d	RfD	5.0E-04	mg/kg/d	RfD	2.4E-03	4.3E-02	3.2E+02
Heptachlor epoxide	1024-57-3	1.3E-05	mg/kg/d	RfD	1.3E-05	mg/kg/d	RfD	6.3E-05	1.1E-03	8.3E+00
Hexachloro-1,3-butadiene	87-68-3	7.7E-02	kg-d/mg	CPF	7.7E-02	kg-d/mg	CPF	6.3E+01	1.1E-03	8.3E+00
Hexachlorobenzene	118-74-1	8.0E-04	mg/kg/d	RfD	8.0E-04	mg/kg/d	RfD	3.9E-03	6.8E-02	5.1E+02
Hexachlorocyclopentadiene	77-47-4	2.0E-04	mg/m3	RfC	6.0E-03	mg/kg/d	RfD	2.0E-04	5.1E-01	3.8E+03
Hexachloroethane	67-72-1	1.0E-03	mg/kg/d	RfD	1.0E-03	mg/kg/d	RfD	4.9E-03	8.5E-02	6.4E+02
Indeno(1,2,3-c,d)pyrene	193-39-5	3.9E-01	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	1.2E+01	7.1E-05	5.3E-01
Isophorone	78-59-1	2.0E-01	mg/kg/d	RfD	2.0E-01	mg/kg/d	RfD	9.7E-01	1.7E+01	1.3E+05
Methoxychlor	72-43-5	5.0E-03	mg/kg/d	RfD	5.0E-03	mg/kg/d	RfD	2.4E-02	4.3E-01	3.2E+03
Methyl bromide	74-83-9	5.0E-03	mg/m3	RfC	1.0E-03	mg/kg/d	RfD	5.0E-03	8.5E-02	6.4E+02
Methyl chloride	74-87-3	9.0E-02	mg/kg/d	RfD	9.0E-02	mg/kg/d	RfD	4.4E-01	7.7E+00	5.8E+04
Methylene chloride	75-09-2	3.0E-01	ppm	MRL	6.0E-02	mg/kg/d	MRL	3.0E-01	5.1E+00	3.8E+04
Naphthalene	91-20-3	3.0E-03	mg/m3	RfC	2.0E-02	mg/kg/d	RfD	3.0E-03	1.7E+00	1.3E+04
Nitrobenzene	98-95-3	6.0E-04	mg/kg/d	RfD	5.0E-04	mg/kg/d	RfD	2.9E-03	4.3E-02	3.2E+02
PCB	608-93-5	2.0E-04	mg/kg/d	RfD	8.0E-04	mg/kg/d	RfD	9.7E-04	6.8E-02	5.1E+02
Pyrene	129-00-0	3.0E-02	mg/kg/d	RfD	3.0E-02	mg/kg/d	RfD	1.5E-01	2.6E+00	1.9E+04
Styrene	100-42-5	1.0E+00	mg/m3	RfC	2.0E-01	mg/kg/d	RfD	1.0E+00	1.7E+01	1.3E+05
2,3,7,8-TCDD	1746-01-6	1.0E-09	mg/kg/d	MRL	1.0E-09	mg/kg/d	MRL	4.9E-09	8.5E-08	6.4E-04
1,1,2,2-Tetrachloroethane	79-34-5	2.7E-01	kg-d/mg	CPF	2.7E-01	kg-d/mg	CPF	1.8E+01	3.2E-04	2.4E+00
Tetrachloroethylene	127-18-4	1.0E-02	mg/kg/d	RfD	1.0E-02	mg/kg/d	RfD	4.9E-02	8.5E-01	6.4E+03
Toluene	108-88-3	4.0E-01	mg/m3	RfC	2.0E-01	mg/kg/d	RfD	4.0E-01	1.7E+01	1.3E+05
Toxaphene	8001-35-2	1.2E+00	kg-d/mg	CPF	1.2E+00	kg-d/mg	CPF	4.1E+00	7.1E-05	5.3E-01
1,2,4-Trichlorobenzene	120-82-1	2.0E-01	mg/kg/d	RfD	2.0E-01	mg/kg/d	RfD	9.7E-01	1.7E+01	1.3E+05
1,1,1-Trichloroethane	71-55-6	3.0E+00	mg/kg/d	RfD	4.0E-02	mg/kg/d	MRL	1.5E+01	3.4E+00	2.6E+04
1,1,2-Trichloroethane	79-00-5	4.0E-03	mg/kg/d	RfD	4.0E-03	mg/kg/d	RfD	1.9E-02	3.4E-01	2.6E+03
Trichloroethylene	79-01-4	1.0E-01	ppm	MRL	2.0E-01	mg/kg/d	RfD	1.0E-01	1.7E+01	1.3E+05
Vinyl chloride	75-01-4	1.0E-01	mg/m3	RfC	3.0E-03	mg/kg/d	RfD	1.0E-01	2.6E-01	1.9E+03
Xylenes (total)	1330-20-7	2.0E+00	mg/kg/d	RfD	2.0E+00	mg/kg/d	RfD	9.7E+00	1.7E+02	1.3E+06
Antimony	7440-36-0				6.0E-03	mg/l	MCL		6.0E-03	
Arsenic	7440-38-2				5.0E-02	mg/l	MCL		5.0E-02	
Barium	7440-39-3				2.0E+00	mg/l	MCL		2.0E+00	
Cadmium	7440-4-39				5.0E-03	mg/l	MCL		5.0E-03	
Chromium (Hexavalent)	18540-24-9				1.0E-01	mg/l	MCL		1.0E-01	
Copper	7440-50-8				1.3E+00	mg/l	MCL		1.3E+00	
Lead	7439-92-1	1.5E-03	mg/m3	NAAQS	1.5E-02	mg/l	MCL	1.5E-03	1.5E-02	
Manganese	7439-96-5	5.0E-05	mg/m3	RfC	1.4E-01	mg/kg/d	RfD	5.0E-05	1.2E+01	9.0E+04
Nickel	7440-02-0	9.0E-05	mg/m3	MRL	2.0E-02	mg/kg/d	RfD	9.0E-05	1.7E+00	1.3E+04
Zinc	7440-66-6				3.0E-01	mg/kg/d	RfD		2.6E+01	1.9E+05
Particulate matter		5.0E-02	mg/m3	NAAQS				5.0E-02		

10.4 Environmental Transport Modeling

10.4.1 Environmental Transport Model

As previously discussed in Chapter 9, an important step in the threshold-oriented technique is the selection of an appropriate environmental fate and transport model. The general criteria for selecting an appropriate model for carrying capacity estimates have also been discussed. The approach for carrying capacity estimate developed in this chapter requires environmental model(s) that can simulate the migrations and concentrations of pollutants between target media (air, surface water, and surface soil).

Basically, well-developed single medium models (e.g., an atmospheric model), which are accurate and allow great spatial resolution, may be used. However, a long-term multimedia model that can simultaneously partition the chemical among multiple media is advantageous (Coulibaly 2000; Bennett et al. 2001). Multimedia environmental models, specifically the fugacity or Mackay-type approach (Mackay 1991; Mackay et al. 1992), have become increasingly popular in the assessment of the environmental fate of toxic chemicals (e.g., risk assessment) over the last several years (Hertwich 2001). The fugacity approach is a simplified yet generic method for quantifying the fate of ecological effects of pollutant releases by assuming a hypothetical box containing air, soil, water, sediment, and biota compartments.

The fugacity approach, or the unit world approach, is a hypothetical box containing air, soil, water, sediment, and biota compartments. The unit world is represented as a set of dynamic equations that describe the partitioning and transformation of a chemical introduced into the box. A relatively small number of chemical-specific parameters are sufficient to predict the partitioning of the chemical between the compartments of the box. If the rate of transformation due to photolysis, oxidation, biodegradation, or other processes can be estimated, then the approach can be used to predict steady-state concentrations. The unit world approach predicts the partitioning of the chemical based on mass balance, diffusion, and residence time.

Currently, integrated multimedia models using the unit world approach are used to set clean-up standards, to assess the relative importance of chemical emissions, to evaluate the partitioning, persistence, and long-range transport of organic pollutants, and to set priorities on pollution prevention (Hertweich 2001). From the LCA perspective, integrated multimedia models have been used to calculate the relative importance of chemical emissions, i.e., the human toxicity potential (HTP) among chemicals (Hertwich et al. 2001; Huijbregts et al. 2000a; 2000b; 2001b; Jolliet et al. 2003). However, the use of the multimedia model in this chapter is to assess the steady-state partitioning of toxic chemicals in three types of environmental media: air, surface water, and surface soil.

There are a number of multimedia models used to describe the relationship between an emission of a pollutant and its concentrations in environmental media including air, surface water, groundwater, sediment, and biota. For example, the models that are developed using U.S. conditions are the Multimedia Environmental Pollutant Assessment System (MEPAS-Streile et al. 1996) and the CalTOX (McKone 1993). The Canadian Environmental Modeling Centre has developed a series of multimedia and other supporting models (CEMC 2003).

The European Union System for the Evaluation of Substances (EUSES) was used to compare the relative toxicity of chemicals in Europe (Vermeire et al. 1994; RIVM 1996). A recently revised model, USES-LCA, was used to determine a set of HTP for LCA purposes (Huijbregts et al. 2000a; 2000b; 2001b). The Simple Box Model determines toxic chemical concentrations in only air, surface soil, surface water, and sediment (Van de Meent 1993; Brandes et al. 1996).

Factors that should be considered in selecting an appropriate environmental model are: the model should rely on simple but applicable models yet should be easily comprehensible to users, the level of detail and accuracy needed for the analysis, the technical validity of the simulation modeling, the resources available, the detail and accuracy of the data used in the modeling. The model should be a generic one that can be applied on a larger scale and appropriate data should be available before a model is chosen. Based on these factors, the CalTOX is selected as the multimedia model for this research because it is simple and transparent. It is also simple to perform in terms of data editing and data acquisition, some input data are already available and little adjustment is required. Moreover, it can be used to simulate the partitioning of particles in multiple media, and it can be used for long-term simulation, which suits the LCA context.

10.4.2 Description of Model and Database Used in Carrying Capacity Estimate

CalTOX is an integrated multimedia environmental model that is based on the fugacity or Mackay-type approach that was originally developed for health risk assessment purposes by the Office of Scientific Affairs of the California Department of Toxic Substances Control (McKone 1993; California DEP 1994). This model employs the unit box fugacity approach and translates the results into contaminant concentrations in the environmental media and evaluates the risks associated with exposure to these concentrations.

CalTOX is a stand-alone spreadsheet model that contains three components: a multimedia transport and transformation model, exposure scenario models, and efforts to quantify and reduce uncertainty in multimedia, multiple-pathway exposure models. CalTOX has been modified from the conventional fugacity approach so that several chemical classes can be addressed: organic chemicals, metals, inorganic chemicals, and radionuclides. The inputs for CalTOX are physical and chemical properties of chemicals, landscape properties, exposure factors, and diffusion and advection rate constants. Equations in CalTOX are derived based on the law of conservation of mass and chemical equilibrium. After partitioning the chemical to soil, air, water, sediment and biota compartments, CalTOX then determines the chemical concentration in these compartments and estimates exposure and risk (Maddelena et al. 1995). The limitations of CalTOX are: it does not allow spatial tracking of a pollutant (no vertical or horizontal dimensions to environmental compartments), the chemical classes are not addressed for surfactants or volatile metals, there are limited environmental settings, and it applies to steady state or long time run only. CalTOX 2.3 is the most recent version available (McKone et al. 1997) and a new version will be released soon (LBNL 2006). Some default inputs are readily available for landscape properties, exposure factors, and diffusion and advection rate constants. The default chemical and physical properties are available for 78 organic chemicals (California EPA 2003). The default landscape and meteorological parameters used in this research are those of U.S. conditions. CalTOX has been used for multimedia exposure modeling for the screening level human risk assessment (California EPA 2003; Bennett et al. 1998; Chang et al., 2004; Glorennec et al. 2005; Chen and Ma 2006). Some studies also used CalTOX to develop LCIA methods, e.g., to determine the human toxicity potentials (Hertwich et al. 2000; Hertwich et al. 2001; Bennett et al. 2002).

10.5 Carrying Capacity Estimate

10.5.1 Emission-Concentration Relationship

The emission-concentration relationship (source-receptor) can be obtained from CalTOX simulations by assuming a linear function between the quantity released to the environment and the chemical concentrations in receiving media. Ambient concentrations of toxic chemicals are determined by physical-chemical and receiving media properties that govern persistence and pollutant fate. The linear function of the emission-concentration relationship implies that the concentration of toxic chemical in a medium is directly proportional to the amount released. As illustrated in Figure 10.2, the linear function of the emission-concentration relationship can be observed at a high emission rate (which is also an emission range for carrying capacity estimate).

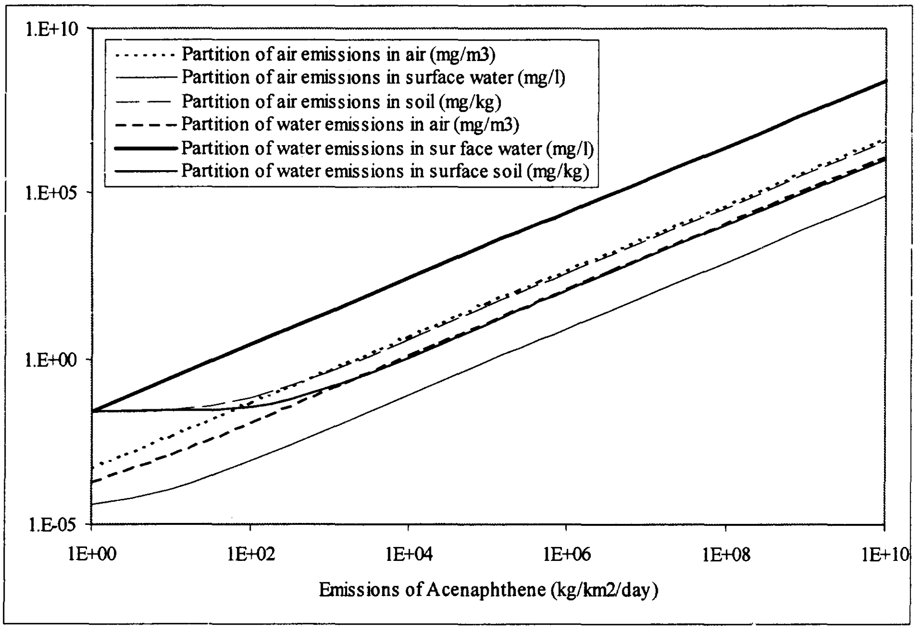

Figure 10.2 Emission-concentration relationship for acenaphthene resulting from CalTOX simulation.

The emission-concentration relationship is expressed in terms of a partitioning factor, i.e., the concentration of a chemical in a medium resulting from a unit emission of a chemical ($kg/km^2/yr$). As an example, Table 10.2 presents the units of partitioning factors for air, surface water, and surface soil resulting from soil and water emissions. Because CalTOX is not based on steady-state simulation, ambient concentrations at one year after the accumulation of chemical releases are used to determine the partitioning factors. By using this one year time frame, the partitioning factors are less than those for a longer period or at steady-state. As a consequence, the carrying capacity values may be overestimated. Attention should be given to this issue in the future.

Table 10.2 Units for Partitioning Factors

Partitioning factor for	Media	Unit
Air emissions	Air	$(mg/m^3)/(kg/km^2/day)$
	Surface water	$(mg/l)/(kg/km^2/day)$
	Surface soil	$(mg/kg)/(kg/km^2/day)$
Water emissions	Air	$(mg/m^3)/(kg/km^2/day)$
	Surface water	$(mg/l)/(kg/km^2/day)$
	Surface soil	$(mg/kg)/(kg/km^2/day)$

And by using CalTOX, partitioning factors for 78 organic chemicals are determined for air, surface water, and surface soil resulting from soil and water emissions (Table 10.3). The partitioning of toxic chemicals resulting from soil release is less relevant in environmental media. Therefore, the partitioning factor resulting from soil releases is not determined. This limitation is considered not significant since life cycle inventory seldom reports on soil release.

Table 10.3 Partitioning Factors for Air and Water Emissions of 78 Organic Chemicals

Substance	Partition of air emissions (Partitioning factor)			Partition of water emissions (Partitioning factor)		
	In air	In surface water	In surface soil	In air	In surface water	In surface soil
	(mg/m^3) (kg/km^2/d)	(mg/l) (kg/km^2/d)	(mg/kg) (kg/km^2/d)	(mg/m^3) (kg/km^2/d)	(mg/l) (kg/km^2/d)	(mg/kg) (kg/km^2/d)
Acenaphthene	4.8E-04	8.5E-06	3.9E-04	1.3E-04	2.7E-02	1.1E-04
Acetone	5.2E-04	5.2E-05	2.6E-05	7.8E-05	2.1E-02	3.8E-06
Aldrin	4.5E-04	7.7E-04	1.1E+00	3.7E-04	1.3E-01	9.2E-01
Anthracene	3.7E-04	4.2E-07	2.2E-09	6.0E-07	2.9E-04	3.4E-05
Benzene	5.2E-04	9.8E-07	5.7E-07	2.2E-04	3.8E-02	2.4E-07
Benzo(a)anthracene	3.6E-04	4.4E-06	7.4E+00	1.5E-07	5.1E-04	3.0E-03
Benzo(a)pyrene	2.5E-04	1.2E-04	6.7E+00	9.9E-08	1.4E-02	2.6E-03
Benzo(b)fluoranthene	4.0E-04	1.0E-03	2.0E+01	2.4E-05	7.6E-02	1.2E+00
Bis(2-chloroethyl)ether	5.2E-04	4.5E-04	2.0E-04	3.4E-04	1.5E-01	1.3E-04
Bis(2-ethylhexyl)phthalate	4.6E-04	2.3E-04	9.8E-02	3.9E-05	7.3E-02	8.2E-03
Bromodichloromethane	5.2E-04	5.2E-06	1.6E-06	3.7E-04	6.4E-02	1.1E-06
Bromoform	5.2E-04	2.7E-05	1.2E-05	4.9E-04	9.1E-02	1.1E-05
Butanol	5.2E-04	1.4E-04	2.0E-04	2.8E-05	2.3E-02	1.1E-05
Butyl benzyl phthalate	4.6E-04	8.4E-05	3.3E-02	3.6E-06	2.4E-02	2.6E-04
Carbon disulfide	5.2E-04	9.4E-07	2.8E-07	4.8E-04	8.1E-02	2.6E-07
Carbon tetrachloride	5.2E-04	4.4E-07	3.6E-07	4.7E-04	8.0E-02	3.2E-07
Chlordane	5.1E-04	3.3E-04	3.8E-01	4.4E-04	1.3E-01	3.3E-01
Chlorobenzene	5.2E-04	4.1E-06	3.3E-06	4.4E-04	7.5E-02	2.8E-06
Chlorodibromomethane	5.2E-04	1.1E-05	3.8E-06	4.3E-04	7.6E-02	3.1E-06
Chloroform	5.2E-04	2.6E-06	7.7E-07	4.4E-04	7.4E-02	6.4E-07
Chrysene	3.3E-04	3.1E-05	9.5E+00	1.7E-07	2.2E-03	4.7E-03
DDD	4.1E-04	1.5E-02	4.0E+01	6.6E-05	5.0E-01	6.5E+00
DDE	4.8E-04	2.2E-04	3.7E+00	4.0E-05	1.9E-02	3.0E-01
DDT	5.1E-04	5.3E-04	1.9E+00	3.3E-04	1.6E-01	1.2E+00
Di-n-butyl phthalate	4.8E-04	1.8E-03	2.8E-01	7.9E-06	4.4E-02	4.6E-03
Di-n-octyl phthalate	3.6E-04	2.6E-04	4.4E-01	1.1E-08	8.8E-02	1.3E-05
Dibenz(a,h)anthracene	3.0E-04	1.7E-03	2.4E+01	3.1E-08	8.5E-02	2.5E-03
1,2-Dichlorobenzene (o)	5.2E-04	5.4E-06	7.2E-06	4.3E-04	7.5E-02	5.9E-06
1,4-Dichlorobenzene (p)	5.2E-04	2.2E-06	8.1E-06	2.3E-04	3.9E-02	3.5E-06
3,3-Dichlorobenzidine	4.5E-05	5.1E-07	3.1E-01	3.6E-12	1.3E-05	2.5E-08
1,1-Dichloroethane	5.2E-04	2.0E-06	6.2E-07	4.3E-04	7.3E-02	5.0E-07
1,2-Dichloroethane	5.2E-04	1.0E-05	1.4E-06	4.8E-04	8.4E-02	1.3E-06
1,1-Dichloroethylene	5.1E-04	5.3E-07	1.2E-07	4.7E-04	8.1E-02	1.1E-07
cis-1,2-Dichloroethylene	5.2E-04	2.7E-06	5.1E-07	4.3E-04	7.4E-02	4.2E-07
trans-1,2-Dichloroethylene	5.2E-04	1.2E-06	3.7E-07	4.4E-04	7.4E-02	3.1E-07
1,2-Dichloropropane	5.2E-04	4.3E-06	9.7E-07	4.8E-04	8.3E-02	8.9E-07
1,3-Dichloropropene	5.2E-04	2.0E-06	8.3E-07	1.9E-04	3.2E-02	2.9E-07
Dieldrin	4.7E-04	2.0E-02	2.2E+00	2.4E-04	4.8E-01	1.2E+00
Diethyl phthalate	5.0E-04	4.6E-03	1.2E-02	1.1E-05	1.6E-01	2.8E-04
Dimethyl phthalate	5.2E-04	1.9E-04	2.0E-03	5.8E-06	2.4E-02	2.2E-05
2,4-Dinitrotoluene	4.6E-04	4.2E-04	3.0E-02	8.7E-08	4.4E-03	5.6E-06
2,6-Dinitrotoluene	3.9E-04	4.9E-04	5.7E-04	1.7E-08	2.4E-03	2.5E-06
Endosulfan	5.0E-04	7.3E-05	8.2E-03	3.5E-05	2.6E-02	5.7E-04
Endrin	4.8E-04	1.9E-02	1.3E+00	2.2E-04	6.9E-01	6.1E-01
Ethylbenzene	5.2E-04	4.8E-07	1.1E-06	1.6E-04	2.7E-02	3.4E-07
Fluoranthene	4.9E-04	7.8E-05	1.0E+00	1.0E-05	1.0E-02	2.1E-02
Fluorene	5.2E-04	9.4E-05	3.1E-03	3.4E-04	8.2E-02	2.0E-03
alpha-HCH (alpha-BHC)	5.1E-04	1.3E-03	2.0E-02	1.3E-04	2.5E-01	5.2E-03
beta-HCH (beta-BHC)	3.9E-04	3.3E-02	1.0E+00	3.4E-06	3.2E-01	9.0E-03
gamma-HCH (lindane)	5.1E-04	3.0E-03	3.6E-02	1.4E-04	3.6E-01	9.9E-03
Heptachlor	4.8E-04	1.1E-05	5.6E-03	7.0E-05	1.6E-02	8.2E-04
Heptachlor epoxide	5.2E-04	1.6E-05	4.8E-03	4.4E-04	9.4E-02	4.1E-03

Table 10.3 Partitioning Factors for Air and Water Emissions of 78 Organic Chemicals (Continued)

Substance	Partition of air emissions (Partitioning factor)			Partition of water emissions (Partitioning factor)		
	In air	In surface water	In surface soil	In air	In surface water	In surface soil
	(mg/m³) (kg/km²/d)	(mg/l) (kg/km²/d)	(mg/kg) (kg/km²/d)	(mg/m³) (kg/km²/d)	(mg/l) (kg/km²/d)	(mg/kg) (kg/km²/d)
Hexachloro-1,3-butadiene	5.2E-04	4.9E-07	4.3E-05	4.3E-04	7.8E-02	3.5E-05
Hexachlorobenzene	5.2E-04	1.4E-05	2.3E-03	4.8E-04	9.5E-02	2.1E-03
Hexachlorocyclopentadiene	4.8E-04	1.1E-07	7.7E-05	8.2E-05	1.8E-02	1.3E-05
Hexachloroethane	5.2E-04	1.3E-06	2.0E-05	4.3E-04	7.5E-02	1.7E-05
Indeno(1,2,3-c,d)pyrene	3.1E-04	5.4E-03	1.7E+01	7.2E-08	3.4E-01	3.9E-03
Isophorone	4.1E-04	3.6E-04	3.1E-04	6.2E-05	8.5E-02	4.7E-05
Methoxychlor	4.6E-04	4.5E-06	5.2E-01	3.6E-07	9.7E-04	4.1E-04
Methyl bromide	5.2E-04	1.1E-06	2.6E-07	2.8E-04	4.7E-02	1.4E-07
Methyl chloride	5.2E-04	8.5E-07	2.1E-07	2.8E-04	4.7E-02	1.1E-07
Methylene chloride	5.2E-04	2.5E-06	6.6E-07	2.8E-04	4.7E-02	3.5E-07
Naphthalene	5.1E-04	1.1E-05	8.0E-05	2.0E-04	3.7E-02	3.1E-05
Nitrobenzene	5.2E-04	4.6E-04	3.5E-04	3.5E-04	1.5E-01	2.3E-04
PCB	5.1E-04	3.2E-04	1.4E+00	2.3E-04	2.3E-01	6.2E-01
Pyrene	3.8E-04	1.4E-06	5.8E-01	3.2E-07	3.5E-04	4 9E-04
Styrene	4.7E-04	2.1E-06	9.0E-06	2.7E-04	5.1E-02	5.1E-06
2,3,7,8-TCDD	4.1E-04	4.8E-03	3.1E+01	3.5E-05	3.8E-01	2.7E+00
1,1,2,2-Tetrachloroethane	5.2E-04	2.9E-05	1.5E-05	2.9E-04	5.6E-02	8.3E-06
Tetrachloroethylene	5.2E-04	5.5E-07	6.0E-07	3.2E-04	5.4E-02	3.7E-07
Toluene	5.2E-04	9.2E-07	9.3E-07	2.4E-04	4.1E-02	4.3E-07
Toxaphene	5.2E-04	2.2E-05	3.4E-05	4.9E-04	8.3E-02	3.2E-05
1,2,4-Trichlorobenzene	5.2E-04	6.5E-06	3.6E-05	4.3E-04	7.5E-02	3.0E-05
1,1,1-Trichloroethane	5.2E-04	7.1E-07	3 6E-07	4.6E-04	7.8E-02	3.2E-07
1,1,2-Trichloroethane	5.2E-04	1.3E-05	4.0E-06	4.7E-04	8.2E-02	3.5E-06
Trichloroethylene	5.2E-04	1.3E-06	5.1E-07	4.4E-04	7.5E-02	4.3E-07
Vinyl chloride	5.2E-04	4.8E-07	1.5E-07	4.9E-04	8.3E-02	1.4E-07
Xylenes (total)	5.1E-04	1.0E-06	1.5E-06	2.7E-04	4.7E-02	8.1E-07
Metals	3.2E-05	1.5E-02	4.0E-03		4.3E-02	4.0E-15
Particulate Matters	3.2E-05	1.5E-02	4.0E-03		4.3E-02	4.0E-15

In general, CalTOX is not recommended for use to determine partitioning of inorganic chemicals in its compartments unless some site-specific data is provided, e.g., soil-water partition coefficients (California DTSC 2004). However, it is assumed in this study that there is no partitioning for water emission in air and the partitioning for water emission in surface soil is very minimal for metals and particles. Therefore, partitioning factors for metals and particles are considered constant regardless of chemical species. This is because metals and particles are conservative and there is no transformation or loss due to chemical reactions. Metals and particles behave the same, in that they distribute evenly over the surface soil and water below when emitted to the air and stay in water when emitted in that medium. As a result, partitioning factors for air emissions of metals and particles are the same at 3.2×10^{-5} (mg/m^3)/(kg/km^2/day), 1.5×10^{-2} (mg/l)/(kg/km^2/day), and 4.0×10^{-3} (mg/kg)/(kg/km^2/day) in air, surface water, and surface soil, respectively. And partitioning factors for water emissions are 4.3×10^{-2} (mg/l)/(kg/m^2/day) and 4.0×10^{-15} (mg/kg)/(kg/m^2/day) in surface water and surface soil, respectively.

10.5.2 Carrying Capacity Estimate

The carrying capacity for human toxicity impact is determined from the emission that results in an environmental concentration reaching the target threshold level. Upon the availability of the threshold level and the partitioning factor, a carrying capacity per unit area can then be calculated (Table 10.4). The national scale (U.S.) carrying capacities are calculated by applying the U.S. land area of 9.4×10^6 km^2. Equations (10.5), (10.6), and (10.7) are used to calculate the U.S. carrying capacity for both air and water emissions. Subscripts air, water, and soil denote the carrying capacity estimate from the threshold levels for air, surface water, and surface soil, respectively.

$$U.S.\ CC_{air}\ (kg/yr) = \frac{\text{Threshold level } (mg/m^3) \times 9.4\times10^6\ km^2 \times 365\ days/yr}{\text{Partitioning factor}_{air}\ (mg/m^3)/(kg/km^2/day)} \tag{10.5}$$

$$U.S.\ CC_{water}\ (kg/yr) = \frac{\text{Threshold level } (mg/l) \times 9.4\times10^6\ km^2 \times 365\ days/yr}{\text{Partitioning factor}_{water}\ (mg/l)/(kg/km^2/day)} \tag{10.6}$$

$$U.S.\ CC_{soil}\ (kg/yr) = \frac{\text{Threshold level } (mg/kg) \times 9.4\times10^6\ km^2 \times 365\ days/yr}{\text{Partitioning factor}_{soil}\ (mg/kg)/(kg/km^2/day)} \tag{10.7}$$

Table 10.4 Summary of Carrying Capacity Estimates for Human Toxicity Impact for 89 Toxic Chemicals

Substance	CAS	U.S. carrying capacity (kg/yr)					
		Air emissions			Water emissions		
		Air	Surface water	Surface soil	Air	Surface water	Surface soil
Acenaphthene	83-32-9	2.09E+12	2.06E+15	3.38E+17	7.71E+12	6.49E+11	1.20E+18
Acetone	67-64-1	3.21E+12	5.62E+14	8.45E+18	2.14E+13	1.39E+12	5.78E+19
Aldrin	309-00-2	1.11E+09	1.14E+10	5.99E+10	1.35E+09	6.74E+07	7.16E+10
Anthracene	120-12-7	1.35E+13	2.09E+17	3.04E+23	8.35E+15	3.02E+14	1.94E+19
Benzene	71-43-2	3.21E+14	2.98E+12	3.85E+16	7.59E+14	7.69E+07	9.15E+16
Benzo(a)anthracene	56-55-3	1.19E+14	5.53E+10	2.47E+08	2.85E+17	4.77E+08	6.10E+11
Benzo(a)pyrene	50-32-8	1.71E+14	2.03E+09	2.73E+08	4.32E+17	1.74E+07	7.04E+11
Benzo(b)fluoranthene	205-99-2	1.07E+14	2.44E+08	9.15E+07	1.78E+15	3.20E+06	1.53E+09
Bis(2-chloroethyl)ether	111-44-4	7.92E+11			1.21E+12		
Bis(2-ethylhexyl)phthalate	117-81-7	4.32E+15	1.51E+11	2.67E+12	5.10E+16	4.77E+08	3.19E+13
Bromodichloromethane	75-27-4	6.42E+11	1.12E+15	2.75E+19	9.03E+11	9.13E+10	3.99E+19
Bromoform	75-25-2	6.42E+11	2.16E+14	3.66E+18	6.82E+11	6.42E+10	3.99E+18
Butanol	71-36-3	3.21E+12	2.09E+14	1.10E+18	5.96E+13	1.27E+12	2.00E+19
Butyl benzyl phthalate	85-68-7	7.26E+12	6.96E+14	1.33E+16	9.28E+14	2.44E+12	1.69E+18
Carbon disulfide	75-15-0	4.62E+12	3.11E+16	7.85E+20	5.00E+12	3.61E+11	8.45E+20
Carbon tetrachloride	56-23-5	2.25E+10	4.65E+14	4.27E+18	2.49E+10	2.56E+09	4.81E+18
Chlordane	57-74-9	4.71E+09	4.43E+11	2.89E+12	5.46E+09	1.12E+09	3.33E+12
Chlorobenzene	108-90-7	1.93E+11	1.43E+15	1.33E+19	2.28E+11	7.79E+10	1.57E+19
Chlorodibromomethane	124-48-1	6.42E+11	5.31E+14	1.16E+19	7.77E+11	7.69E+10	1.42E+19
Chloroform	67-66-3	1.69E+15	3.63E+12	9.20E+16	2.00E+15	1.27E+08	1.11E+17
Chrysene	218-01-9	1.30E+15	7.86E+10	1.93E+09	2.52E+18	1.11E+09	3.90E+12
DDD	72-54-8	1.70E+14	8.12E+07	2.29E+08	1.05E+15	2.44E+06	1.41E+09
DDE	72-55-9	1.02E+14	3.91E+09	1.75E+09	1.23E+15	4.52E+07	2.15E+10
DDT	50-29-3	9.63E+13	1.62E+09	3.40E+09	1.49E+14	5.37E+06	5.38E+09
Di-n-butyl phthalate	84-74-2	3.48E+09	1.62E+10	7.85E+11	2.11E+11	6.64E+08	4.78E+13
Di-n-octyl phthalate	117-84-0	9.28E+11	2.25E+13	9.99E+13	3.04E+16	6.64E+10	3.38E+18
Dibenz(a,h)anthracene	53-70-3	3.34E+12	1.55E+13	8.24E+12	3.23E+16	3.09E+11	7.91E+16
1,2-Dichlorobenzene (o)	95-50-1	1.93E+12	4.87E+15	2.75E+19	2.33E+12	3.51E+11	3.35E+19
1,4-Dichlorobenzene (p)	106-46-7	8.03E+14	3.32E+12	6.78E+15	1.81E+15	1.87E+08	1.57E+16
3,3-Dichlorobenzidine	91-94-1	3.09E+14	4.77E+11	5.91E+09	3.87E+21	1.87E+10	7.32E+16
1,1-Dichloroethane	75-34-3	5.63E+15	2.56E+13	6.22E+17	6.81E+15	7.02E+08	7.71E+17
1,2-Dichloroethane	107-06-2	4.59E+14	4.17E+11	2.24E+16	4.97E+14	4.97E+07	2.41E+16
1,1-Dichloroethylene	75-35-4	2.95E+11	4.96E+15	1.65E+20	3.20E+11	3.25E+10	1.80E+20
cis-1,2-Dichloroethylene	540-59-0		3.25E+16	1.29E+21		1.18E+12	1.57E+21
trans-1,2-Dichloroethylene	156-60-5	6.42E+11	4.87E+15	1.19E+20	7.59E+11	7.90E+10	1.42E+20
1,2-Dichloropropane	78-87-5	5.10E+14	1.08E+12	3.60E+16	5.52E+14	5.59E+07	3.92E+16
1,3-Dichloropropene	542-75-6	1.32E+11	4.38E+15	7.94E+19	3.61E+11	2.74E+11	2.27E+20
Dieldrin	60-57-1	1.78E+09	7.31E+08	4.99E+10	3.48E+09	3.04E+07	9.15E+10
Diethyl phthalate	84-66-2	2.67E+13	5.08E+13	1.46E+17	1.21E+15	1.46E+12	6.28E+18
Dimethyl phthalate	131-11-3	3.21E+13	1.54E+15	1.10E+18	2.88E+15	1.22E+13	9.99E+19
2,4-Dinitrotoluene	121-14-2	7.26E+10	1.39E+12	1.46E+14	3.84E+14	1.33E+11	7.85E+17
2,6-Dinitrotoluene	606-20-2	4.28E+10	5.96E+11	3.85E+15	9.82E+14	1.22E+11	8.79E+17
Endosulfan	115-29-7	2.00E+11	2.40E+13	1.61E+15	2.86E+12	6.74E+10	2.31E+16
Endrin	72-20-8	1.04E+10	4.61E+09	5.07E+11	2.28E+10	1.27E+08	1.08E+12
Ethylbenzene	100-41-4	6.60E+12	6.09E+16	2.00E+20	2.14E+13	1.08E+12	6.46E+20
Fluoranthene	206-44-0	1.36E+12	1.50E+14	8.79E+13	6.68E+13	1.17E+12	4.18E+15
Fluorene	86-73-7	1.28E+12	1.24E+14	2.83E+16	1.96E+12	1.43E+11	4.39E+16
alpha-HCH (alpha-BHC)	319-84-6	5.20E+12	3.57E+07	1.74E+10	2.04E+13	1.86E+05	6.71E+10
beta-HCH (beta-BHC)	319-85-7	2.25E+13	4.66E+06	1.16E+09	2.58E+15	4.81E+05	1.28E+11
gamma-HCH (lindane)	58-89-9	9.82E+09	2.92E+10	1.83E+13	3.58E+10	2.44E+08	6.66E+13
Heptachlor	76-44-8	1.74E+10	1.33E+13	1.96E+14	1.19E+11	9.13E+09	1.34E+15

Table 10.4 Summary of Carrying Capacity Estimates for Human Toxicity Impact for 89 Toxic Chemicals (Continued)

Substance	CAS	U.S. carrying capacity (kg/yr)					
		Air emissions			Water emissions		
		Air	Surface water	Surface soil	Air	Surface water	Surface soil
Heptachlor epoxide	1024-57-3	4.17E+08	2.37E+11	5.95E+12	4.93E+08	4.04E+07	6.97E+12
Hexachloro-1,3-butadiene	87-68-3	4.17E+14	7.74E+12	6.64E+14	5.04E+14	4.87E+07	8.15E+14
Hexachlorobenzene	118-74-1	2.57E+10	1.67E+13	7.64E+14	2.78E+10	2.46E+09	8.37E+14
Hexachlorocyclopentadiene	77-47-4	1.43E+09	1.59E+16	1.71E+17	8.37E+09	9.74E+10	1.01E+18
Hexachloroethane	67-72-1	3.21E+10	2.25E+14	1.10E+17	3.88E+10	3.90E+09	1.29E+17
Indeno(1,2,3-c,d)pyrene	193-39-5	1.38E+14	4.51E+07	1.08E+08	5.95E+17	7.16E+05	4.69E+11
Isophorone	78-59-1	8.15E+12	1.62E+14	1.42E+18	5.39E+13	6.88E+11	9.35E+18
Methoxychlor	72-43-5	1.81E+11	3.25E+14	2.11E+13	2.32E+14	1.51E+12	2.68E+16
Methyl bromide	74-83-9	3.30E+10	2.66E+14	8.45E+18	6.13E+10	6.22E+09	1.57E+19
Methyl chloride	74-87-3	2.89E+12	3.09E+16	9.42E+20	5.37E+12	5.60E+11	1.80E+21
Methylene chloride	75-09-2	1.98E+12	7.01E+15	2.00E+20	3.68E+12	3.73E+11	3.77E+20
Naphthalene	91-20-3	2.02E+10	5.31E+14	5.49E+17	5.15E+10	1.58E+11	1.42E+18
Nitrobenzene	98-95-3	1.93E+10	3.18E+11	3.14E+15	2.86E+10	9.74E+08	4.78E+15
PCB	608-93-5	6.55E+09	7.31E+11	1.26E+12	1.45E+10	1.02E+09	2.83E+12
Pyrene	129-00-0	1.32E+12	6.26E+15	1.14E+14	1.57E+15	2.50E+13	1.35E+17
Styrene	100-42-5	7.30E+12	2.78E+16	4.88E+19	1.27E+13	1.15E+12	8.62E+19
2,3,7,8-TCDD	1746-01-6	4.07E+04	6.09E+04	7.09E+04	4.77E+05	7.69E+02	8.14E+05
1,1,2,2-Tetrachloroethane	79-34-5	1.19E+14	3.73E+10	5.42E+14	2.13E+14	1.93E+07	9.80E+14
Tetrachloroethylene	127-18-4	3.21E+11	5.31E+15	3.66E+19	5.22E+11	5.41E+10	5.94E+19
Toluene	108-88-3	2.64E+12	6.35E+16	4.72E+20	5.72E+12	1.43E+12	1.02E+21
Toxaphene	8001-35-2	2.68E+13	1.11E+10	5.38E+13	2.84E+13	2.93E+06	5.72E+13
1,2,4-Trichlorobenzene	120-82-1	6.42E+12	8.99E+15	1.22E+19	7.77E+12	7.79E+11	1.46E+19
1,1,1-Trichloroethane	71-55-6	9.63E+13	1.65E+16	2.44E+20	1.09E+14	1.50E+11	2.75E+20
1,1,2-Trichloroethane	79-00-5	1.28E+11	8.99E+13	2.20E+18	1.42E+11	1.43E+10	2.51E+18
Trichloroethylene	79-01-4	6.60E+11	4.50E+16	8.62E+20	7.80E+11	7.79E+11	1.02E+21
Vinyl chloride	75-01-4	6.60E+11	1.83E+15	4.39E+19	7.00E+11	1.06E+10	4.71E+19
Xylenes (total)	1330-20-7	6.55E+13	5.84E+17	2.93E+21	1.24E+14	1.24E+13	5.42E+21
Antimony	7440-36-0		1.37E+09			4.79E+08	
Arsenic	7440-38-2		1.14E+10			3.99E+09	
Barium	7440-39-3		4.57E+11			1.60E+11	
Cadmium	7440-4-39		1.14E+09			3.99E+08	
Chromium (Hexavalent)	18540-29-9		2.29E+10			7.98E+09	
Copper	7440-50-8		2.97E+11			1.04E+11	
Lead	7439-92-1	1.61E+11	3.43E+09			1.20E+09	
Manganese	7439-96-5	5.36E+09	2.73E+12	7.69E+16		9.51E+11	
Nickel	7440-02-0	9.65E+09	3.90E+11	1.10E+16		1.36E+11	
Zinc	7440-66-6		5.84E+12	1.65E+17		2.04E+12	
Particulate matter		5.36E+12					

10.5.3 Empirical Approach for Carrying Capacity Estimate

One can estimate the carrying capacity for chemicals other than those listed in Table 10.4 by following the steps previously described. Even though the CalTOX is a simple model, some input data are still required, especially for specific chemical and physical properties. The target threshold levels can be adopted from regulatory levels and associated databases and the specific chemical and physical properties can be obtained by various means from several sources. However, the collection of specific chemical and physical properties is costly and time consuming considering that a large number of chemicals are used worldwide and toxicological information for only a portion of them is reported in the literature. An applicable streamlined procedure to estimate the carrying capacity for other chemicals is the use of an empirical approach. A simplified calculation for carrying capacity estimate can be made using the average of the partitioning factors of those 78 organic chemicals discussed in the preceding section.

Basically, an arithmetic mean can be used for the calculation of an empirical carrying capacity estimate. However, in this research the median values derived from probability analysis are used instead to minimize the bias due to outlier data. Figure 10.3 presents probability plots for the partitioning factors of those 78 organic chemicals. These plots are the Z-test statistical analysis of the partitioning factors (in Table 10.3) to determine their average values for the partition in air (top), water (middle), and soil (bottom) from both air and water emissions. Table 10.5 summaries the average partitioning factors obtained from the log normal probability plots.

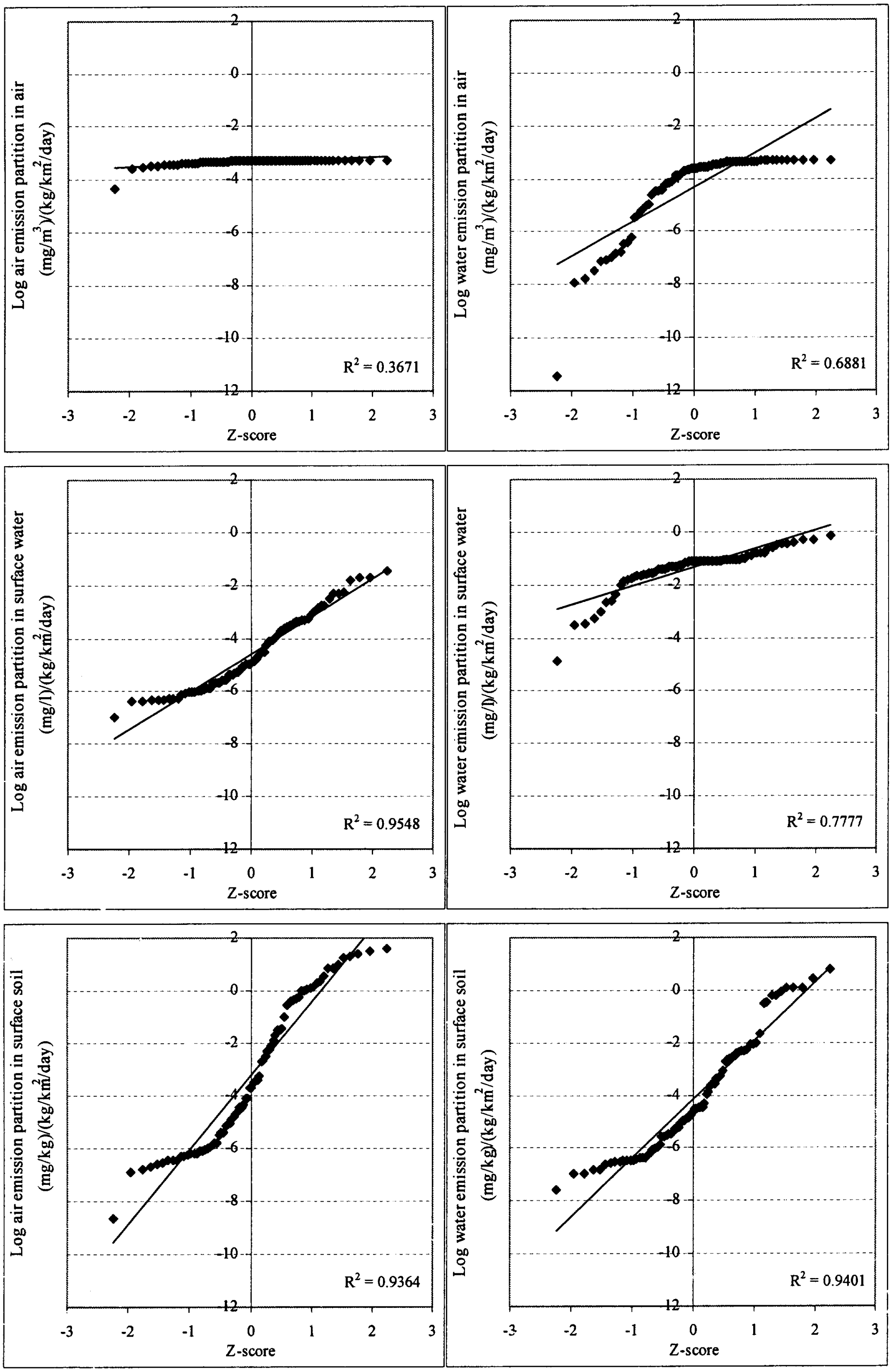

Figure 10.3 Probability plots for the partitioning factors of 78 organic chemicals derived from CalTOX results.

Table 10.5 Average Partitioning Factors for Empirical Carrying Capacity Estimation

Partition of air emissions (Partitioning factor)			Partition of water emissions (Partitioning factor)		
In air	In surface water	In surface soil	In air	In surface water	In surface soil
(mg/m^3) / (kg/km^2/d)	(mg/l) / (kg/km^2/d)	(mg/kg) / (kg/km^2/d)	(mg/m^3) / (kg/km^2/d)	(mg/l) / (kg/km^2/d)	(mg/kg) / (kg/km^2/d)
4.6E-04	2.5E-05	5.5E-04	4.6E-05	4.4E-02	7.1E-05

An empirical carrying capacity can be calculated using Equations (10.5), (10.6), and (10.7) providing that the target threshold levels have been selected. For the empirical estimate, the average partitioning factors in Table 10.5 are used instead of the chemical-specific partitioning factors derived from CalTOX results. Threshold levels for some commonly emitted pollutants are presented in Table 10.6. These toxic chemicals are some of the top 40 chemicals on the 1997 TRI list (in terms of the release amount) and some selected chemicals commonly used in electronics industries. The carrying capacity estimate for these chemicals using the empirical approach is presented in Table 10.7.

Table 10.6 Summary of the Threshold Levels for Some Commonly Emitted Chemicals

Substance	CAS	Reference value						Threshold level		
		Inhalation			Ingestion			Air	Surface water	Surface soil
		Value	Unit	Note	Value	Unit	Note	mg/m^3	mg/l	mg/kg
Ammonia	7664-41-7	3.0E-01	ppm	MRL	3.0E-01	mg/kg/d	MRL	2.1E-01	2.3E+04	1.9E+05
1,3-Butadiene	106-99-0	3.0E-05	mg/m3	RfC				3.0E-05		
Carbon Monoxide	630-08-0	1.0E-02	mg/m3	NAAQS				1.0E-02		
Chlorine	7782-50-5				1.0E-01	mg/kg/d	RfD		7.7E+03	6.4E+04
Cyanide	57-12-5				2.0E-02	mg/kg/d	RfD		1.5E+03	1.3E+04
Cyclohexane	110-82-7	6.0E+00	mg/m3	RfC				6.0E+00		
Decabromodiphenyl Oxide	1163-19-5				1.0E-02	mg/kg/d	RfD		7.7E+02	6.4E+03
Dichloromethane	75-09-2	2.0E-03	mg/m3	CPF	6.0E-02	mg/kg/d	RfD	2.0E-03	4.6E+03	3.8E+04
Diisocyanates	26471-62-5	7.0E-05	mg/m3	RfC				7.0E-05		
Ethylene Glycol	107-21-1	5.0E-01	ppm		2.0E+00	mg/kg/d	RfD	3.5E-01	1.5E+05	1.3E+06
Formaldehyde	50-00-0	8.0E-05	mg/m3	CPF	2.0E-01	mg/kg/d	RfD	8.0E-05	1.5E+04	1.3E+05
Formic Acid	64-18-6	9.4E+00	mg/m3	OSHA				9.4E+00		
Hydrochloric Acid	7647-01-0	2.0E-02	mg/m3	RfC				2.0E-02		
Hydrogen Fluoride	7664-39-3	2.0E-02	ppm	MRL				1.6E-02		
Methanol	67-56-1				5.0E-01	mg/kg/d	RfD		3.8E+04	3.2E+05
Methyl Ethyl Ketone	78-93-3	5.0E+00	mg/m3	RfC	6.0E-01	mg/kg/d	RfD	5.0E+00	4.6E+04	3.8E+05
Methyl Isobutyl Ketone	108-10-1	3.0E+00	mg/m3	RfC				3.0E+00		
N-Butyl Alcohol	71-36-3				1.0E-01	mg/kg/d	RfD		7.7E+03	6.4E+04
n-Hexane	110-54-3	2.0E-01	mg/m3	RfC				2.0E-01		

Table 10.6 Summary of the Threshold Levels for Some Commonly Emitted Chemicals (Continued)

Substance	CAS	Reference value						Threshold level		
		Inhalation			Ingestion			Air	Surface water	Surface soil
		Value	Unit	Note	Value	Unit	Note	mg/m^3	mg/l	mg/kg
Nitrate Compounds	14797-55-8				1.6E+00	mg/kg/d	RfD		1.2E+05	1.0E+06
Nitric Acid	7697-37-2	5.1E+00	mg/m3	OSHA				5.1E+00		
Nitrogen Dioxide	10102-44-0	1.0E-01	mg/m3	NAAQS				1.0E-01		
Ozone	10028-15-6	8.0E-02	ppm	NAAQS				1.6E-01		
Phenol	108-95-2				3.0E-01	mg/kg/d	RfD		2.3E+04	1.9E+05
Phosphoric Acid	7664-38-2	1.0E-02	mg/m3	RfC				1.0E-02		
Sulfur Dioxide	7446-09-5	8.0E-02	mg/m3	NAAQS				8.0E-02		
Sulfuric Acid	7664-93-9	1.0E+00	mg/m3	OSHA				1.0E+00		

Table 10.7 Carrying Capacity Estimate for Some Commonly Emitted Chemicals Using Empirical Approach

Substance	CAS	U.S. carrying capacity (kg/yr)					
		Air emissions			Water emissions		
		Air	Surface water	Surface soil	Air	Surface water	Surface soil
Ammonia	7664-41-7	1.55E+12	3.16E+18	1.20E+18	1.55E+13	1.79E+15	9.28E+18
1,3-Butadiene	106-99-0	2.24E+08			2.24E+09		
Carbon Monoxide	630-08-0	7.46E+10					
Chlorine	7782-50-5		1.05E+18	3.99E+17		5.98E+14	3.09E+18
Cyanide	57-12-5		2.10E+17	7.99E+16		1.20E+14	6.19E+17
Cyclohexane	110-82-7	4.48E+13			4.48E+14		
Decabromodiphenyl Oxide	1163-19-5		1.05E+17	3.99E+16		5.98E+13	3.09E+17
Dichloromethane	75-09-2	1.49E+10	6.31E+17	2.40E+17	1.49E+11	3.59E+14	1.86E+18
Diisocyanates	26471-62-5	5.22E+08			5.22E+09		
Ethylene Glycol	107-21-1	2.59E+12	2.10E+19	7.99E+18	2.59E+13	1.20E+16	6.19E+19
Formaldehyde	50-00-0	5.97E+08	2.10E+18	7.99E+17	5.97E+09	1.20E+15	6.19E+18
Formic Acid	64-18-6	7.00E+13			7.00E+14		
Hydrochloric Acid	7647-01-0	1.49E+11			1.49E+12		
Hydrogen Fluoride	7664-39-3	1.22E+11					
Methanol	67-56-1		5.26E+18	2.00E+18		2.99E+15	1.55E+19
Methyl Ethyl Ketone	78-93-3	3.73E+13	6.31E+18	2.40E+18	3.73E+14	3.59E+15	1.86E+19
Methyl Isobutyl Ketone	108-10-1	2.24E+13			2.24E+14		
N-Butyl Alcohol	71-36-3		1.05E+18	3.99E+17		5.98E+14	3.09E+18
n-Hexane	110-54-3	1.49E+12			1.49E+13		
Nitrate Compounds	14797-55-8		1.68E+19	6.39E+18		9.56E+15	4.95E+19
Nitric Acid	7697-37-2	3.83E+13			3.83E+14		
Nitrogen Dioxide	10102-44-0	7.46E+11					
Ozone	10028-15-6	1.17E+12					
Phenol	108-95-2		3.16E+18	1.20E+18		1.79E+15	9.28E+18
Phosphoric Acid	7664-38-2	7.46E+10			7.46E+11		
Sulfur Dioxide	7446-09-5	5.97E+11					
Sulfuric Acid	7664-93-9	7.46E+12			7.46E+13		

10.6 Summary

The carrying capacity for human toxicity impact is estimated for 89 chemicals using the threshold-oriented technique in conjunction with CalTOX (Table 10.4). In this approach, the chemical-specific emission-concentration relationship (the partitioning factor) is estimated using CalTOX. Concentrations of a chemical in target media (air, water, and surface soil) resulting from a unit emission of the pollutant are obtained. As the linear function of emission-concentration relationship is assumed, the carrying capacity can be determined from the emission of the pollutant that results in the concentration in a target medium not exceeding the desirable threshold level. The threshold levels for air, surface water, and surface soil are taken into account. The target threshold levels are adopted mainly from regulatory standards (e.g., NAAQS, MCL) and the toxicity database for risk assessment (e.g., IRIS and MRL). The carrying capacity is estimated for those three types of environmental media resulting from both air and water emissions.

The uncertainty associated with the carrying capacity estimate in the threshold-oriented technique using CalTOX is the result of several factors including the selected threshold levels, specific physical and chemical properties of the pollutants, and the assumptions made from the landscape information in CalTOX. The uncertainty assessment of using CalTOX has been well documented by Hertwich (1999).

There is a difficulty in providing the carrying capacity for all chemicals or pollutants, not only because the data are not available, but also because it is costly to do so. Therefore, an empirical approach may be used to estimate the carrying capacity for chemicals not assessed using the threshold-oriented technique in conjunction with CalTOX. The empirical approach can be used for the carrying capacity estimate based on the fact that the carrying capacity is relatively proportional to the target threshold levels. The approach also assumes average partitioning factors derived from the data set of 78 organic chemicals. By using the empirical approach, the carrying capacity can be estimated directly from a target threshold level (as presented in Table 10.7). In this chapter, the carrying capacity of 20[1] from the top 40 chemicals on the TRI list is estimated using the empirical approach.

Major assumptions used in making of carrying capacity estimates in this chapter are twofold. First, one year concentrations obtained from CalTOX simulations are used in place of steady-state concentrations in the determination of partitioning factors. This assumption causes an overestimation of carrying capacity values. Second, it is assumed that the partitioning factors obtained from CalTOX are valid for inorganic chemicals. Therefore, one should be aware of this assumption when using carrying capacities for particulate matter, metals, and particles. The particulate carrying capacity need to be used with care and will need further attention in the future. Other than addressing these assumptions, further improvements of the carrying capacity for human toxicity impact are certainly possible. For example, the carrying capacity for a chemical can be revised when the target threshold level is updated. A more recent target threshold may be due to changes in the regulatory levels and standards and updates of the toxicity databases. The revision of the carrying capacity can be accomplished by using the CalTOX simulation or by using the empirical approach. Further improvement of the accuracy of carrying capacity estimates may also be achieved by the use of a more sophisticated and accurate environmental fate and transport model. An estimate of the carrying capacity that takes into account the spatial variation can be carried out using locally-specific modeling.

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[1] The carrying capacity for other chemicals from top 40 TRI chemicals have been estimated using the threshold-oriented technique in conjunction with CalTOX. The top 40 TRI chemicals were accounted for approximately 90% by weight of the total toxic release in 1997.

CHAPTER 11

ECO-TOXICITY IMPACT

11.1 Overview

Hazardous substances are not only toxic to human health, but they also affect ecological living organisms in many ways. In most considerations of risk assessment in the past, risks to human health have always been emphasized and risks to the ecological effects have largely been ignored. This may lead to the mistaken belief that protection of human health is more important or that it automatically protects ecosystems. However, some chemicals that may pose no risk or negligible risk to human health may cause severe effects to other organisms.

In LCA, eco-toxicity impact is an impact category that should be included in the assessment (ISO 1998b; 2001; Udo de Haes 1999a; 1999b). However, the difficulty in addressing eco-toxicity impact in LCA is due to the discrepancy in specific types of species/subsystem in ecosystems (Udo de Haes 1999b; Hauschild and Pennington 2002). Furthermore, toxic chemicals exhibit a wide range of effect mechanisms and, as a result, it is not possible to list all toxic chemicals and relate their effects to one reference substance having the same effect mechanism (Olsen and Hauschild 1998).

Generally, mass release, fate (partitioning, removal and exposure), and effects (toxicity) of chemicals are taken into account in LCIA for both human toxicity and eco-toxicity impacts. As in the case of the human toxicity impact, the eco-toxicity impact assessed in LCA is usually associated with toxicity potentials or damage potentials of toxic chemicals. The fate component for the determination of toxicity potentials or characterization factors for human toxicity and eco-toxicity is usually identical.

However, the methodology for eco-toxicity impact differs primarily in terms of the effect endpoint measure (Pennington 2001). In the LCIA characterization, the calculation of an eco-toxicity (as well human toxicity) impact score due to a release of a toxic chemical can be expressed in equation form as (Margni et al. 2002):

$$S = \sum_{m} CFT_m M_m \tag{11.1}$$

Where S is the impact score, CFT is the characterization factor or effect factor (in terms of toxicity potential or damage potential), M is the inventory emission of the toxic chemical, and subscript m is the environmental medium. Impact subcategories (environmental compartments) that may be considered are aquatic eco-toxicity for fresh water and marine water, sediment eco-toxicity for freshwater and marine water, and terrestrial eco-toxicity (Wenzel et al. 1997; Guinee 2001; Hauschild and Pennington 2002; Jolliet et al. 2003). Typically, NOAEL or LOAEL (no-observed-adverse-effect level or lowest-observed-adverse-effect), PNEC (predicted no-effect environmental concentration level), LC50 (lethal concentration for 50% of test organisms), and EC50 (effect concentration 50% of test organisms) are used as the basis for estimating the toxicity potential.

There are a few methodologies that have been developed to estimate the characterization factors for eco-toxicity impact. Guinee et al. (1996) and Huijbregts et al. (2000) used a multimedia environmental model (Uniform System for the Evaluation of Substances; USES) to assess exposure and estimate the toxicity potentials, the relative contribution of emissions to related ecosystems, in a fashion similar to human toxicity potentials. Koudijs and Dutilh (1998) calculated aquatic toxicity potentials for 65 frequently used herbicides and pesticides using this approach for crop protection.

Huijbregts et al. (2000) also extended this approach to estimate aquatic and terrestrial toxicity potentials for 181 toxic chemicals in terms of 1,4-dichlorobenzene-equivalent.

Jolliet (1994) and Jolliet and Crettaz (1997) developed the semi-empirical Critical Surface Time approach to determine the overall response of the environmental system and to calculate the effect factors as the ratio of the measured ambient concentration to the corresponding total emission of the system. The equivalent dilution volume per unit surface (water or land) of toxic chemicals can then be calculated. For chemicals for which the measured ambient concentrations are not available, the effect factors can be extrapolated as a function of their lifetimes. The effect factors for approximately 100 toxic chemicals derived from this approach are expressed in terms of lead-equivalent.

Olsen and Hauschild (1998) developed the critical volume approach to determine the toxicity factors for EDIP. The toxicity factors are the volume of the environmental compartment (air, soil, or water) needed to dilute the polluted compartment to the no-effect concentration (toxicity factor = no-effect-concentration^{-1}).

In EI95 (Goedkoop 1998), eco-toxicity was estimated from pesticide effects. The aggregation of pesticides can be made without the application of toxicity potentials. A reduction factor of 25 was used as the distance-to-target valuation parameter. This relatively high reduction factor indicates that pesticide is a major concern in Europe. It was assumed that this reduction factor was necessary to reduce the 65% contamination of groundwater across Europe to 10%. By using this reduction factor, the European target values of 1.92×10^7 kg/yr for pesticides was used in EI95. EI99 (Goedkoop and Spriensma 2001) extends the toxic substances affecting ecosystems to chemicals other than pesticides. The damage model in EI99 uses the multimedia modeling methodology developed by Guinee et al. (1996) mentioned earlier to determine the damage equivalency to European ecosystem quality.

Eco-toxicity impact is also considered as a non-homogeneous impact category as discussed in Chapter 10, wherein adverse impacts to ecosystems are caused by different toxic substances and these substances are not interchangeable in terms of environmental exposure and effects. Different toxic chemicals affect different species in an ecosystem and it is not possible to calculate the additive and synergistic effects on this ecosystem caused by all toxic chemicals. Therefore, the carrying capacity for eco-toxicity must be estimated individually for specific chemicals and an aggregation or trade-off in terms of toxicity potential among toxic chemicals is not allowed. In other words, the carrying capacity for eco-toxicity impact consists of multiple carrying capacity estimates for impact subcategories, representing the individual toxic chemicals.

This chapter presents an empirical approach for carrying capacity estimate for eco-toxicity impact. This approach uses empirical partitioning factors, calculated in the evaluation of the carrying capacity for human toxicity impact (Chapter 10), coupled with the threshold levels for ecosystems to estimate the carrying capacity. The important aspects of the evaluation of carrying capacity for eco-toxicity impact are the selection of the indicator for ecosystem health and the selection of the threshold levels.

11.2 Approach

The carrying capacity for toxic chemicals affecting ecosystems can be estimated using the threshold-oriented technique discussed in Section 4.2, Chapter 4. Basically, the approach for eco-toxicity impact is the same as the one for human toxicity impact in terms of fate and transport of toxic chemicals. The carrying capacity estimate for eco-toxicity impact differs in terms of the effect endpoint measure, i.e., threshold levels. Therefore, the carrying capacity estimate for eco-toxicity impact can be determined simply by using the empirical partitioning factors (a partitioning factor is the factor that expresses the concentration of a toxic chemical in an environmental medium due to a unit of continuous air or surface-water emission of that toxic chemical), which have been determined in the evaluation of carrying capacity for human toxicity, coupled with threshold levels for ecosystem health. The carrying capacity for eco-toxicity impact can be calculated using the empirical equation (subscript m denotes environmental medium):

$$Carrying\ Capacity_m = \frac{Threshold\ level_m \times Area}{Partitioning\ factor_m} \tag{11.2}$$

11.3 Threshold Levels for Carrying Capacity Estimate

Unlike human toxicity impact where the assessment endpoint is known such as damage to human health, the assessment endpoint for eco-toxicity cannot be identified due to the variation of living organisms in ecosystems. As such, estimating the carrying capacity for eco-toxicity is a complicated process involving the selection of end point indicators and employing a definite threshold level. Hauschild and Pennington (2002) suggested that the criteria for selecting an indicator for eco-toxicity are: scientific validity, environmental relevance, transparency, reproducibility, uncertainty quantification, complexity, feasibility, and data availability.

In risk assessment, the assessment endpoint for an ecosystem is selected on a case-by-case basis with a well-defined problem formulation (EPA 1998b). For example, Suter and Tsao (1996) proposed using toxicological benchmarks for more than 100 chemicals for their effects on aquatic biota. These benchmarks are based on a 20%

reduction in the population of a fish species (largemouth bass) as the assessment endpoint for Oak Ridge Reservation ecological risk assessments. A similar process was used to propose benchmarks for wildlife (Sample et al. 1996).

In the eco-toxicity impact, the carrying capacity may be thought of as the absorbing capacity that keeps the ecosystems sustainable. It is unrealistic to select an assessment endpoint of a single organism to represent an entire regional scale ecosystem. Selection of an assessment endpoint as a surrogate for the entire ecosystem will carry a high degree of subjectivity. To date, the absolute generic assessment endpoint does not exist. As pointed out by Suter (2000), generic assessment endpoints are needed and will be very useful for ecological risk assessment. Assessment endpoints may be derived from the concepts of ecological integrity, sustainability, resiliency, and biodiversity (EPA 1997b). In an attempt to define generic assessment endpoints, an EPA workgroup proposed an initial list of priorities for ecological protection (Table 11.1). However, specific entities are not yet established at this early stage.

Table 11.1 List of Priority Ecological Entities in Ecosystems
(Source: EPA 1997b)

Category	Ecological Entity
Animals, plants, and their habitats	1. Aquatic communities in lakes, streams, and estuaries 2. Regional populations of native species and their habitats -terrestrial and aquatic 3. Groups of native or migratory species exposed to severe or acute threat
Whole ecosystems	4. Ecosystem functions and services 5. Wetlands and stream corridors
Special places and species	6. Endangered ecosystems (e,g,, old-growth forests, tall-grass prairies) 7. Endangered species and their habitats 8. Other places with high ecological or societal value, as appropriate

The target threshold levels may also be adopted from the chronic ecosystem-level no-observed-effect concentration ($NOEC_e$). $NOEC_e$ can be calculated from the toxicity values of the ECOTOX database (EPA 2000c). ECOTOX is an online-based system that provides chemical-specific toxicity values summarized from laboratory tests for aquatic life, terrestrial plants, and terrestrial wildlife. Toxicity endpoint values acquired from ECOTOX are at the organism-level. Therefore, the $NOEC_e$ (mg/l or mg/m^3) must be extrapolated from the single-species acute tests (LC_{50} and EC_{50}, mg/l or mg/m^3) or chronic tests ($NOEC_s$, mg/l or mg/m^3) for the same chemical. The extrapolation may be carried out using the following equations proposed by Sloof et al. 1986:

$$NOEC_e = 10^{(0.81 \log LC50 - 0.55)} \tag{12.3}$$

$$NOEC_e = 10^{(0.85 \log NOECs + 0.63)} \tag{12.4}$$

For example, a NOAEL or NOEC for plankton or minnow may be selected as a representative to define the endpoint or threshold level for an environmental performance assessment. In some cases, where NOEC are not available, it can be calculated from chronic low-observed-adverse level (LOAEL) applying the uncertainty factor of 10.

$$NOAEL\ (mg/l) = LOAEL\ (mg/l)\ /\ 10 \tag{12.5}$$

However, the difficulty of applying the above approach is also related to the selection of an appropriate representative living organism. For many chemicals, ECOTOX provides different laboratory results from testing several living organisms. There are no fixed criteria in the selection of a single data set associated with one living species over the others.

Due to the difficulty previously discussed in the determination of generic assessment endpoints, the more agreeable PNEC (predicted-no-effect-concentration) is adopted as the threshold level for the eco-toxicity carrying capacity estimate. In this application, PNEC is assumed to be the level that is desired to protect the functioning of the ecosystem as a whole, which in concept is a most desirable approach to ecosystem integrity. Basically, PNEC is used in the ecological risk assessment's effects assessment step as an ecotoxicological measure for multiple-species systems to determine the risk quotient or hazard quotient[1]. PNEC is defined as the highest environmental concentration expected to cause no effects, acute or chronic, on the structure or functioning of ecosystems (EC 2003). PNEC may be practically considered as the concentration below which a specified percentage of species in an ecosystem is expected to be protected; e.g., the protection level of 95% of species is often selected as an initial basis for PNEC derivation (Pennington 2003). PNEC is frequently used in several LCIA methodology studies to determine the characterization of risk to an ecosystem associated with a chemical use (Wenzel et al. 1997; Huijbregts et al. 2000; Guinee 2001). Evaluation of PNEC requires data on chronic toxicity to several different species at different (three or more) trophic levels of the ecosystems (Hauschild and Pennington 2002). Methods for evaluation of PNEC can be classified into the following categories (Hauschild and Pennington 2002):

- Measuring in field meso- and microcosm tests;
- Measuring of PNEC on the basis of laboratory tests performed on individual species applying extrapolating factors/assessment factors/uncertainty factors;
- Evaluation and comparison of the different PNEC estimation methods; and
- Estimation of PNEC for terrestrial ecosystems and sediments.

In this research, PNECs are obtained from the database developed in an OECD (Organization for Economic Co-operation and Development) project on High Production Volume (HPV) chemicals (OECD 2000). HPV chemicals are defined as chemicals

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[1] Risk Quotient (RQ) or Hazard Quotient (HQ) = PEC/PNEC where PEC = Predicted environmental concentration and PNEC = Predicted no effect concentration.

reported to be produced or imported at levels greater than 1,000 tons per year in at least one member country of the OECD. In this project, parameters for risk assessment of HPV chemicals are to be investigated, reported, and periodically updated by OECD member countries that volunteer to work on the project. At this stage, there are 15 of 30 OECD member countries working on more than 1,000 HPV chemicals. For the U.S., the EPA is responsible for the investigation and reporting of 420 HPV chemicals (EPA 2003j). PNECs developed in the OECD project may be perceived as a consensus because the evaluations for HPV chemicals need to be approved by the OECD before they publicized in the Integrated HPV Database.

However, PNEC of only 132 chemicals have been reported so far. (Table E.1, Appendix E). These PNEC reports are limited because:

- The project is still in the early stage and only a portion of the listed HPVC chemicals have been investigated;
- The supporting laboratory results and field tests for some chemicals are not sufficient to determine their PNECs; and
- Some chemicals are determined to pose a very low risk to ecosystems; therefore, it is not necessary to evaluate associated risks as well as their PNECs.

An example of PNEC assessment is the one arranged for chromates by the U.K. Environment Agency (OECD 2000). For aquatic PNEC, long term NOEC values for 28 species, derived from the literature, were in the range of 0.0047 to 3.5 mg Cr(VI)/liter. By using statistical log-normal distribution, a value of lower 5% with 50% confidence was determined as 10.2 μg Cr(VI)/liter. A factor of 3 was applied when the limitations in the database were taken into account, giving the aquatic PNEC of 3.4 μg Cr(VI)/liter. For sediment PNEC, using of an equilibrium partitioning method and different partition coefficients for acidic and neutral-alkaline environments gives sediment PNECs of 1.5

mg Cr(VI)/liter for acidic conditions and 0.15 mg Cr(VI)/liter for alkaline conditions. For terrestrial PNEC, long term toxicity data were available for three trophic levels (plants, earthworms, and soil microorganisms), with plants being the most sensitive species group. The lowest NOEC from a plant growth test was 0.35 mg Cr(VI)/kg. A factor of 10 was applied, giving the soil PNEC of 35 μg Cr(VI)/kg.

11.4 Carrying Capacity Estimate

When utilizing PNECs as the target threshold levels, the national scale (U.S.) carrying capacity for both air and water emissions for eco-toxicity impact can then be estimated using the empirical equations similar to the ones developed from the evaluation of carrying capacity for human toxicity impact:

$$U.S.\ CC_{water}\ (kg/yr) = \frac{\underline{Threshold\ level\ (mg/l) \times 9.4{\times}10^{6}\ km^{2} \times 365\ days/yr}}{Partitioning\ factor_{water}\ (mg/l)/(kg/km^{2}/day)} \qquad (10.6)$$

$$U.S.\ CC_{soil}\ (kg/yr) = \frac{\underline{Threshold\ level\ (mg/kg) \times 9.4{\times}10^{6}\ km^{2} \times 365\ days/yr}}{Partitioning\ factor_{soil}\ (mg/kg)/(kg/km^{2}/day)} \qquad (10.7)$$

Subscripts water and soil denote the carrying capacity estimate from the threshold levels for water (aquatic eco-toxicity) and soil (terrestrial eco-toxicity), respectively. The partitioning factors are the same ones developed in Chapter 10 (Table 11.2). Most PNECs reported in literature are for aquatic ecosystems; therefore, only aquatic carrying capacity estimates are determined. Table 11.3 presents the U.S. carrying capacity estimates for eco-toxicity impact for 131 chemicals using the empirical approach.

Table 11.2 Average Partitioning Factors for Empirical Carrying Capacity Estimate

Partition of air emissions (Partitioning factor)			Partition of water emissions (Partitioning factor)		
In air	In surface water	In surface soil	In air	In surface water	In surface soil
(mg/m^3) / (kg/km^2/d)	(mg/l) / (kg/km^2/d)	(mg/kg) / (kg/km^2/d)	(mg/m^3) / (kg/km^2/d)	(mg/l) / (kg/km^2/d)	(mg/kg) / (kg/km^2/d)
For organic chemicals					
4.6E-04	2.5E-05	5.5E-04	4.6E-05	4.4E-02	7.1E-05
For metals and particulate matter					
3.2E-05	1.5E-02	4.0E-03	-	4.3E-02	4.0E-15

11.5 Summary

The carrying capacity for eco-toxicity impact is estimated using the threshold-oriented technique developed in this research. The empirical carrying capacity estimate for eco-toxicity impact is based on two components: the assessment of environmental transport of toxic chemicals and the assessment of threshold levels. The assessment of environmental transport of toxic chemicals is adopted from the one that has been carried out in the evaluation of human toxicity impact. The partitioning of toxic chemicals in the environmental media as a result of a unit of emission are expressed in terms of partitioning factors, which were calculated using CalTOX, an integrated multimedia environmental model.

For the assessment of the threshold levels, it is difficult to select an appropriate set of threshold levels. This is because there are multiple effects on ecosystems caused by individual substances and there is insufficient supporting information to select a single entity to represent the functioning of ecosystem as a whole. In this study, the PNECs are used as a surrogate for the generic assessment endpoint to protect the ecosystem as a whole. The PNEC is defined as the highest environmental concentration expected to cause no effects, acute or chronic, on the structure or functioning of ecosystems. PNECs are therefore used as the target threshold levels.

The U.S. carrying capacity for eco-toxicity impact have been estimated using the empirical equations derived for human toxicity impact in Chapter 10. However, the carrying capacity is available for only 131 chemicals for aquatic eco-toxicity impact due to the limited number of reports on PNECs (Table 11.3). In the future, the carrying capacity for more chemicals can be determined in the same manner based upon the increased availability of PNECs or other endpoint indicators for ecological health. PNECs and other indicators, which are feasible as the threshold levels, may be evaluated from several toxicology databases. These databases are, for instance, ECOTOX (EPA 2000), EPA-National Recommended Water Quality Criteria (EPA 1999d), Pennsylvania-Guidelines for Development of Criteria for Toxic Substances and Water Quality Criteria for Toxic Substances (Pennsylvania EPA 2000), ORNL-Preliminary Remediation Goals for Ecological Endpoints (Efroymson et al. 1997), ORNL-Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota (Suter and Tsao 1996).

The next step in estimating eco-toxicity carrying capacity may also be the development of a consensus methodology or approach that can evaluate generic assessment endpoints/indicators at the ecosystem-level. Additional aspects include the specification of areas of protection in ecosystems. A refinement also could include the use of a more sophisticated and accurate environmental fate and transport model. Furthermore, a carrying capacity estimate that takes into account local conditions could be carried out using locally specific modeling.

Table 11.3 Carrying Capacity Estimate for Eco-Toxicity Impact

	CAS	Substance	Aquatic PNEC (mg/l)	U.S. Carrying capacity for aquatic eco-toxicity (kg/yr)	
				Air emissions	Water emissions
1	103231	(2-ethylhexyl) adipate	0.0035	4.80E+11	2.73E+08
2	482893	3H-Indol-3-one, 2-(1,3-dihydro-3-oxo-2H-indol-2-ylidene)-1,2-dihydro-	0.0078	1.07E+12	6.08E+08
3	127195	Acetamide, N,N-dimethyl-	0.5	6.86E+13	3.90E+10
4	103844	Acetamide, N-phenyl-	0.135	1.85E+13	1.05E+10
5	102012	Acetoacetanilide	0.32	4.39E+13	2.50E+10
6	67641	Acetone	21	2.88E+15	1.64E+12
7	107028	Acrolein	0.0034	4.67E+11	2.65E+08
8	79107	Acrylic acid	0.003	4.12E+11	2.34E+08
9	85535859	Alkanes, C 14-17, chloro-	0.0002	2.74E+10	1.56E+07
10	85535848	Alkanes, C10 - C13, chloro-	0.0005	6.86E+10	3.90E+07
11	7789095	Ammonium dichromate	0.0034	4.67E+11	2.65E+08
12	90040	Aniline, 2-methoxy-	0.0055	7.55E+11	4.29E+08
13	88744	Aniline, 2-nitro-	0.008	1.10E+12	6.24E+08
14	95761	Aniline, 3,4-dichloro-	0.003	4.12E+11	2.34E+08
15	101779	Aniline, 4,4'-methylenebis-	0.003	4.12E+11	2.34E+08
16	82451	Anthraquinone, 1-amino-	0.001	1.37E+11	7.80E+07
17	105602	Azepin-2-one, hexahydro-	0.13	1.78E+13	1.01E+10
18	98839	Benzene, (1-methylethenyl)-	0.018	2.47E+12	1.40E+09
19	26447405	Benzene, 1,1'-methylenebis(isocyanato-)	1	1.37E+14	7.80E+10
20	32534819	Benzene, 1,1'-oxybis-, pentabromo deriv.	0.00053	7.27E+10	4.13E+07
21	120821	Benzene, 1,2,4-trichloro-	0.004	5.49E+11	3.12E+08
22	95501	Benzene, 1,2-dichloro-	0.0063	8.65E+11	4.91E+08
23	106467	Benzene, 1,4-dichloro-	0.02	2.74E+12	1.56E+09
24	89612	Benzene, 1,4-dichloro-2-nitro-	0.01	1.37E+12	7.80E+08
25	105055	Benzene, 1,4-diethyl-	0.0093	1.28E+12	7.25E+08
26	88733	Benzene, 1-chloro-2-nitro-	0.026	3.57E+12	2.03E+09
27	100005	Benzene, 1-chloro-4-nitro-	0.0028	3.84E+11	2.18E+08
28	611063	Benzene, 2,4-dichloro-1-nitro-	0.00056	7.69E+10	4.37E+07
29	16470249	Benzenesulfonic acid, 2,2'-(1,2-ethenediyl)bis[5-[4-[bis(2-hydroxyethyl)am	0.2	2.74E+13	1.56E+10
30	100447	Benzyl chloride	0.001	1.37E+11	7.80E+07
31	590863	Butanal, 3-methyl-	0.0033	4.53E+11	2.57E+08
32	106887	Butane, 1,2-epoxy-	0.02	2.74E+12	1.56E+09
33	109693	Butane, 1-chloro-	0.14	1.92E+13	1.09E+10
34	110634	Butanediol	0.85	1.17E+14	6.63E+10
35	760236	Butene, 3,4-dichloro-	0.0083	1.14E+12	6.47E+08
36	623916	Butenedioic acid (2E)-, diethyl ester	0.0056	7.69E+11	4.37E+08
37	5281049	C.I.Pigment Red 57:1	0.03	4.12E+12	2.34E+09
38	58082	Caffeine	0.0058	7.96E+11	4.52E+08
39	1333820	Chromium trioxide	0.0034	4.67E+11	2.65E+08
40	5392405	Citral	0.01	1.37E+12	7.80E+08
41	119471	Cresol, 6,6'-di-tert-butyl-2,2'-methylenedi-	0.0068	9.33E+11	5.30E+08
42	98828	Cumene	0.022	3.02E+12	1.72E+09
43	110827	Cyclohexane	0.009	1.24E+12	7.02E+08
44	123422	Diacetone alcohol	1	1.37E+14	7.80E+10
45	81118	Diaminostilbene-2,2'-disulfonic acid	0.32	4.39E+13	2.50E+10
46	105997	Dibutyl adipate	0.02	2.74E+12	1.56E+09
47	107664	Dibutyl phosphate	0.66	9.06E+13	5.15E+10
48	118694	Dichlorotoluene	0.0032	4.39E+11	2.50E+08

Table 11.3 Carrying Capacity Estimate for Eco-Toxicity Impact (Continued)

	CAS	Substance	Aquatic PNEC (mg/l)	U.S. Carrying capacity for aquatic eco-toxicity (kg/yr)	
				Air emissions	Water emissions
49	77736	Dicyclopentadiene	0.032	4.39E+12	2.50E+09
50	111400	Diethylenetriamine	12	1.65E+15	9.36E+11
51	77781	Dimethyl sulfate	0.014	1.92E+12	1.09E+09
52	120616	Dimethyl terephthalate	0.096	1.32E+13	7.49E+09
53	6864375	dimethyl-4,4'-methylenebism (cyclohexylamine)	0.0021	2.88E+11	1.64E+08
54	123911	Dioxane	57.5	7.89E+15	4.48E+12
55	26444495	Diphenyl tolyl phosphate	0.0012	1.65E+11	9.36E+07
56	7681574	Disodium disulphite	0.1	1.37E+13	7.80E+09
57	128370	Di-tert-butyl-p-cresol	0.0014	1.92E+11	1.09E+08
58	112185	Dodecanamine, N,N-dimethyl-	0.0000235	3.23E+09	1.83E+06
59	60004	E.D.T.A	0.1	1.37E+13	7.80E+09
60	79345	Ethane, 1,1,2,2-tetrachloro-	0.14	1.92E+13	1.09E+10
61	1717006	Ethane, 1,1-dichloro-1-fluoro-	0.31	4.25E+13	2.42E+10
62	107062	Ethane, 1,2-dichloro-	1.1	1.51E+14	8.58E+10
63	75683	Ethane, 1-chloro-1,1-difluoro-	0.045	6.18E+12	3.51E+09
64	111773	Ethanol, 2-(2-methoxyethoxy)-	1	1.37E+14	7.80E+10
65	112345	Ethanol, 2-(butoxyethoxy)-	57.5	7.89E+15	4.48E+12
66	100378	Ethanol, 2-(diethylamino)-	0.044	6.04E+12	3.43E+09
67	127184	Ethene, tetrachloro-	0.051	7.00E+12	3.98E+09
68	141979	Ethyl acetoacetate	0.275	3.77E+13	2.14E+10
69	90387578	Formaldehyde, prods. with sulfonated 1,1'-oxybistoluene, sod	0.029	3.98E+12	2.26E+09
70	56815	Glycerol	777	1.07E+17	6.06E+13
71	106912	Glycidyl methacrylate	0.01	1.37E+12	7.80E+08
72	107222	Glyoxal	0.215	2.95E+13	1.68E+10
73	102067	Guanidine, 1,3-diphenyl-	0.006	8.23E+11	4.68E+08
74	822060	Hexamethylene diisocyanate	0.0774	1.06E+13	6.04E+09
75	7664393	Hydrofluoric acid	0.9	1.24E+14	7.02E+10
76	7722841	Hydrogen peroxide	0.01	1.37E+12	7.80E+08
77	123319	Hydroquinone	0.00044	6.04E+10	3.43E+07
78	868779	Hydroxyethyl methacrylate	0.141	1.94E+13	1.10E+10
79	1854268	Imidazolidinone, 4,5-dihydroxy-1,3-bis	0.6	8.23E+13	4.68E+10
80	80626	Methacrylate, methyl-	0.74	1.02E+14	5.77E+10
81	79414	Methacrylic acid	0.164	2.25E+13	1.28E+10
82	79209	Methyl acetate	0.32	4.39E+13	2.50E+10
83	1634044	Methyl t-butyl ether	2.6	3.57E+14	2.03E+11
84	6386385	Metilox	0.0025	3.43E+11	1.95E+08
85	135193	Naphthol	0.00085	1.17E+11	6.63E+07
86	107642	Octadecanaminium, N,N-dimethyl-N-octadecyl-, chloride	0.0062	8.51E+11	4.83E+08
87	78706	Octadien-3-ol, 3,7-dimethyl-	0.2	2.74E+13	1.56E+10
88	75569	Oxirane, methyl-	0.052	7.14E+12	4.05E+09
89	115775	Pentaerythritol	0.6	8.23E+13	4.68E+10
90	109660	Pentane	0.027	3.71E+12	2.11E+09
91	107415	Pentanediol, 2-methyl-	4.3	5.90E+14	3.35E+11
92	4457710	Pentanediol, 3-methyl-	1	1.37E+14	7.80E+10
93	123546	Pentanedione	0.005	6.86E+11	3.90E+08
94	80057	phenol A	0.0016	2.20E+11	1.25E+08
95	1879090	Phenol, 2-(1,1-dimethylethyl)-4,6-dimethyl-	0.14	1.92E+13	1.09E+10

Table 11.3 Carrying Capacity Estimate for Eco-Toxicity Impact (Continued)

	CAS	Substance	Aquatic PNEC (mg/l)	U.S. Carrying capacity for aquatic eco-toxicity (kg/yr)	
				Air emissions	Water emissions
96	1570645	Phenol, 4-chloro-2-methyl-	0.05	6.86E+12	3.90E+09
97	25154523	Phenol, nonyl-	0.00033	4.53E+10	2.57E+07
98	1477550	Phenylene-bis(methylamine)	0.047	6.45E+12	3.66E+09
99	101724	Phenylenediamine, N-(1-methylethyl)-N'-phenyl-	0.00034	4.67E+10	2.65E+07
100	91156	Phthalonitrile	0.14	1.92E+13	1.09E+10
101	104905	Picoline, 5-ethyl-	0.0689	9.46E+12	5.37E+09
102	2403885	Piperidinol, 2,2,6,6-tetramethyl-	0.037	5.08E+12	2.89E+09
103	7778509	Potassium dichromate (K2Cr2O7)	0.0034	4.67E+11	2.65E+08
104	109557	Propane, 1-Amino-3-dimethylamino-	0.056	7.69E+12	4.37E+09
105	57556	Propanediol	183	2.51E+16	1.43E+13
106	78977	Propanenitrile, 2-hydroxy-	0.0017	2.33E+11	1.33E+08
107	6846500	Propanoic acid, 2-methyl-, 2,2-dimethyl-1-(1-methylethyl)-1,3-propanediyl e	0.032	4.39E+12	2.50E+09
108	24800440	Propanol, [(1-methyl-1,2-ethanediyl)bis(oxy)]bis-	10	1.37E+15	7.80E+11
109	34590948	Propanol, 1(or 2)-(2-methoxymethylethoxy)-	19	2.61E+15	1.48E+12
110	110985	Propanol, 1,1'-oxydi-	32	4.39E+15	2.50E+12
111	13674845	Propanol, 1-chloro-, phosphate (3:1)	0.64	8.78E+13	4.99E+10
112	107982	Propanol, 1-methoxy-	208	2.85E+16	1.62E+13
113	108656	Propanol, 1-methoxy-, acetate	0.635	8.71E+13	4.95E+10
114	25265718	Propanol, oxybis-	32	4.39E+15	2.50E+12
115	79061	Propenamide	0.0204	2.80E+12	1.59E+09
116	7775113	Sodium chromate (Na2CrO4)	0.0034	4.67E+11	2.65E+08
117	10588019	Sodium dichromate	0.0034	4.67E+11	2.65E+08
118	100210	Terephthalic acid	8	1.10E+15	6.24E+11
119	64028	Tetrasodium E.D.T.A.	0.0015	2.06E+11	1.17E+08
120	58559	Theophylline	0.087	1.19E+13	6.78E+09
121	108883	Toluene	0.074	1.02E+13	5.77E+09
122	95738	Toluene, 2,4-dichloro-	0.02	2.74E+12	1.56E+09
123	95498	Toluene, 2-chloro-	0.014	1.92E+12	1.09E+09
124	88722	Toluene, 2-nitro-	0.05	6.86E+12	3.90E+09
125	108441	Toluidine	0.0001	1.37E+10	7.80E+06
126	91769	Triazine-2,4-diamine, 6-phenyl-	0.0191	2.62E+12	1.49E+09
127	126738	Tributyl phosphate	0.037	5.08E+12	2.89E+09
128	512561	Trimethyl phosphate	3.2	4.39E+14	2.50E+11
129	115866	Triphenyl phosphate	0.00074	1.02E+11	5.77E+07
130	2432997	Undecanoic acid, 11-amino-	0.045	6.18E+12	3.51E+09
131	88120	Vinyl-pyrrolidinone	0.045	6.18E+12	3.51E+09

CHAPTER 12

RESOURCE DEPLETION IMPACT

12.1 Overview

Resource depletion can be defined as the decreasing availability of natural resources that influence human health, biodiversity, or material welfare (Guinee and Heijungs 1995). Resource depletion is a major impact often taken into account in LCA studies (Guinee and Heijungs 1995; Udo de Haes et al. 1999b; Brentrup et al. 2002). Furthermore, resource depletion is a primary indicator of whether sustainability is being maintained (Murcott 1997). Preservation of natural resources for the needs of future generations is a practice that is important in sustainable development. It is impractical to expect humans to discontinue consuming natural resources to accommodate their own needs if a growing economy is still expected. A more appropriate method is to reduce, if not avoid, consuming resources that have a greater potential for depletion. For example, if two different materials can be used to manufacture the same product, the more abundant resource should be used and the scarcer one conserved. An appropriate environmental performance indicator can play an important role as a decision-making tool for choosing more environmentally friendly raw materials and resources.

From the LCA perspective, three input-related impact categories have been proposed (Table 12.1). To date, only the impact assessment methods for consumption of abiotic resources such as fossil fuels and minerals have been developed and widely used. For the conventional LCIA, it is widely accepted that the potential impact for the consumption/depletion of resources can be aggregated using characterization or resource equivalency factors (Guinee and Heijungs 1995; Ecobalance 2000; Pennington 2000;

Brentrup et al. 2002; Van Oers et al. 2002). However, there is still no consensus on the value of the resources or on the most appropriate method to characterize these resources (Bare et al. 2002). Characterization factors can be determined from different methods including ones based on energy content of resources (Finnveden 1994; Van Oers et al. 2002) or the energy used for material production (Goedkoop 1998; Goedkoop and Spriensma 2001), based on the monetary value of resources (Steen 1999), and based on the physical data about reserves and consumption rate (Guinee and Heijungs 1995; Lee 1998; Ecobalance 2000; Brentrup et al. 2002; Van Oers et al. 2002).

Table 12.1 Proposed Input-Related Impact Categories
(Source: Udo de Haes et al. 1999b)

Impact category	Impact subcategory
Extraction of abiotic resources	Extraction of Deposits (e.g., fossil fuels, minerals) Extraction of Funds (e.g., groundwater, sand, clay) Extraction of Flow resources (e.g., solar energy, wind, surface water)
Extraction of biotic resources	-
Land use	Increase of land competition Degradation of life support functions Bio-diversity degradation

For the conventional LCIA, a general equation for calculating the potential impact for resource depletion is given by (Guinee and Heijungs 1995):

$$\textit{Impact or depletion} = \sum_i \textit{Equivalency factor}_i \times \textit{Extraction}_i \qquad (12.1)$$

Substituting an equivalency factor by a relationship suggested by Fava et al. (1993); Equation (12.1) can then be expressed as:

$$\textit{Depletion}\ (yr^{-1}) = \sum_i \frac{\textit{Deaccumulation, kg.yr}^{-1}}{(\textit{Reserve}_i, kg)^2} \times \textit{Extraction}_i, kg \qquad (12.2)$$

Guinee and Heijungs (1995) proposed equations to calculate the dimensionless abiotic depletion potential (ADP) and biotic depletion potentials (BDP) as:

$$ADP_i, kg_r / kg_i \quad = \quad \frac{Production_i, kg.yr^{-1}}{Production_r, kg.yr^{-1}} \times \left(\frac{Reserve_r, kg}{Reserve_i, kg} \right)^2 \qquad (12.3)$$

$$BDP_i, kg_r / kg_i \quad = \quad \frac{Deaccumulation_i, kg.yr^{-1}}{Deaccumulation_r, kg.yr^{-1}} \times \left(\frac{Reserve_r, kg}{Reserve_i, kg} \right)^2 \qquad (12.4)$$

Where subscripts i and r denote the resource in question and the reference resource, respectively. The aggregation of resources can be made using the equations from Guinee and Heijungs (1995):

$$Equivalent\ abiotic\ use\ (kg_r) = \sum_i ADP_i\ (kg_r.kg_i^{-1}) \times Extraction_i\ (kg) \qquad (12.5)$$

$$Equivalent\ biotic\ use\ (kg_r) = \sum_i BDP_i\ (kg_r.kg_i^{-1}) \times Extraction_i\ (kg) \qquad (12.6)$$

Some studies have used time scale in the consideration of resource depletion equivalency. Lee (1998) defined the Resource Depletion Index (RDI) as a quantitative indicator to compare the depletion condition of various resources. A smaller RDI indicates the possibility of encountering future depletion crises for that resource. The RDI was also used to determine the reference level for resource depletion in a study by Schriefl (2001). RDI is expressed as the maximum extractable years remaining for a resource and is represented as (Lee 1998):

$$RDI\ (Maximum\ extractable\ years) \quad = \quad \frac{Global\ reserves}{Net\ annual\ consumption\ rate} \qquad (12.7)$$

Brentrup et al. (2002) proposed an equation to determine the resource-weighting factor (analogous to resource equivalency factor):

$$Weighting\ factor_i = \frac{Current\ annual\ production_i}{Tolerable\ annual\ production_{i,T}} \qquad (12.8)$$

Where subscript i denotes the resource in question and subscript T denotes the target time period. The tolerable annual production depends on a designated time period. And it can be calculated from the equation:

$$Tolerable\ annual\ production_i = \frac{Global\ recoverable\ reserve_i}{Target\ time\ period_i} \qquad (12.9)$$

This chapter elaborates the carrying capacity estimates for natural fossil fuels, minerals, and water resources. These resources are the basic needs for industrial manufacturing. The biotic resources, loss of biodiversity, and land use are not included. Also, the resources concerned are for primary extraction. For example, bauxite ore available from the earth's crust is considered in resource depletion instead of the production of aluminum ingot. Resource availability is defined according to current extraction practices, e.g., iron ore available from the earth's crust rather than iron available in water bodies, plants, landfills, etc. Resources can be divided into non-renewable resources and renewable resources based on the rate of natural formation (or flow resources and stock resources).

12.2 Carrying Capacity Estimation for Consumption of Fossil Fuels

Fossil fuels are considered to be a non-renewable resource because of their rather slow renewal rate compared to the consumption rate. The concept of carrying capacity is not as apparent for non-renewable resources because some may consider these resources to be non-sustainable by definition. However, in practice, a rate of depletion can be considered as sustainable if it is defined based on the need to find a replacement and to allow sufficient reserves ultimately to remain in place for future generations. This can be done by defining a time horizon required for resource availability that will meet these criteria (Dickinson 1999; Yossapol et al. 2002). The carrying capacity can then be calculated using the reserve-time horizon approach, defined as the maximum allowable yearly consumption of the resource, which is the total quantity of existing accessible reserves divided by a specified time horizon (Equation (12.10)).

$$\textit{Carrying Capacity (kg/yr)} = \frac{\textit{Existing reserves (kg)}}{\textit{Time horizon (year)}} \qquad (12.10)$$

The existing or accessible reserves, i.e., the reserve base, is that part of an identified resource that meets specified minimum physical and chemical criteria related to current mining and extracting practices. The reserve base includes those resources that are economic and sub-economic (USGS 2002).

The advantage of using the reserve-time horizon approach is that the carrying capacity of a resource can be adjusted by varying either the existing reserves or an appropriate time horizon. Information on existing reserves can be updated when necessary as a result of new resource discoveries. The use of an appropriate time horizon may reflect the perception of the sustainability concept or other concepts that may vary over time as a function of world events.

In this research, the time horizon of 200 years is used for carrying capacity estimate for consumption of fossil fuels (as well as minerals). This 200 year time horizon is selected on the basis that existing coal reserves are sufficient to last for over 210 years at the current rate of extraction. Coal is used as the reference resource because it is the most used and the most abundant fossil fuel in the world's current mix of power sources

(WCI 2000). This time horizon does not take into account that more coal reserves may be found by on-going exploration and may become more accessible as further improvements are made in mining technology. If more supporting literature on the time horizon for resources become available, a more appropriate time scale may be examined and applied for individual resources. Hence, the carrying capacity may be recalculated.

This reserve-time horizon technique is the reciprocal of the RDI discussed earlier; but the number of maximum extractable years (RDI) will be fixed to the specified time horizon and the net annual consumption is defined as the carrying capacity instead. Note that when using Equation (12.10) with the time horizon held constant, the carrying capacity decreases as reserves are depleted each year to assure a continuous 200 year-supply. In other words, the depleting reserves due to continuous use will result in decreasing the level of the carrying capacity for the subsequent years. Carrying capacity estimates for fossil fuels using Equation (12.10) are shown in Table 12.2. Lignite is used in power generation but it is usually consumed at or close to the mining sites due to its low quality and the relative inefficiency of transportation costs relative to energy content.

Table 12.2 Carrying Capacity for Consumption of Fossil Fuels

Fuel	Reserves (kg)		Reference	Carrying Capacity (kg)	
	U.S.	World		U.S.	World
Coal	2.51E+14	1.00E+15	WEC 1998	1.26E+12	5.00E+12
Crude oil	3.02E+12	1.36E+14	EIA 2000a	1.51E+10	6.80E+11
Lignite	3.39E+13	1.99E+14	WEC 1998	1.70E+11	9.95E+11
Natural gas	3.33E+12	1.02E+14	EIA 2000a	1.67E+10	5.10E+11

The technique for the carrying capacity estimation is totally based on the time horizon. Changing the time horizon will affect the carrying capacity and will affect the outcome of the environmental performance assessed by the STM. Should a time horizon be altered, the carrying capacity will vary as shown in Figure 12.1.

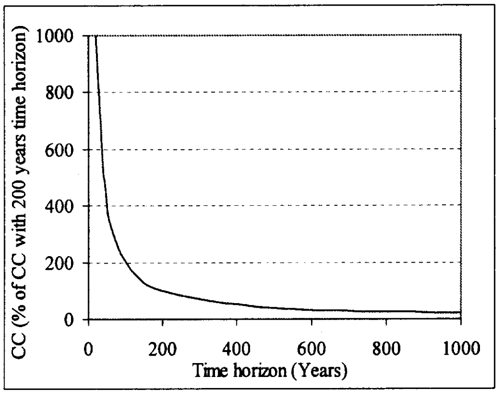

Figure 12.1 Change in carrying capacity due to change in time horizon.

The figure shows that when the time horizon is varied from 10 to 1,000 years, the carrying capacity is also lowered by two orders of magnitude. However, the carrying capacity will be less than one order of magnitude should the time horizon be greater than 200 years (as assigned here) to 2,000 years. The effect of varying the carrying capacity estimates to the overall performance of the STM is further discussed in Chapter 14.

12.3 Carrying Capacity Estimation for Consumption of Minerals

Minerals are also considered as a non-renewable resource. The environmental concern associated with mining, refining, and recycling technologies for minerals is an issue that should also be considered. Environmental problems such as severe impacts to the environment and human health may result in the reduction of mineral production and use.

Recovery is considered as a secondary source for metals (refining of virgin metals is the primary source). Secondary sources for some metals, such as lead, may be as high as 67% of the total production. It should be noted that the carrying capacity estimated by Equation (12.10) is for virgin minerals/fuels, so the application of the recycling rate may alter the carrying capacity figure. However, documented recycling rates are available for only a few minerals/metals. Therefore, the carrying capacity for consumption of minerals in this research is related to virgin minerals only. The carrying capacity then can also be estimated using Equation (12.10) as presented in Table 12.3 for minerals with reserve base figures available in a USGS database (USGS 2002).

Table 12.3 Carrying Capacity for Consumption of Selected Minerals

Mineral	Reserve Base			Carrying Capacity		
	U.S.	World	Unit	U.S.	World	Unit
Aluminum (A)				3.40E+07	2.89E+10	kg/yr
Antimony	9.00E+04	3.20E+06	tons	4.50E+05	1.60E+07	kg/yr
Arsenic	8.00E+04	1.05E+06	tons	4.00E+05	5.25E+06	kg/yr
Barite	6.00E+07	5.50E+08	tons	3.00E+08	2.75E+09	kg/yr
Bauxite	4.00E+07	3.40E+10	tons	2.00E+08	1.70E+11	kg/yr
Bismuth	1.40E+04	6.90E+05	tons	7.00E+04	3.45E+06	kg/yr
Boron	8.00E+07	4.70E+08	tons	4.00E+08	2.35E+09	kg/yr
Bromine	1.10E+07	N/A	tons	5.50E+07		kg/yr
Cadmium	2.70E+05	1.20E+06	tons	1.35E+06	6.00E+06	kg/yr
Chromium	1.00E+07	7.60E+09	tons	5.00E+07	3.80E+10	kg/yr
Cobalt	8.60E+05	1.00E+07	tons	4.30E+06	5.00E+07	kg/yr
Columbium (Niobium) and Tantalum						
Columbium (Niobium)	0.00E+00	5.70E+06	tons	0.00E+00	2.85E+07	kg/yr
Tantalum	0.00E+00	1.20E+05	tons	0.00E+00	6.00E+05	kg/yr
Copper	9.00E+07	6.50E+08	tons	4.50E+08	3.25E+09	kg/yr
Diamond (Industrial)	2.30E+08	1.20E+09	carats	1.15E+06	6.00E+06	carats/yr
Diatomite	5.00E+08	N/A	tons	2.50E+09		kg/yr
Fluorspar	6.00E+06	6.40E+08	tons	3.00E+07	3.20E+09	kg/yr
Garnet	2.50E+07	N/A	tons	1.25E+08		kg/yr
Germanium	5.00E+02	N/A	tons	2.50E+03		kg/yr
Gold	6.00E+03	7.80E+04	tons	3.00E+04	3.90E+05	kg/yr
Graphite	1.00E+06	3.60E+08	tons	5.00E+06	1.80E+09	kg/yr
Helium	8.90E+09	2.50E+10	m^3	4.45E+07	1.25E+08	m^3/yr
Indium	6.00E+02	5.70E+03	tons	3.00E+03	2.85E+04	kg/yr
Iodine	5.50E+02	2.70E+04	tons	2.75E+03	1.35E+05	kg/yr
Iron Ore	4.60E+09	1.60E+11	tons	2.30E+10	8.00E+11	kg/yr
Lead	2.00E+07	1.30E+08	tons	1.00E+08	6.50E+08	kg/yr
Lithium	4.10E+05	9.40E+06	tons	2.05E+06	4.70E+07	kg/yr
Magnesium Compounds	1.50E+07	2.50E+09	tons	7.50E+07	1.25E+10	kg/yr
Manganese	8.20E+07	5.00E+09	tons	4.10E+08	2.50E+10	kg/yr
Mercury	7.00E+03	2.40E+05	tons	3.50E+04	1.20E+06	kg/yr
Molybdenum	5.40E+03	1.10E+04	tons	2.70E+04	5.50E+04	kg/yr
Nickel	2.50E+06	1.60E+08	tons	1.25E+07	8.00E+08	kg/yr
Peat	6.40E+09	2.00E+11	tons	3.20E+10	1.00E+12	kg/yr
Perlite	2.00E+08	2.00E+09	tons	1.00E+09	1.00E+10	kg/yr
Phosphate Rock	4.00E+09	4.70E+10	tons	2.00E+10	2.35E+11	kg/yr
Platinum-Group Metals	2.20E+03	7.30E+04	tons	1.10E+04	3.65E+05	kg/yr
Potash	3.00E+08	1.70E+10	tons	1.50E+09	8.50E+10	kg/yr
Rare Earths –Rare Earth oxide	1.40E+07	1.10E+08	tons	7.00E+07	5.50E+08	kg/yr
Yittrium	1.30E+05	5.60E+05	tons	6.50E+05	2.80E+06	kg/yr
Rhenium	4.50E+03	1.00E+04	tons	2.25E+04	5.00E+04	kg/yr
Selenium and Tellurium						
Selenium	1.90E+04	1.30E+05	tons	9.50E+04	6.50E+05	kg/yr
Tellurium	6.00E+03	3.80E+04	tons	3.00E+04	1.90E+05	kg/yr
Silver	7.50E+04	4.30E+05	tons	3.75E+05	2.15E+06	kg/yr
Soda Ash	3.90E+10	4.00E+10	tons	1.95E+11	2.00E+11	kg/yr
Strontium	1.40E+06	1.20E+07	tons	7.00E+06	6.00E+07	kg/yr
Sulfur	2.30E+08	3.50E+09	tons	1.15E+09	1.75E+10	kg/yr
Talc	5.40E+08	N/A	tons	2.70E+09	N/A	kg/yr
Thorium (B)	3.00E+05	1.40E+06	tons	1.50E+06	7.00E+06	kg/yr
Tin	4.00E+04	1.20E+07	tons	2.00E+05	6.00E+07	kg/yr
Titanium – Titanium mineral	5.90E+07	4.70E+08	tons	2.95E+08	2.35E+09	kg/yr
Titanium and Titanium dioxide	1.54E+06	4.30E+07	tons	7.70E+06	2.15E+08	kg/yr
Tungsten	2.00E+05	3.10E+06	tons	1.00E+06	1.55E+07	kg/yr
Uranium	5.64E+05	3.00E+06	tons	2.82E+06	1.50E+07	kg/yr
Vanadium	4.00E+06	2.70E+07	tons	2.00E+07	1.35E+08	kg/yr
Vermiculite	1.00E+08	2.00E+08	tons	5.00E+08	1.00E+09	kg/yr
Zinc	8.00E+07	4.40E+08	tons	4.00E+08	2.20E+09	kg/yr
Zirconium and Hafnium						
Zirconium (ZrO_2)	5.30E+06	6.50E+07	tons	2.65E+07	3.25E+08	kg/yr
Hafnium (HfO_2)	9.70E+04	1.00E+06	tons	4.85E+05	5.00E+06	kg/yr

Note: (A) Estimated assuming 17% conversion of bauxite to aluminum
(B) Equivalent to thorium oxide, ThO_2

12.4 Carrying Capacity Estimation for Consumption of Freshwater

The carrying capacity for freshwater consumption considered here can be defined as "offstream use" or "withdrawal use", i.e., water removed from the ground or diverted from a surface-water source for use. The use of water includes any sector of value-added activities such as public-supply, domestic, commercial, irrigation, industrial, mining, and livestock uses. The issue of pollutant discharge or the contamination of water is addressed in eutrophication, human toxicity, and eco-toxicity impacts.

The main source of freshwater recharge is precipitation (rainfall and snow) as it contributes to potential renewable water resources. Despite some annual fluctuations, the distribution of water has not changed significantly over the past few thousand years (Vos 1997-1998). Although there is more than adequate water available to meet the annual requirement globally, water crises occur because the available water is not evenly distributed. More importantly, degradation of quality through human activities has reduced the amount of usable water that is available. In addition, some parts of the world receive very little runoff, whereas other parts receive greater amounts. Spatially, 80% of the precipitation is concentrated in the northern and equatorial zones, particularly in wet places where relatively small populations live (Vos 1997-1998).

The average annual flow of rivers and recharge of groundwater generated from precipitation is termed as the annual Internal Renewable Water Resources (IRWR) (WRI 1999; FAO 2003). The Food and Agriculture Organization of the UN (FAO 2003) estimates that the IRWR is 43,764 km^3/yr for the entire world. At the continental level, the Americas have the largest share of the world's total freshwater resources with 45 percent, followed by Asia with 28 percent, Europe with 15.5 percent and Africa with 9 percent (FAO 2003). The total withdrawals in 1997 were about 8% of IRWR (WRI

1999). The agricultural sector accounted for 69% of total withdrawals, while the domestic and industrial sectors accounted for 8% and 23% respectively.

The IRWR of the U.S. is 2,000 km^3/yr (FAO 2003). This amount includes both surface water flow and groundwater recharge and it is approximately 34% of the natural water generated from precipitation (5,800 km^3/yr) with the remaining 66% being the portion involved in evaporation and transpiration (FAO 2003). The IRWR can be assumed here as the water availability that is left after evaporation and transpiration (evapotranspiration) by natural vegetation and rain-fed agriculture have taken their share. Evapotranspiration is defined as the combination of direct evaporation from a wet surface and water consumption or transpiration by vegetation. Baumgartner and Reichel (1975) estimated the average discharge that is left after evapotranspiration to be 36% of the water generated from precipitation. Shiklomanov (1993) (as cited in Mook and de Vries 2001) also reported a similar portion at 39%. These typical proportions are comparable to that of the U.S. Therefore, it is reasonable to define the carrying capacity as that which is left after natural take-out from the IRWR.

For groundwater, the disadvantage is that its renewal is slow compared to that of surface water, while an advantage is that groundwater undergoes filtration through the porous aquifer media, which potentially improves its quality. The storage of groundwater reservoirs guarantees that freshwater is continuously available even in regions with high variations in annual precipitation. Although global groundwater use seems to be small compared to that of surface water, the importance is reversed if only drinking water is considered (Lead Intl. Inc. 2000). This is due to groundwater's quality, limited vulnerability to contamination, and availability.

In groundwater management, sustainability is achieved when the withdrawal rate is no greater than the long-term natural recharge rate. For the U.S., groundwater recharge is estimated at 1,300 km^3/yr (FAO 2003). Therefore, this overall average annual groundwater recharge rate for the U.S. can be defined as the groundwater carrying capacity. The remaining portion of IRWR is surface water flow of 700 km^3/yr and it is the surface water carrying capacity. Likewise, carrying capacity for other countries can be estimated in the same manner. Table 12.4 presents the annual recharge rate for groundwater and surface water for some selected countries.

Table 12.4 Annual Water Recharge Rate in Selected Countries
(Source: FAO 2003)

Country	Annual Groundwater Recharge (km^3/yr)	Annual Surface Water Recharge (km^3/yr)	Total Annual Fresh Water Recharge Rate (km^3/yr)	Country	Annual Groundwater Recharge (km^3/yr)	Annual Surface Water Recharge (km^3/yr)	Total Annual Fresh Water Recharge Rate (km^3/yr)
World	11,301.0	32,463.0	43,764.0	*Oceania*			
North America				Australia	72.0	420.0	492.0
Canada	370.0	2,480.0	2,850.0	New Zealand	198.0	129.0	327.0
United States	1,300.0	700.0	2,000.0	*Africa*			
Central America				Algeria	1.7	12.2	13.9
Mexico	139.0	270.0	409.0	Ethiopia	40.0	70.0	110.0
South America				Kenya	3.0	17.2	20.2
Argentina	128.0	148.0	276.0	Nigeria	87.0	134.0	221.0
Brasil	1,874.0	3,544.0	5,418.0	*Europe*			
Colombia	510.0	1,602.0	2,112.0	Belgium	0.9	11.1	12.0
Asia				France	100.0	78.5	178.5
China	829.0	1,983.0	2,812.0	Germany	45.7	61.3	107.0
India	418.5	842.0	1,260.5	Netherlands	4.5	5.5	10.0
Japan	27.0	403.0	430.0	Norway	96.0	286.0	382.0
Thailand	41.9	168.1	210.0	United Kingdom	9.8	135.2	145.0
Malaysia	64.0	516.0	580.0				

Because the precipitation is not evenly distributed, the recharge also varies from area to area. The locally specific carrying capacity can be estimated assuming that the recharge rate of the carrying capacity is proportional to the precipitation. The equation for calculating the locally specific carrying capacity then can be written as in Equation (12.11). Applying this equation, the locally specific (state level) carrying capacity for freshwater consumption can be calculated as presented in Table 12.5.

$$\textit{Local CC (km}^3\textit{/yr)} = \textit{Locally specific factor} \times \textit{U.S. Carrying capacity (km}^3\textit{/yr)} \qquad (12.11)$$

Table 12.5 Locally Specific Carrying Capacity for Freshwater Consumption
(Source of precipitation intensity data: NOAA 2002)

State	Precipitation (inches/yr)	Locally specific factor	Carrying Capacity (km³/yr)		State	Precipitation (inches/yr)	Locally specific factor	Carrying Capacity (km³/yr)	
			Surface water	Groundwater				Surface water	Groundwater
AK	15.9	0.46	321	596	MT	15.3	0.44	309	573
AL	53.7	1.55	1083	2012	NC	49.4	1.42	997	1851
AR	49.2	1.42	993	1843	ND	17.4	0.50	351	652
AZ	12.7	0.37	256	476	NE	22.8	0.66	460	854
CA	22.3	0.64	450	835	NH	42.4	1.22	855	1588
CO	15.9	0.46	321	596	NJ	44.7	1.29	902	1675
CT	45.1	1.30	910	1690	NM	13.4	0.39	270	502
DE	44.3	1.28	894	1660	NV	8.7	0.25	176	326
FL	53.9	1.55	1087	2019	NY	38.9	1.12	785	1457
GA	50	1.44	1009	1873	OH	38	1.10	767	1424
HI	27.5	0.79	555	1030	OK	33.9	0.98	684	1270
IA	32.2	0.93	650	1206	OR	26.8	0.77	541	1004
ID	18.8	0.54	379	704	PA	39.7	1.14	801	1487
IL	37.8	1.09	763	1416	RI	43.1	1.24	869	1615
IN	40	1.15	807	1499	SC	47.9	1.38	966	1795
KS	27.4	0.79	553	1027	SD	18.3	0.53	369	686
KY	47.3	1.36	954	1772	TN	52	1.50	1049	1948
LA	57	1.64	1150	2135	TX	28	0.81	565	1049
MA	42.7	1.23	861	1600	UT	11.5	0.33	232	431
MD	42.8	1.23	863	1603	VA	42.5	1.22	857	1592
ME	42.6	1.23	859	1596	VT	40.5	1.17	817	1517
MI	31.2	0.90	629	1169	WA	37	1.07	746	1386
MN	26	0.75	524	974	WI	31.3	0.90	631	1173
MO	40.8	1.18	823	1529	WV	43.9	1.27	886	1645
MS	54.9	1.58	1107	2057	WY	13.1	0.38	264	491
MT	15.3	0.44	309	573	Average	34.7	1.00	700	1300

12.5 Summary

The carrying capacity for fossil fuels, minerals, and freshwater consumption is estimated for the resource depletion impact (Table 12.2, Table 12.3, and Table 12.5 respectively). For minerals, the carrying capacity is related to virgin minerals only. The carrying capacity estimates for resource depletion are in the category of economic carrying capacity because they are associated with value-added activities.

For non-renewable resources, such as fossil fuels and minerals, the carrying capacity can be determined using the reserve-time horizon technique developed in this research. This technique calculates the maximum allowable yearly consumption of existing resources for a specified time horizon. The carrying capacity decreases as reserves are depleted each year to assure a continuous supply for the specified time horizon. For the carrying capacity estimate in this research, the time horizon of 200 years is used for consumption of fossil fuels and minerals. This time horizon is selected on the basis that existing world coal reserves can last for approximately 200 years at today's consumption rate. Coal is the most used and the most abundant fossil fuel in the world's current mix of power sources. Should supporting literature on the time horizon for resources become available, a more appropriate time scale may be examined and applied. The time horizon use in this technique can be altered to a more appropriate figure depending on the purpose of an LCA study and the judgment of the practitioner. An advantage of applying the reserve-time horizon technique to estimate the carrying capacity is that the existing amount of reserves is flexible. The reserves are depleted each year by human consumption but, on the other hand, they can be expanded due to mining exploration and improved technology.

The carrying capacity for the consumption of renewable resources such as freshwater is simply determined from their replenishment rate. The renewable rate for freshwater, which is calculated as precipitation left after losses and natural uses, is used as the carrying capacity for both groundwater and surface water. The carrying capacity for water consumption is evaluated in terms of water availability only. The carrying capacity of pollutant discharge to water resources is addressed in eutrophication impact, human toxicity, and eco-toxicity impacts. For the U.S., the locally specific carrying capacity estimates are provided in Table 10.5.

The carrying capacity estimate in this research can be refined by updating the information on remaining reserves of non-renewable resources. The carrying capacity estimate for resource depletion can also be further advanced by inclusion of other resources such as biotic resources, biodiversity, and land use.

CHAPTER 13

APPLICATIONS OF CARRYING CAPACITY

13.1 Overview

The primary objectives of this research are to develop a set of carrying capacity estimates and to apply them to an appropriate environmental performance evaluation metric. This chapter demonstrates how the carrying capacity can be used in conjunction with the STM in three case studies following the LCA framework. First, the environmental performance evaluation of four types of electrical energy generation sources is compared as a case study at the process level. Second, a similar evaluation is made for four basic material production operations as a case study at the supply line level. And the third case study is the evaluation of a household plastic coffee maker as an application at the product level. The third case study is also an attempt to demonstrate the evaluation that takes into account the temporal and spatial variations of the production process.

The environmental performance of the products and processes in case studies is extensively evaluated as well by other frequently used LCIA methods using the same LCI data. These methods are Eco-Indicator 95 (EI95), Eco-Indicator 99 (EI99), Environmental Priority Strategies (EPS), and Environmental Design of Industrial Products (EDIP). The results are compared to those of the STM. Comparative analyses of the environmental performance of the first and the second case studies are performed. The superiority among these sources is identified based on the scores of the STM compared to those of the other LCIA methods. The same analysis is carried out for the second case study of basic material production. A comparative analysis of environmental impact contributing to the single scores is also provided for all three case studies.

The feasibility of applying the carrying capacity evaluation technique and approaches used in this research to evaluate the carrying capacity for Europe as well as other regions is also investigated. The European carrying capacity estimates using the technique and approaches used in this research are compared to EI95 target values, a feasible approach for carrying capacity estimation. And vice versa, EI95 target values can also be modified to become the U.S. target values. These modified U.S. target values are used in the STM as a substitute to the carrying capacity in order to evaluate the environmental performance of the three case studies. The results are compared to those using the carrying capacity estimates developed in this research.

13.2 Case Study I: LCA for Electrical Energy Generation Sources

13.2.1 LCA for Electrical Energy Generation Sources Using the STM

A question concerning energy resources that is frequently raised in debate is what would be the best resource alternative for energy generation. Using the STM in conjunction with the U.S. estimates of carrying capacity can provide an answer to this question in terms of environmental performance. For this case study, electrical energy generation sources are selected as alternatives for comparison. The sources taken into account are electrical energy generation from coal, gas, lignite, and oil.

Two major energy generation sources that are not included in this evaluation are hydropower and nuclear power. This is because it is not possible or appropriate at this stage to implement environmental performance assessments of sources for which the environmental concerns are dominated by impacts for which carrying capacities have not yet been evaluated. For hydropower, it is well known that land use and loss of biodiversity are the major issues to be considered due to the construction of dams. For nuclear power, it is anticipated that the major issue is the management of nuclear wastes. Land use, loss of biodiversity, and nuclear waste management impacts are not included in the STM in its current version. Evaluation of these two sources without their major impacts would result in a misleading and biased comparison. Any comparison should be made on an unbiased basis with comparable data. Firstly, the problems, the difficulties, and the limitations of implementing the environmental performance assessment should (always) be considered. Furthermore, other assumptions that have been made should also be stated in the interpretation of the results.

Coal is defined as the combination of sub-bitumen, bitumen, anthracite, and lignite. The major uses of coal are for electrical generation and use in some heavy industries. Lignite is a lower grade than sub-bitumen and bitumen, meaning that it has lower energy content per unit mass. Lignite is evaluated in this case study because its use in energy generation usually takes place at or close to the mining sites due to its low quality. Furthermore, by considering lignite the differences in environmental performance between high quality coal and low quality coal sources can be compared.

Goal and Scope Definition

The purpose of this evaluation is to analyze the environmental performance of electrical energy generation sources by comparing the use of the following fossil fuels: coal, gas, lignite, and oil. For this purpose, the comparison will be carried out on the basis of the functional unit of 1 kWh_e[1] of electrical generation. The life-cycle stages of the energy generation process in this evaluation begin with the extraction of fossil fuels and ends with the emissions of wastes to environmental media. Figure 13.1 presents the system

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[1] kWh_e = kilowatt-hour of electricity

boundary of the energy generation processes considered in this case study. The result of this evaluation will be used to discuss the applicability of the U.S. estimates of carrying capacity as the baseline reference for the STM, an environmental performance metric. Although, the relative ranking of these four fuel sources is given by this evaluation, it cannot suggest the superiority of the use of any particular fuel over the others for all situations.

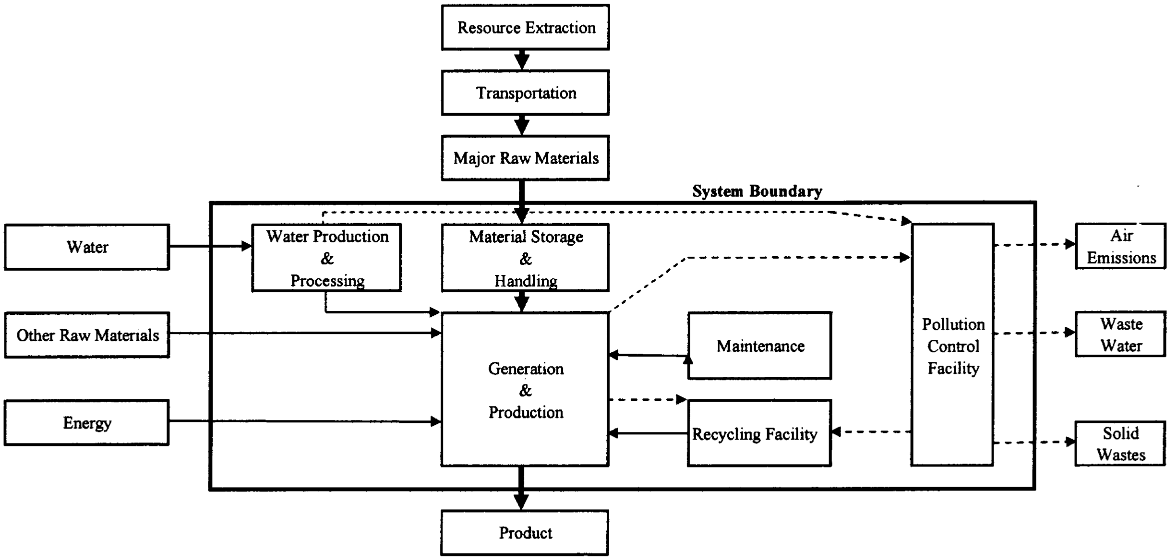

Figure 13.1 System boundary for LCA case studies

Life Cycle Inventory

Life cycle inventory (LCI) data for these energy generation sources were obtained from the IDEMAT 96 database, which is integrated in the SimaPro LCA software (PRe' 2006). It should be noted that this inventory database was gathered from studies conducted mostly in Europe. However, for this research it is assumed that this inventory database is applicable for the same type of process or product in the U.S. The functional unit for this assessment is the electrical generation of 1 kWh_e.

Life Cycle Impact Assessment

The inventory data can be classified into various impact categories (classification step) and the impacts can be aggregated within the same homogeneous category or subcategory using appropriate characterization factors or equivalency factors (characterization step). The next step is to identify the U.S. carrying capacity and the Impact Reference level (I_R) of impact and sub-impact categories for the STM calculations. The I_R associated with the impact categories in electrical generation sources are calculated from the estimated carrying capacity available. The calculations for I_R are carried out using the year 2000 based carrying capacity values as well as economic information.

For other STM parameters, the Value Reference level (V_R) of 100 million dollars per year is arbitrarily selected (using the same approach as in Dickinson et al. 2002) for the calculation of I_R. This V_R represents the revenue of a typical corporate enterprise. As noted by Dickinson et al. (2002), selection of a different V_R will adjust the calculations for productivity indicators equally, but will not change the Eco-Efficiency (EE). The calculation for I_R is based on U.S. gross domestic product (GDP) and global gross product (GGP) for the year 2000 (base year for calculation) which are $9,810 Billion and $31,500 Billion respectively (World Bank 2003). The calculations are carried out on a yearly basis. This long-term evaluation is relevant to the LCA context.

When the Impact Reference levels (I_R) are available, EI_{PR}, RP, and EE can be calculated. Table 13.1 is the summary of results calculated for all four electrical energy generation sources. The price of energy used in the calculations is the average retail price in the U.S. of $0.07 per kWh_e for all sources (EIA 2000b). The calculations are carried out using the equations from the STM methodology (Appendix B).

Table 13.1 STM Indicators for Electrical Energy Generation Sources

	Environmental impact, EI_{PR} $(kWh_e/yr)^{-1}$				Eco-Efficiency (%)			
	Coal	Gas	Lignite	Oil	Coal	Gas	Lignite	Oil
Global warming	3.6E-09	2.2E-09	4.3E-09	3.0E-09	1.9E+01	3.2E+01	1.6E+01	2.3E+01
Stratospheric ozone depletion	0	0	0	0	-	-	-	-
Acidification	8.7E-09	0	7.0E-09	1.0E-08	8.0E+00	-	1.0E+01	6.9E+00
Eutrophication	8.0E-10	5.0E-10	6.7E-10	5.8E-10	8.8E+01	1.4E+02	1.0E+02	1.2E+02
Photochemical smog	9.5E-11	6.3E-11	7.9E-11	6.6E-11	7.4E+02	1.1E+03	8.9E+02	1.1E+03
Human toxicity	9.3E-10	2.3E-10	9.4E-10	6.0E-10	1.5E+02	5.0E+02	1.5E+02	1.2E+02
Eco-toxicity	6.3E-11	1.4E-11	6.2E-11	6.2E-11	2.0E+03	5.7E+03	1.8E+03	1.5E+03
Resource depletion	9.6E-09	4.6E-07	2.1E-07	5.7E-07	7.3E+00	1.5E-01	3.4E-01	1.2E-01
Overall	2.4E-08	4.6E-07	2.2E-07	5.9E-07	2.9E+00	1.5E-01	3.2E-01	1.2E-01
Rank	1	3	2	4	1	3	2	4
Resource productivity, RP (kWh_e/yr)	4.2E+07	2.2E+06	4.5E+06	1.7E+06				
Value productivity, VP, (\$/yr)	2.9E+06	1.5E+05	3.2E+05	1.2E+05				

Interpretation

An interpretation can be presented using the STM results. As discussed in Chapter 2 and Chapter 4, the STM can be used to assess environmental performance and provide indicators related to sustainability. This method can be used for all stages of the entire life cycle following the LCA framework. The key function of the STM is to link the economic value with the environmental carrying capacity. The STM provides five indicators for environmental performance (Dickinson et al. 2002): Environmental Impact (EI), Resource Productivity (RP), Service Productivity (SP), Value Productivity (VP), and Eco-Efficiency (EE). Environmental impacts can be interpreted in terms of a single indicator EI_{PR}, the environmental impact per unit of production. In this case, EI_{PR} has the unit of $(kWh_e/yr)^{-1}$. EI_{PR} is the aggregate of environmental impacts normalized by using Impact Reference levels (I_R) that relate impact to economic value and sustainability. I_R is an essential component of the STM and is based on the carrying capacity estimates.

The relative indicators for productivity (RP, SP, VP) can be determined from EI_{PR} (see Appendix B). These relative indicators provide a basis for comparing environmental performance, i.e., a better alternative is the one that yields a higher productivity. The practical absolute indicator for sustainability is EE, which can be calculated from RP or VP. The criterion for sustainability is EE ≥ 100%. A product with EE > 100% indicates less impact than the sustainable level and vice versa for EE <100%. EE can also be used to interpret the degree of sustainability of a product or service (Table 4.2, Chapter 4). By comparing the sustainability of the electrical energy generation sources using overall EE, the results from Table 13.1 show that all of these sources are much lower than the sustainability level (EE for coal 2.9%, gas 0.15%, lignite 0.32%, and oil 0.12%).

By using the overall EE as the criterion, the energy sources can be ranked from the largest EE to the smallest EE as: coal, lignite, gas, and oil (Figure 13.2). Even though the criterion for sustainability is EE ≥ 100%, the result of an alternative for which the overall EE < 100% does not necessarily indicate that every impact category does not meet the sustainability level. The sustainability levels expressed in terms of EE of impact categories must be determined individually. As seen in Table 13.1, there are only a few impact categories that do not meet the sustainability criterion while the remaining are higher than the cut-off level (note that the Eco-Efficiency from the impact that consists of multiple subcategories is obtained from the most critical one). By using the individual Eco-Efficiency values, one can identify the impact categories and subcategories that should be addressed in terms of improving its environmental performance.

The consumption of raw materials is the worst or most limiting impact (shaded cells in Table 13.1) for all of the sources evaluated. The EI_{PR} for resource consumption of each source are 9.6×10^{-9} for coal, 4.6×10^{-7} for gas, 2.2×10^{-7} for lignite, and 5.9×10^{-7} for oil. These EI_{PR} values indicate that oil is the scarcest fuel while coal is the most abundant fuel in the U.S.

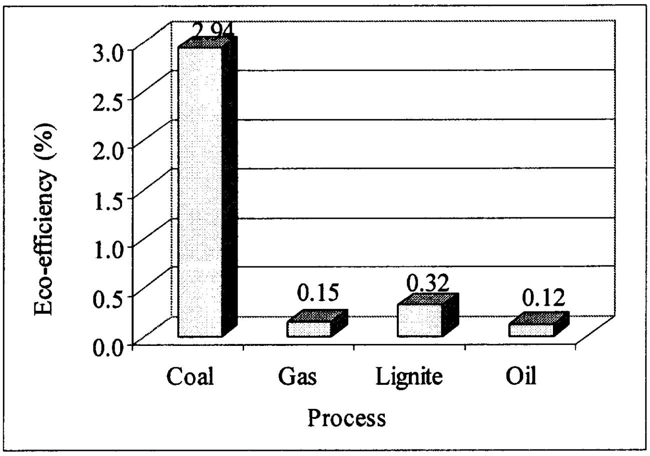

Figure 13.2 Eco-Efficiency (EE) of four electrical energy generation sources.

Global warming impact is not the major contributor to the environmental impact (EI_{PR}) for any of the sources and none of the sources has an impact on ozone depletion (no ozone depleting substance emission). The major contributor to the overall environmental impact is resource depletion for all sources (40.2% for coal, 99.4% for gas, 94.1% for lignite, and 97.5% for oil). Overall, resource depletion and acidification are the two most significant impacts for all four sources. It is worth noting here that human toxicity and eco-toxicity do not show significantly in the assessment. This may be due to the limitation of the carrying capacity estimates for both impact categories in this research. This is because there is still a small number of carrying capacity estimates for the chemicals for both impacts (89 for human toxicity and 131 chemicals for eco-toxicity). In the inventory, there may be some chemicals emitted significantly and the impact is not observed due to the absence of their carrying capacity values. This is surely a point that needs to be improved in future work.

By using the STM, the environmental performance of the four energy generation sources thus can be ranked. This environmental performance assessment is carried out following the LCA framework. In the following subsection, the ranking of these four energy generation sources using the STM Eco-Efficiency indicator (EE) is compared to the rankings assessed by EI95, EI99, EPS, and EDIP.

13.2.2 LCA for Electrical Energy Generation Sources Using Other LCIA Methods

Goal and Scope Definition

The LCA for the case study of energy generation sources is also carried out using four other different LCIA methods: EI95, EI99, EPS, and EDIP. The purpose of this evaluation is to compare the implementation of these LCIA methods with that of the STM. The comparative analysis is to investigate the superiority among the energy generation sources in terms of environmental performance. The environmental impact contributors to the overall single-score indicators are discussed. Some characteristics of the STM compared to four LCIA methods are highlighted and discussed.

Life Cycle Inventory

The LCI data are the same as the ones used in the STM implementation (obtained from the SimaPro LCA Software). The functional unit is the energy generation of 1 kWh_e.

Life Cycle Impact Assessment

The LCIA using four different methods is evaluated in the SimaPro LCA software, in which the four LCIA methods are readily available. The methodologies of these LCIA exercises have been reviewed in Chapter 2. The results obtained from the evaluations are presented in matrix form showing the single scores of individual impact categories. The single scores for energy generation sources are displayed in Table 13.2 to Table 13.5 for the assessment using EI95, EI99, EPS, and EDIP, respectively.

Interpretation

The results are interpreted, compared, and discussed in the following subsection.

Table 13.2 Eco-Indicator 95 Scores for Energy Generation Sources

Eco-Indicator 95 score	Type of energy generation			
	Coal	Gas	Lignite	Oil
Weighted indicator (millipoints/kWh$_e$)				
Greenhouse	2.3E-03	8.6E-04	7.9E-03	1.4E-03
Ozone layer	0	0	0	0
Acidification	8.2E-03	9.2E-04	2.5E-02	6.8E-03
Eutrophication	6.1E-04	2.5E-04	1.15E-03	3.0E-04
Heavy metals	0	0	0	0
Carcinogens	0	0	0	0
Winter smog	3.8E-03	0	1.2E-02	3.5E-03
Summer smog	6.9E-06	2.6E-05	2.0E-05	4.9E-06
Pesticides	0	0	0	0
Energy resources	0	0	0	0
Solid waste	0	0	0	0
Crude fuel acquisition	5.7E-03	3.4E-03	4.9E-03	2.5E-03
Overall	2.1E-02	5.4E-03	5.1E-02	1.4E-02
Rank	3	1	4	2

Table 13.3 Eco-Indicator 99 Scores for Energy Generation Sources

Eco-Indicator 99 score	Type of energy generation			
	Coal	Gas	Lignite	Oil
Weighted indicator (millipoints/kWh$_e$)				
Carcinogens	0	0	0	0
Respiratory damage-organics	3.1E-09	1.2E-08	9.0E-09	2.2E-09
Respiratory damage-inorganics	1.5E-04	2.6E-05	4.1E-04	1.0E-04
Climate change	5.0E-05	1.9E-05	1.7E-04	3.0E-05
Radiation	0	0	0	0
Ozone layer	0	0	0	0
Ecotoxicity	0	0	0	0
Acidification/Eutrophication	2.1E-05	6.6E-06	5.6E-05	1.3E-05
Land use	0	0	0	0
Minerals	0	0	0	0
Fossil fuel use & crude fuel acquisition	4.9E-05	2.9E-04	9.7E-05	4.7E-04
Overall	2.7E-04	3.4E-04	7.3E-04	6.2E-04
Rank	1	2	4	3

Table 13.4 EPS Scores for Energy Generation Sources

EPS 2000 score	Type of energy generation			
	Coal	Gas	Lignite	Oil
Weighted indicator (ELU/kWh$_e$)				
Life expectancy	1.3E-03	3.4E-04	4.1E-03	8.0E-04
Severe mobility	3.8E-04	1.6E-04	1.3E-03	2.1E-04
Morbidity	8.8E-05	3.0E-05	3.0E-04	5.5E-05
Severe nuisance	0	0	0	0
Nuisance	5.3E-05	3.7E-06	1.6E-04	4.6E-05
Crop growth capacity	5.2E-06	2.1E-06	1.4E-05	2.7E-06
Wood growth capacity	-2.3E-05	-8.8E-06	-7.5E-05	-1.3E-05
Fish and meat production	-1.2E-06	-5.2E-07	-2.9E-06	-5.8E-07
Soil acidification	1.5E-06	1.7E-07	4.4E-06	1.2E-06
Production capacity from irrigation water	0	0	0	0
Production capacity from drinking water	0	0	0	0
Depletion of reserves	0	0	0	0
Species extinction	1.5E-05	6.4E-06	5.1E-05	8.2E-06
Crude fuel acquisition	3.4E-04	2.0E-03	2.9E-03	1.2E-03
Overall	2.2E-03	2.5E-03	8.7E-03	2.3E-03
Rank	1	3	4	2

Table 13.5 EDIP Scores for Energy Generation Sources

EDIP score	Type of energy generation			
	Coal	Gas	Lignite	Oil
Weighted indicator (Person-equiv./kWh$_e$)				
Global warming	1.8E-06	6.8E-07	6.2E-06	1.1E-06
Ozone depletion	0	0	0	0
Acidification	9.7E-07	1.1E-07	2.9E-06	8.0E-07
Eutrophication	2.2E-07	8.5E-08	5.3E-07	1.1E-07
Photochemical smog	2.2E-08	4.2E-09	6.5E-08	2.4E-09
Ecotoxicity water chronic	0	0	0	0
Ecotoxicity water acute	0	0	0	0
Ecotoxicity soil chronic	0	0	0	0
Human toxicity air	3.1E-08	8.4E-10	9.9E-08	2.7E-08
Human toxicity water	0	0	0	0
Human toxicity soil	0	0	0	0
Bulk waste	0	0	0	0
Hazardous waste	0	0	0	0
Radioactive waste	0	0	0	0
Slags/ashes	0	0	0	0
Resources	0	0	0	0
Crude fuel acquisition	0	0	0	0
Overall	3.1E-06	8.8E-07	9.8E-06	2.0E-06
Rank	3	1	4	2

13.2.3 Comparison and Discussion

Method Comparison - Single Scores

The single scores vary from method to method. The Eco-Indicator point (or millipoint) represents the aggregated impact scores of EI95 and EI99. EDIP presents the weighted environmental impacts in the unit of person-equivalent, i.e., impact per person per year. The EPS offers the results in ELU (environmental load units), which is more physically meaningful. ELU is equivalent to the environmental cost (in Euro currency) because the EPS impact assessment is based on the cost of damage restoration or the prevention of environmental impacts.

The results from the EPS (Table 13.4) reveal that all four energy generation sources have environmental costs about an order of magnitude lower than their economic value. The ELUs of these sources are 2.2×10^{-3}, 2.5×10^{-3}, 8.7×10^{-3}, and 2.3×10^{-3} €/kWh$_e$ for coal, gas, lignite, and oil respectively compared to the economic value (sell price) of 0.07 \$/kWhe[1] of electricity.

The STM, however, interprets the environmental performance of a system in several ways. Because the STM calculations are based on the economic value and the sustainable level of the system in question, its parameters can be used for multiple purposes. The aggregated environmental impact is interpreted in a dimensionless unit Environmental Impact (EI), which is the environmental impact per unit production rate related to the sustainability level of the system. Resource Productivity (RP) and Value Productivity (VP) are the relative indicators expressing the rate of production and value provided, respectively, per unit of environmental impact. And Eco-Efficiency (EE) is a practical absolute indicator for sustainability.

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[1] It is conveniently assumed that 1€ = 1\$ (2003).

Method Comparison-Impact categories

STM implementation as well as other LCIA methods basically cover major impact categories suggested by SETAC (Table 13.6), which are resource depletion, land use, global warming, stratospheric ozone depletion, photochemical ozone formation, human toxicity, and toxicity to ecosystems (acidification, eutrophication, ecosystem quality). The STM implementation illustrated in this chapter does not cover the land use impact. However, it should be noted that the impact categories selected for the STM do not represent an intrinsic limitation of the STM. The number of impact categories that can be applied to the STM is basically unlimited. The current limitation is rather due to the availability of reference levels (or carrying capacity values). In this research, an appropriate approach to determine the carrying capacity for land use is not yet determined. It is also considered that the omission of land use impact in the STM implementation will not give a result significantly different from the one with land use impact because most of the LCI data do not include the land use associated with products/processes.

However, it is recommended that solid wastes, hazardous wastes, and radioactive wastes impacts should be taken into account in further development of carrying capacity evaluation. This is because the management of these wastes is one of today's primary environmental concerns. These impacts are not evaluated in this research because of the difficulty of determining natural absorption capacity of these wastes. The management capacity (the existing capability to manage solid wastes, hazardous wastes, and radioactive wastes) is considered not appropriate because it is an emission-based capacity, which differs from the carrying capacity concept in this research.

Table 13.6 Impact Categories Included in the STM and Other LCIA Implementations

Default impact categories	STM	EI95	EI99	EPS	EDIP
Input related categories					
Resource depletion	X		X	X	X
Land use			X		X
Output related categories					
Global warming	X	X	X		X
Stratospheric ozone layer depletion	X	X	X		X
Photochemical ozone formation	X	X	X		X
Toxic to human health					
General toxic substances	X			X	X
Carcinogens		X	X		
Substances causing winter smog		X			
Pesticides		X			
Heavy metals		X			
Substances damaging respiratory system			X		
Radiation			X		X
Toxic to ecosystem					
Acidification	X	X	X	X	X
Eutrophication	X	X	X		X
Ecosystem quality	X		X	X	X
Production capability				X	
Solid wastes		X			X
Hazardous wastes					X

EI95 does not identify resource depletion (availability of resource) as an impact. However, the environmental damage due to resource consumption is evaluated in terms of embedded energy for raw material production and fuel acquisition. Another impact that is not included in EI95 is eco-toxicity (other than acidification and eutrophication). EI95 also separates the human health impact into subcategories on an emission-related basis. This may cause the omission of several toxic substances that are not included in any class under subcategories. This procedure is similar to that of EI99, which considers only the toxic substances that are carcinogenic or radioactive substances or cause damage to human respiratory system.

A cursory survey of Table 13.6, EPS seems to consider a number of impact categories. However, some of them are subcategories divided from the human health impact and production capability. The EPS omits a number of common impacts for which their damages cannot be measured in terms of monetary value. These omitted impacts are mostly on a global scale such as climate change and ozone depletion impacts.

EDIP covers most of the listed emission-related impact categories. Although EDIP implementation in the SimaPro separates environmental impacts from resource consumption, the environmental damage associated with resource consumption is evaluated in terms of embedded energy for raw material production and fuel acquisition. A major characteristic of the EDIP is the separation of the weighting calculation from the preceding LCIA steps. This characteristic, the same as the STM, allows an unlimited number of impact categories to be included as long as the characterization factors are prepared. For the STM, however, the carrying capacity value is required in addition to the characterization factors should a new impact category be added.

Result Comparison-Contribution analysis

Table 13.7 is a summary of the primary environmental impact of energy generation sources assessed using different LCIA methods. The contribution analysis of environmental impacts reveals different trends of what would be the major contributors of energy generation sources. Energy generation from gas that shows resource depletion impact is the major impact for all methods but the EDIP. For EDIP, the global warming impact is the major impact for all energy generation sources. Figure 13.3 to Figure 13.7 present the contributions of major impact categories for the environmental performance assessments using the STM as well as the other LCIA methods. It should be noted that the STM implementation is made using the U.S. based carrying capacity and economy while the results from other LCIA methods are based on European conditions.

Table 13.7 Primary Contributing Impact to Total Impact for Energy Generation Sources Assessed from the STM and Other LCIA Methods

Method	Major impact			
	Coal	Gas	Lignite	Oil
STM	Resource depletion	Resource depletion	Resource depletion	Resource depletion
Eco-Indicator 95	Acidification	Fuel acquisition	Acidification	Acidification
Eco-Indicator 99	Respiratory damage	Fuel acquisition	Respiratory damage	Fuel acquisition
EPS 2000	Life expectancy	Fuel acquisition	Life expectancy	Fuel acquisition
EDIP	Global warming	Global warming	Global warming	Global warming

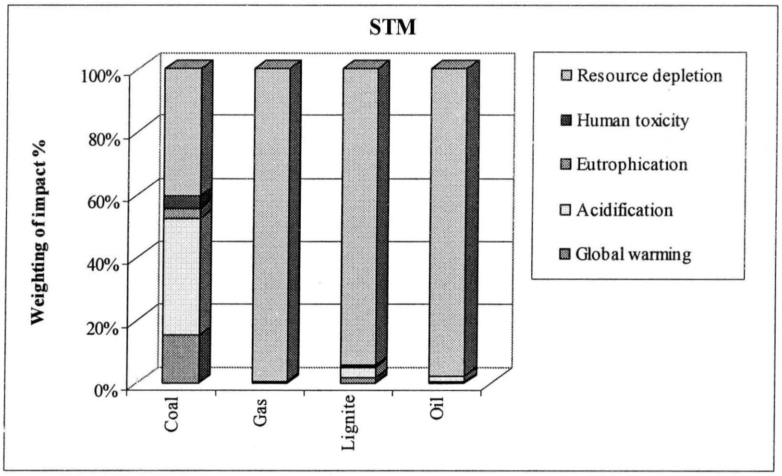

Figure 13.3 Major contributors of energy generation sources assessed by the STM.

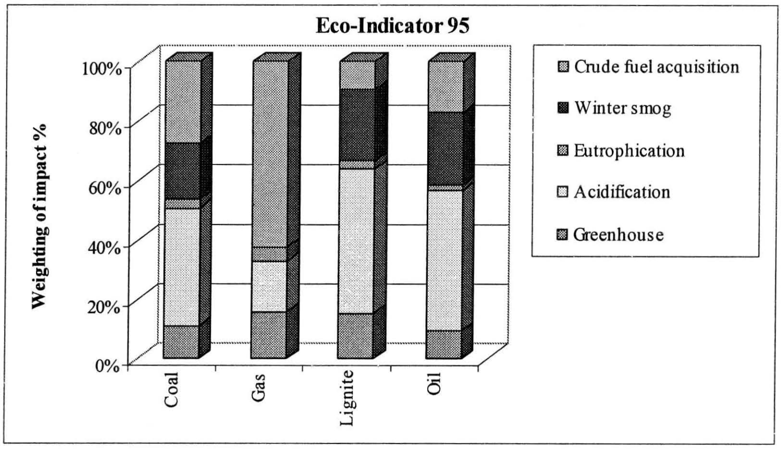

Figure 13.4 Major contributors of energy generation sources assessed by Eco-Indicator 95.

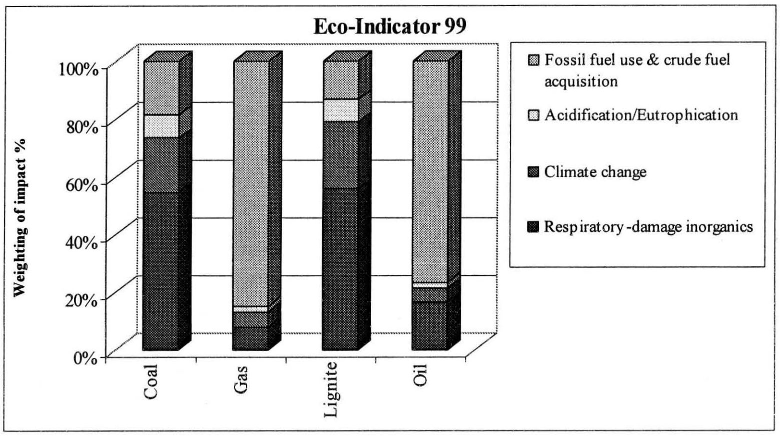

Figure 13.5 Major contributors of energy generation sources assessed by Eco-Indicator 99.

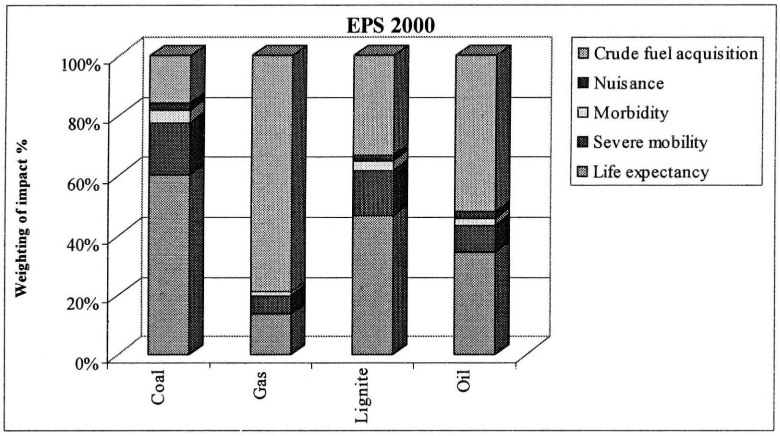

Figure 13.6 Major contributors of energy generation sources assessed by EPS 2000.

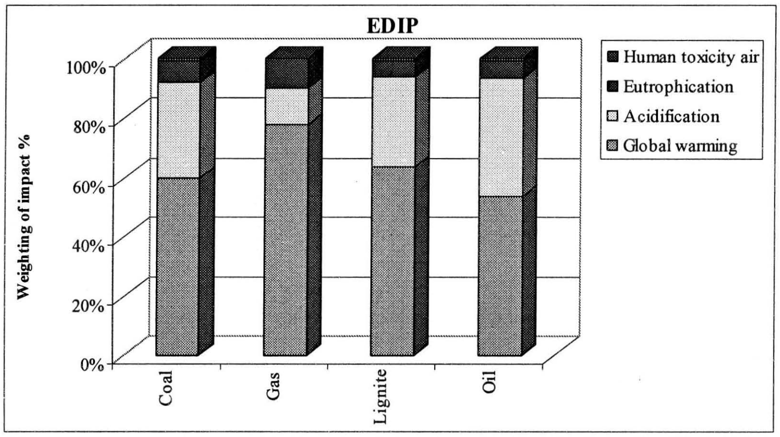

Figure 13.7 Major contributors of energy generation sources assessed by EDIP.

It can be observed from Table 13.7 that human health is the major contributing impact in only a few assessments even though burning of fossil fuel is known to be a major source of anthropogenic mercury emissions all over the world. This discrepancy is due to the absence of mercury emission reported or recorded in the LCI database, which shows only the major emissions, i.e., CO_2, CO, N_2O, NO_X, and hydrocarbons associated with fossil fuels burning for electricity generation. It should also be noted that the carrying capacity for mercury emission (human heath impact) in this research is not estimated because it was outside the boundary set for the study. A further study including mercury emission would help address the quality of the relationship of human health impacts of this type on sustainable of the process.

Result Comparison-Ranking of alternatives

The ranking of alternatives using EE of the STM can be compared to those using single-scores of other LCIA methods (Table 13.8). The trend from EI95 is the same as that of EDIP. The average trend of all five methods is ranked from coal, gas, oil, and lignite (same as EI99). Lignite shows the worst environmental performance among the four energy generation alternatives for all methods except the STM. Comparison of these rankings with the rankings that take into account only the common impact categories (global warming, ozone depletion, acidification, eutrophication, and photochemical ozone formation) indicates that the rankings that take into account only the common impact categories provide a more agreeable trend (Table 13.9). The EPS is not included in this comparison because it does not consider some of these common impacts.

Table 13.8 Ranking in Terms of Environmental Performance of Energy Generation Sources Assessed by the STM and Other LCIA Methods

Method	Environmental performance ranking			
	1	2	3	4
STM	Coal	Lignite	Gas	Oil
Eco-Indicator 95	Gas	Oil	Coal	Lignite
Eco-Indicator 99	Coal	Gas	Oil	Lignite
EPS 2000	Coal	Oil	Gas	Lignite
EDIP	Gas	Oil	Coal	Lignite

Table 13.9 Comparison between the Rankings with All Impact Categories and the Rankings with only Five Common Impact Categories

	Coal	Gas	Lignite	Oil
Rankings with all impacts				
STM	1	3	2	4
EI95	3	1	4	2
EI99	1	2	4	3
EDIP	3	1	4	2
Rankings with only common impacts				
STM	3	1	2	4
EI95	3	1	4	2
EI99	3	1	4	2
EDIP	3	1	4	2

Discussion

This section has demonstrated the application of the carrying capacity in conjunction with the STM to assess the environmental performance related to sustainability for four types of energy generation sources based on environmental impact and economic value. This application is an example of ranking the process level alternatives in terms of environmental performance. However, the alternative ranking for the case study obtained from the STM gives a different trend from the rankings obtained from other LCIA methods. This is may be due to the number of impact categories considered.

A conclusion that can be drawn from the environmental performance assessment for the energy generation sources is that different LCIA methods obtain different ranking trends. Use of a ranking trend to identify the superiority of an alternative over the others should not be made without indicating the method used and its associated assumptions, especially the impact categories considered. Otherwise, the results can be misleading to those who are not familiar with the environmental performance evaluation process.

One aspect that can be pointed out here is that environmental performance evaluation using the STM can be accomplished using the currently available carrying capacity estimates. However, some important impacts are not included in the evaluation, e.g., land use, loss of biodiversity, solid wastes, hazardous wastes, and radioactive wastes. These impacts include the major public concerns for power generation using nuclear power and hydropower. The STM scores and the evaluation results may be altered should the carrying capacity for these impacts become available. It is important to state the limitations and assumptions made in the interpretation of LCA results.

13.3 Case Study II: LCA for Basic Material Productions

13.3.1 LCA for Basic Material Productions Using the STM

Goal and Scope Definition

The purpose of this evaluation is to analyze the environmental performance of the production for four basic materials: ABS plastic, aluminum, copper, and steel. This is a demonstration of application of the STM implementation at the supply line level. For this purpose, the comparison will be carried out on the basis of one-kilogram production for the basic materials. The life-cycle stages of a basic material production in this evaluation begin with the extraction of minerals and ends with the emissions of wastes to environmental media. The system boundary of the basic material productions is similar to that of the first case study (Figure 13.1). The result of this evaluation will be used to discuss the applicability of the U.S. carrying capacity as the baseline reference for the STM, an environmental performance metric. The relative ranking of these four basic materials is an illustration of how to identify the superior option among alternatives.

Life Cycle Inventory

The life cycle inventory (LCI) data for this case study were obtained from the IDEMAT 2001 database, which is integrated in the SimaPro LCA software (PRe' 2006). The functional unit of this assessment is 1 kg production of those four basic materials.

Life Cycle Impact Assessment

The inventory data can be classified into various impact categories (classification) and the impacts can be aggregated within the same homogeneous category (or sub-category) using appropriate characterization factors or equivalency factors (characterization). After that, the characterizations and aggregations of Impact (I) can be made. Next, the Impact Reference levels (I_R) associated with the impact categories and impact subcategories are calculated from the available estimated carrying capacity. The calculations for I_R are carried out using the year 2000 based carrying capacity values as well as economic information. The Value Reference (V_R) of 100×10^6 \$/year is also selected for the calculation of I_R. The U.S. GDP and GGP for the I_R calculation are also the same as the ones used in the first case study.

When the Impact Reference levels (I_R) are available, EI_{PR}, RP, and EE can then be calculated. Table 13.10 is the summary of the results calculated for all four basic material products. The recent published prices of the basic materials used in the calculations of EE are 0.75 \$/kg for ABS, 1.38 \$/kg for aluminum, 1.53 \$/kg for copper, and 0.40 \$/kg for steel (USGS 2003).

Table 13.10 STM Indicators for Basic Material Productions

	Environmental impact, EI_{PR} (kg/yr)$^{-1}$				Eco-Efficiency (%)			
	ABS	Aluminum	Copper	Steel	ABS	Aluminum	Copper	Steel
Global warming	1.0E-08	3.3E-08	2.5E-08	9.5E-09	7.2E+01	4.2E+01	6.1E+01	4.2E+01
Stratospheric ozone depletion	2.8E-10	1.1E-08	5.9E-15	1.6E-12	2.7E+03	1.2E+02	2.6E+08	2.6E+05
Acidification	4.8E-08	2.4E-07	2.2E-06	2.3E-08	1.6E+01	6.0E+00	7.0E-01	1.8E+01
Eutrophication	2.0E-08	3.9E-08	1.7E-08	8.6E-09	3.7E+00	5.8E+01	9.0E+01	9.0E+01
Photochemical smog	2.1E-09	2.3E-09	2.4E-09	5.7E-10	7.5E+02	8.1E+02	6.9E+02	9.6E+02
Human toxicity	4.4E-08	1.8E-07	1.3E-05	4.0E-08	3.4E+01	1.7E+01	1.0E-01	4.8E+01
Eco-toxicity	3.6E-07	6.8E-07	8.0E-06	3.9E-07	3.0E+00	8.0E+00	2.0E-01	3.0E+00
Resource depletion	2.2E-05	1.8E-03	2.4E-04	1.1E-05	1.4E-01	6.0E-01	7.0E-03	9.0E-02
Overall	2.2E-05	1.9E-03	2.6E-04	1.2E-05	3.4E-02	7.0E-04	5.9E-.03	3.5E-02
Rank	2	4	3	1	2	4	3	1
Resource productivity, RP (kWh_e/yr)	4.5E+04	5.4E+02	3.9E+03	8.6E+04				
Value productivity, VP, ($/yr)	3.4E+04	7.5E+02	5.9E+03	3.5E+04				

Interpretation

By comparing the sustainability level using overall EE, the results from Table 13.10 show that all of the basic material productions are much lower than the sustainable level (EE for ABS 3.4×10^{-2} %, aluminum 7.0×10^{-4} %, copper 5.9×10^{-3} %, and steel 3.5×10^{-2} %). These very low EE values indicate that production of these basic materials is many times worse than the sustainability level, which is EE =100%. In other words, a very low EE values indicate that the market prices of the basic material productions do not reflect their potential environmental costs. By using EE as the criterion, the basic material productions can be ranked from the one with the greatest EE to the one with the smallest EE as: steel, ABS plastic, copper, and aluminum (Figure 13.8). It should be noted that the EE for ABS plastic and steel are almost equal while EI for ABS is about two time higher (steel is more favorable). As discussed, the result for an alternative for which the overall EE < 100% does not necessarily indicate that every impact category does not meet the sustainability level.

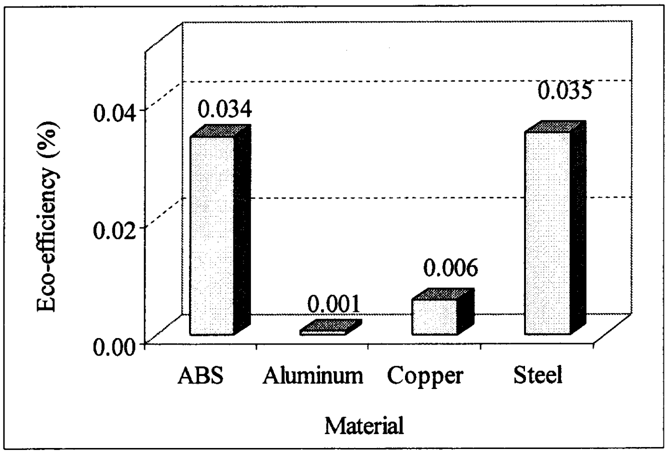

Figure 13.8 Eco-efficiency (EE) of four basic material productions.

EI_{PR} values for individual impacts indicate that resource depletion is the worst or the most limiting impact for all of the basic material productions evaluated. EI_{PR} for resource depletion accounts for most of the overall EI_{PR} for every basic material production (97.8% for ABS, 99.9% for aluminum, 91.2% for copper, and 95.9% for steel). The results also indicate that the major EI_{PR} for resource depletion is basically the one associated with the use of raw minerals for the material production (except for ABS which EI_{PR} is the worst for the consumption of natural gas).

13.3.2 LCA for Basic Material Productions Using Other LCIA Methods

Goal and Scope Definition

LCA for the case study of basic material productions is carried out using four other LCIA methods: EI95, EI99, EPS, and EDIP. The purpose of this evaluation is to compare the implementation of these LCIA methods with that of the STM. The comparative analysis is to investigate the superiority among the material production methods in terms of environmental performance. The environmental impacts contributing to the overall single-score indicators are discussed. Some characteristics of the STM compared to those other four LCIA methods are highlighted and discussed.

Life Cycle Inventory

LCI data are the same as the ones used in the STM implementation, and are obtained from the SimaPro LCA software. The functional unit is a basic material production of one kilogram.

Life Cycle Impact Assessment

The LCIAs using four different methods are evaluated by the SimaPro LCA software, which the four LCIA methods are readily available. The results obtained from the evaluations are presented in Table 13.11 to Table 13.14 for the assessment using EI95, EI99, EPS, and EDIP, respectively.

Interpretation

The results are interpreted, compared, and discussed in the following subsection.

13.3.3 Comparison and Discussion

Result Comparison-Contribution analysis

Table 13.15 is the summary of the primary environmental impact of basic material productions assessed by the different LCIA methods. The contribution analysis of environmental impacts reveals different trends of what would be the major contributors of energy generation sources. Basically, resource depletion is the major contributor for most of the four material productions that are assessed using the STM, EI99, and EPS.

Table 13.11 Eco-Indicator 95 Scores for Basic Material Productions

Eco-Indicator 95 score	Material			
	ABS	Aluminum	Copper	Steel
Weighted indicator (millipoints/kg)				
Greenhouse	6.2E-04	2.3E-03	1.4E-03	5.5E-04
Ozone layer	1.1E-04	6.6E-05	5.0E-08	1.3E-05
Acidification	1.6E-03	6.6E-03	6.1E-02	6.2E-04
Eutrophication	2.1E-04	3.5E-04	4.2E-04	8.5E-05
Heavy metals	9.8E-05	5.7E-03	1.0E-05	4.1E-03
Carcinogens	1.9E-05	2.2E-02	4.8E-07	5.00E-05
Winter smog	5.3E-04	3.2E-03	3.6E-02	2.2E-04
Summer smog	2.9E-04	1.5E-04	3.9E-05	4.00E-05
Pesticides	0	0	0	0
Energy resources	0	0	0	0
Solid waste	0	0	0	0
Overall	3.5E-03	4.0E-02	9.9E-02	5.7E-03
Rank	1	3	4	2

Table 13.12 Eco-Indicator 99 Scores for Basic Material Productions

Eco-Indicator 99 score	Material			
	ABS	Aluminum	Copper	Steel
Weighted indicator (millipoints/kg)				
Carcinogens	4.0E-04	2.6E-02	2.9E-05	4.4E-03
Respiratory damage-organics	1.3E-04	5.3E-05	1.7E-05	1.6E-05
Respiratory damage-inorganics	3.6E-02	1.3E-01	7.6E-01	1.3E-02
Climate change	1.4E-02	5.3E-02	3.1E-02	1.2E-02
Radiation	0	0	0	0
Ozone layer	0	9.4E-06	7.2E-09	1.9E-06
Ecotoxicity	9.1E-05	4.0E-03	1.2E-04	3.6E-02
Acidification/Eutrophication	5.7E-03	1.3E-02	6.6E-02	2.4E-03
Land use	2.4E-03	2.5E-02	7.4E-02	3.3E-03
Minerals	1.2E-05	6.6E-02	1.3E+00	2.0E-03
Fossil fuel use & crude fuel acquisition	3.4E-01	3.9E-01	4.2E-01	7.1E-02
Overall	4.0E-01	7.1E-01	2.7E+00	1.4E-01
Rank	2	3	4	1

Table 13.13 EPS Scores for Basic Material Productions

EPS 2000 score	Material			
	ABS	Aluminum	Copper	Steel
Weighted indicator (ELU/kg)				
Life expectancy	3.2E-01	3.8E+00	2.7E+00	2.6E-01
Severe mobility	1.1E-01	1.2E+00	-1.8E-01	1.0E-01
Morbidity	2.4E-02	9.3E-02	1.2E-01	2.0E-02
Severe nuisance	8.2E-05	3.2E-03	1.5E-05	3.5E-02
Nuisance	9.3E-03	4.6E-02	4.4E-01	3.7E-03
Crop growth capacity	1.6E-03	3.5E-03	1.6E-03	8.6E-04
Wood growth capacity	-6.3E-03	-1.8E-02	-1.5E-02	-5.1E-03
Fish and meat production	-3.6E-04	-5.8E-04	-3.8E-05	-1.5E-04
Soil acidification	2.8E-04	1.2E-03	1.1E-02	1.1E-04
Production capacity from irrigation water	0	0	0	0
Production capacity from drinking water	0	0	0	0
Depletion of reserves	1.6E+00	1.8E+00	2.1E+02	1.4E+00
Species extinction	6.1E-03	1.7E-02	-8.4E-03	5.1E-03
Overall	2.0E+00	7.0E+00	2.2E+02	1.8E+00
Rank	2	3	4	1

Table 13.14 EDIP Scores for Basic Material Productions

EDIP score	Material			
	ABS	Aluminum	Copper	Steel
Weighted indicator (Person-equiv./kg)				
Global warming	5.1E-04	1.5E-03	1.1E-03	4.4E-04
Ozone depletion	0	5.2E-05	4.0E-08	1.1E-05
Acidification	1.9E-04	7.8E-04	7.2E-03	7.4E-05
Eutrophication	6.5E-05	1.1E-04	1.3E-04	2.7E-05
Photochemical smog	3.4E-05	1.2E-04	6.2E-06	7.0E-06
Ecotoxicity water chronic	8.4E-05	9.9E-03	7.7E-06	2.7E-03
Ecotoxicity water acute	8.8E-05	9.0E-03	6.1E-06	2.3E-03
Ecotoxicity soil chronic	6.0E-08	3.2E-03	6.9E-06	7.2E-04
Human toxicity air	2.3E-06	2.3E-04	2.8E-04	4.6E-04
Human toxicity water	8.4E-05	1.2E-03	5.9E-06	2.3E-04
Human toxicity soil	2.6E-05	1.7E-03	3.8E-04	9.2E-04
Bulk waste	6.8E-05	8.9E-04	1.1E-01	1.3E-04
Hazardous waste	1.1E-03	1.1E-06	0	0
Radioactive waste	0	0	0	0
Slags/ashes	3.8E-05	3.3E-03	3.2E-07	3.3E-04
Resources	0	0	0	0
Overall	2.2E-03	3.2E-02	1.2E-01	8.3E-03
Rank	1	3	4	2

Table 13.15 Major Contributing Impact to Total Impact for Basic Material Productions Assessed from the STM and Other LCIA Methods

Method	Major impact			
	ABS Plastic	Aluminum	Copper	Steel
STM	Resource depletion	Resource depletion	Resource depletion	Resource depletion
Eco-Indicator 95	Acidification	Carcinogens	Acidification	Heavy metals
Eco-Indicator 99	Fuel acquisition	Fuel acquisition	Minerals	Fuel acquisition
EPS 2000	Depletion of reserves	Life expectancy	Depletion of reserves	Depletion of reserves
EDIP	Hazardous	Eco-toxicity	Bulk waste	Eco-toxicity

Again, it should be noted that EI95 and EDIP do not include resource depletion in their methods explicitly. Instead, the environmental impacts associated with resource use are determined in terms of embedded energy for raw material and fuel acquisitions. However, LCI data for the basic material productions do not identify this portion of embedded energy. Therefore, the contribution of resource depletion impact to the overall impacts cannot be identified specifically for EI95 and EDIP. It is plausible that the environmental impact due to resource depletion is also the major contributor assessed by EI95 and EPIP as well as other methods. Figure 13.9 to Figure 13.13 present the contributions of major impact categories for the environmental performance assessments using the STM as well as other four LCIA methods.

The results from EPS also indicate a similar aspect to the STM, in which the market prices of the material productions do not reflect their actual environmental costs. The ELUs, which are equivalent to the environmental cost, of approximately 2.0, 7.0, 216.0, and 1.8 €/kg are greater than the published market prices of 0.75, 1.38, 1.53, 0.40 $/kg for ABS, aluminum, copper, steel respectively (It is conveniently assumed that 1€ = 1$). The environmental cost of copper acquisition is far beyond the market price.

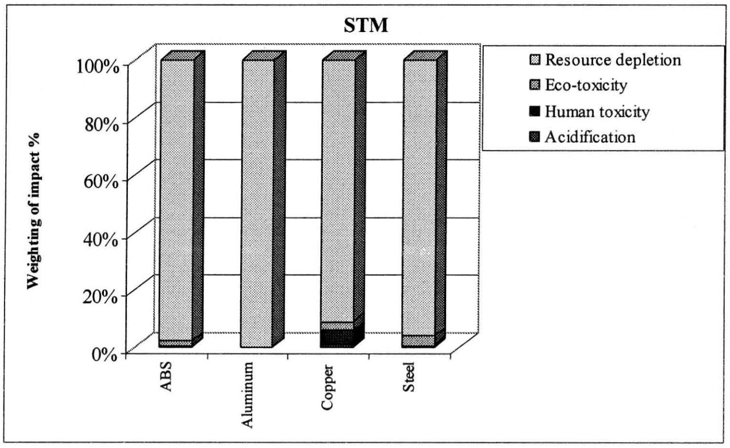

Figure 13.9 Major contributors of basic material productions assessed by the STM.

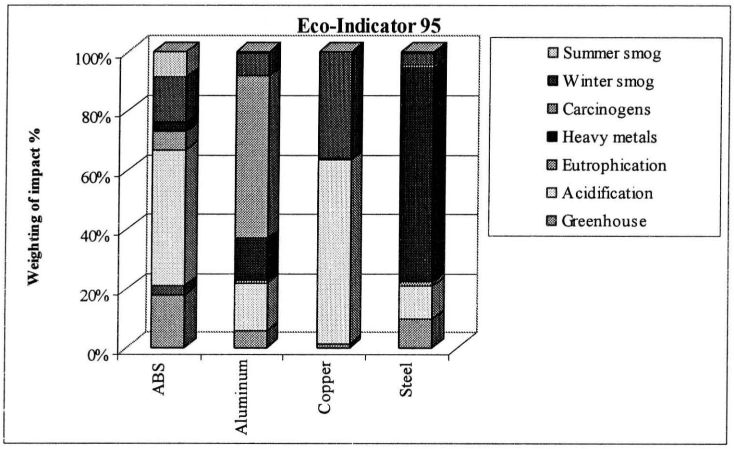

Figure 13.10 Major contributors of basic material productions assessed by Eco-Indicator 95.

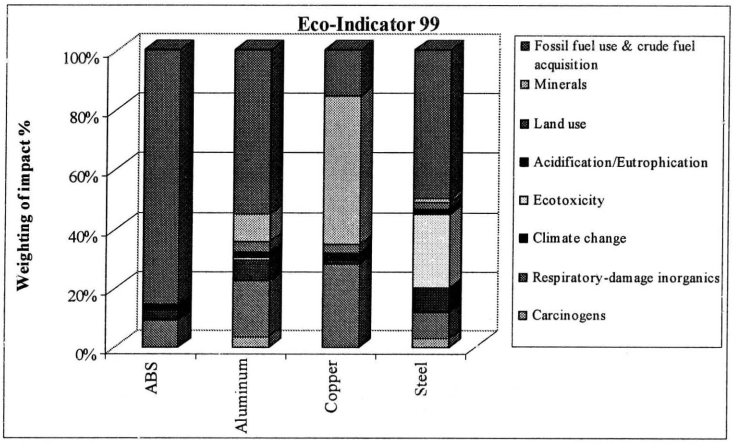

Figure 13.11 Major contributors of basic material productions assessed by Eco-Indicator 99.

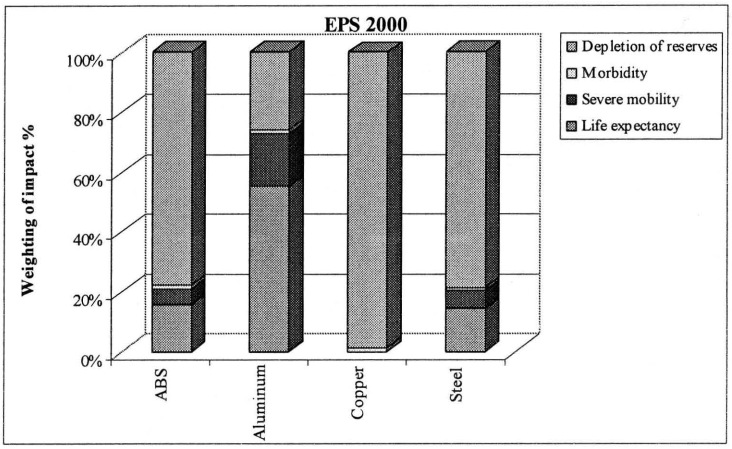

Figure 13.12 Major contributors of basic material productions assessed by EPS.

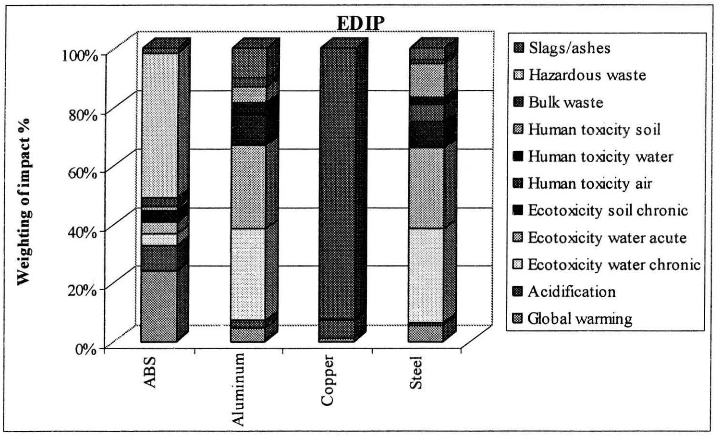

Figure 13.13 Major contributors of basic material productions assessed by EDIP.

Result Comparison-Ranking of alternatives

The ranking of alternatives using EI_{PR} of the STM can be compared to those using single-scores of other LCIA methods (Table 13.16). The average trend of all five methods is ranked from steel, ABS, aluminum, and copper (same as EI99 and EPS). When comparing these rankings with the rankings that consider only the five common impact categories, the results indicate that the rankings that take into account only the common impact categories provide a more agreeable trend (Table 13.17).

Table 13.16 Ranking in Terms of Environmental Performance of Basic Material Productions Assessed by the STM and Other LCIA Methods

Method	Environmental performance ranking			
	1	2	3	4
STM	Steel	ABS	Copper	Aluminum
Eco-Indicator 95	ABS	Steel	Aluminum	Copper
Eco-Indicator 99	Steel	ABS	Aluminum	Copper
EPS 2000	Steel	ABS	Aluminum	Copper
EDIP	ABS	Steel	Aluminum	Copper

Table 13.17 Comparison between the Rankings with All Impact Categories and the Rankings with only Five Common Impact Categories

	ABS	Aluminum	Copper	Steel
Rankings with all impacts				
STM	2	4	3	1
EI95	1	3	4	2
EI99	2	3	4	1
EDIP	1	3	4	2
Rankings with only common impacts				
STM	2	3	4	1
EI95	3	2	4	1
EI99	2	3	4	1
EDIP	2	3	4	1

Discussion

This section has shown the application of the carrying capacity in conjunction with the STM to evaluate the environmental performance for four types of material production, reveals a more complex system than the first case study. This case study confirms that the STM is capable not only of identifying the best environmentally sound option among alternatives, but it can also relate their environmental performance to sustainability.

The alternative ranking for the case study obtained from the STM gives a slightly different trend from those obtained from other LCIA methods. One aspect that can be pointed out here is the comparison of the environmental performance between the production of copper and aluminum. The other LCIA methods all reveal that aluminum production has less environmental impact than that of copper production while the STM shows the result in the other way. The STM gives a different result because the calculation for the resource depletion impact is based on the regional level (U.S.). In the U.S., bauxite (basic of aluminum ore) is far less available than copper when compared to the world reserve base (0.12% for bauxite and 13.8% for copper). Consequently, this lesser availability of bauxite results in a smaller EE for aluminum production than that for copper production. An investigation reveals that should the resource depletion impact be based on the global level, EE of aluminum production will be greater than that of copper production (the results not shown here). The ranking of the latter case will be the same as those of EI99 and EPS, which is: steel, ABS, aluminum, and copper.

A conclusion that can be drawn from the above paragraph is that spatial variation should be taken into consideration in the environmental performance evaluation since it can affect the outcome in terms of superiority among alternatives. The following section is the environmental performance evaluation of a household coffee maker, a more complex product level. This third case study is attempt to illustrate the potential of using the STM to deal with both the temporal and spatial variations.

13.4 Case Study III: LCA for a Household Coffee Maker

13.4.1 LCA for a Household Coffee Maker Using the STM

Goal and Scope Definition

This case study is conducted to investigate the implementation of the STM to assess the environmental performance at the product level, which has more complex inventory data than the process and the supply line levels as illustrated in the first and the second case studies. The other purpose is to study the capability of the STM in conjunction with the estimated carrying capacity to deal with the temporal and spatial variations of aspects at the production process. A typical household plastic coffee maker is selected for this non-comparative assessment. The results of this assessment will be used to discuss the applicability of the carrying capacity as the baseline reference for the STM to assess the environmental performance at the product level, the capability of the STM in conjunction with the carrying capacity to deal with the temporal and spatial variations, and the contribution analysis of individual impacts to the overall impact score. This assessment is limited to the production stage only, and is not extended to the use or other stages.

A production process scenario of a typical plastic coffee maker has been established so that this case study can be conducted to meet the purposes discussed above. This plastic coffee maker consists of five parts namely: aluminum heating element, polypropene plastic housing, glass jug, power cord, and small parts. There are three energy aspects associated with the assembly of these parts, which are energy used for injection molding of the plastic housing, heat used for making of the glass jug, and energy used for the final assembly. The numbers of parts and their associated energy uses are the same as the plastic coffee maker demonstrated in the SimaPro LCA software (Pre' 2006). It is assumed for this case study that the subassemblies and assembly of the coffee maker were carried out in three different years from 2000-2002. It is also assumed that all parts were made in the U.S. but the location of the manufacturing facilities were not identified. However, the final assembly was done in a facility in New Jersey and its associated energy use was generated from the sources within the state. The system boundary for the productions of each part or component for the coffee maker is similar to that of the first case study (Figure 13.1). The schematic diagram of the production process of the coffee maker is presented in Figure 13.14.

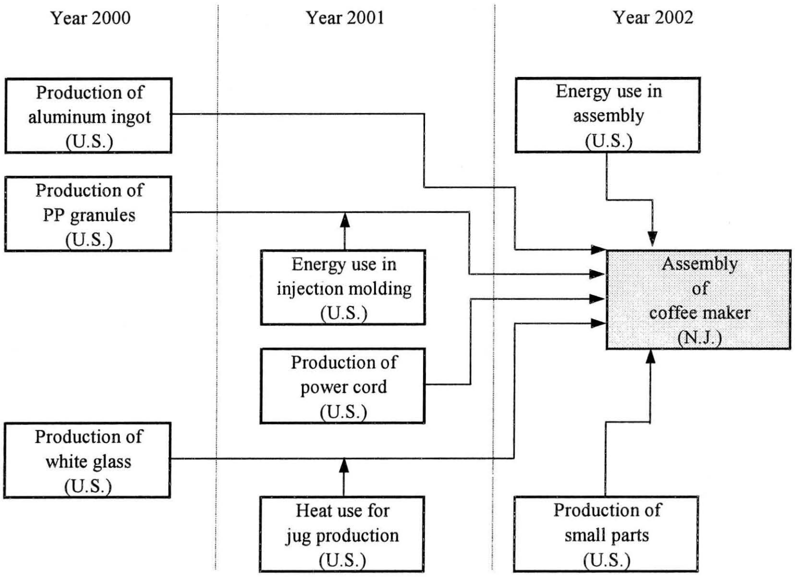

Figure 13.14 Schematic diagram shows the temporal and spatial variations of the production of household plastic coffee maker case study.

Life Cycle Inventory

LCI data for the manufacturing of the typical household coffee maker were obtained from the SimaPro LCA software (PRe' 2006), the coffee machine demo model Sima. The inventory data are associated with the manufacturing of a typical household plastic coffee maker with a serving size of ten cups. The total weight is approximately 1.7 kg. LCI data are separated into eight processes as presented in Table 13.18.

Table 13.18 Processes in the Household Plastic Coffee Maker

Subassembly/Assembly	Material		Energy	
	Amount	Unit	Amount	Unit
Aluminum ingot	0.1	kg		
PP granules	1.0	kg		
Injection molding for 1 kg			3.6	MJ
Small parts	0.1			
Glass	0.4	kg		
Heat use for jug production			4	MJ
Power cord	0.1			
Energy use in assembly				
Total	1.7	kg	7.6	MJ

Life Cycle Impact Assessment

The inventory data are categorized into various impact categories in the classification step and the impacts are aggregated within the same category in the characterization step using the characterization factors. After that, the STM's Impact Reference levels (I_R) are made for eight sets corresponding to the processes as shown in Table 13.18. These I_R sets are calculated from the carrying capacity estimates according to the designated time and location for individual processes as summarized in Table 13.19.

Table 13.19 Temporal and Spatial Variations of Carrying Capacity and Impact Reference Levels Used in the Evaluation of the Production of Household Coffee Maker

Subassembly/Assembly	Impact category															
	Global warming		Ozone depletion		Acidification		Eutrophication		Photo. Smog		Resource depletion		Human toxicity		Eco-toxicity	
	year	Scale	year	Scale	year	Scale	year	Scale	year	Scale	year	Scale	year	Scale	year	Scale
Production of aluminum ingot	2000	glob.	2000	glob.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.
Production of PP granules	2000	glob.	2000	glob.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.
Energy use injection molding	2001	glob.	2001	glob.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.
Production of small parts	2002	glob.	2002	glob.	2002	U.S.	2002	U.S.	2002	U.S.	2002	U.S.	2002	U.S.	2002	U.S.
Production of white glass	2000	glob.	2000	glob.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.	2000	U.S.
Heat use for jug production	2001	glob.	2001	glob.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.
Production of power cord	2001	glob.	2001	glob.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.	2001	U.S.
Energy use in assembly	2002	glob.	2002	glob.	2002	U.S./NJ	2002	U.S.	2002	U.S.	2002	US/NJ	2002	U.S.	2002	U.S.

The carrying capacity estimates that are time-dependent are global warming and stratospheric ozone depletion impacts. For these two impacts, the carrying capacity estimates in the time frame of the established scenario are presented in Table 13.20 (see also Chapter 5 and Chapter 6). The carrying capacity estimates that take into account the local conditions are acidification and water consumption impacts. For acidification, the carrying capacity estimate with local (regional-northeast) conditions is 2.37×10^{10} kg SO_2/yr (compared to U.S. average of 3.04×10^{10} kg SO_2/yr N/yr). For surface water consumption, the carrying capacity estimate is calculated from the average rainfall intensity in New Jersey of 44.7 inches/yr (see also Chapter 12) to become the carrying capacity of 9.02×10^{11} m^3/yr (compared to U.S. average of 7.00×10^{11} m^3/yr with the average rainfall intensity of 34.7 inches/yr). It should be noted that not only locally-specific environmental sensitivities are taken into account in this LCA using the STM, but the spatial distributions of emissions are also considered. However, the spatial distributions of emissions are addressed by applying site factors in the characterization of emission inventory. The emissions that contribute to acidification, eutrophication, and photochemical ozone formation impacts need to be characterized using site factors prior to the STM calculations (see also Chapter 7, 8, and 9).

Table 13.20 Variations in Carrying Capacity Estimates Used in the Third Case Study

Impact category	Scale	Unit	Year		
			2000	2001	2002
Global warming	Global	kg CO_2/yr	2.93E+13	2.57E+13	2.20E+13
Stratospheric ozone depletion	Global	kg CFC11/yr	3.48E+08	3.30E+08	3.19E+08
Acidification	U.S. scale with N.J. conditions	kg SO_2/yr	2.37E+10		
Resource depletion-Surface water consumption	U.S. scale with N.J. conditions	m^3/yr	9.02E+11		

Given the appropriate carrying capacity estimates, the calculations for I_R can then be made. The Value Reference (V_R) of 100×10^6 \$/year is also used for the calculation of I_R. The U.S. GDP and GGP for I_R calculation are varied during the years 2000-2002 (Table 13.21). When the I_R are available, EI_{PR}, RP, and EE can also be calculated. The price of a typical household coffee maker is arbitrarily selected from the average U.S. market price of \$20.00 (May 2003). Table 13.22 is the summary of the STM indicators calculated for the household coffee maker. Table 13.23 presents the Eco-Efficiency values associated with the production aspects of the coffee maker.

Table 13.21 U.S. GDP and GGP (unit: $billion)
(Source: World Bank 2003)

	Year		
	2000	2001	2002
Global domestic product (GDP)	9,810	10,065	10,417
Gross global product (GGP)	31,500	31,136	32,252

Table 13.22 STM Indicators for the Production of the Plastic Household Coffee Maker

Subassembly/assembly	Environmental impact, EI_{PR}, (Unit/yr)$^{-1}$								
	Global warming	Ozone depletion	Acidification	Eutrophication	Photochemical ozone formation	Human toxicity	Eco-toxicity	Resource depletion	Total
Aluminum ingot	2.7E-09	8.4E-11	1.6E-08	3.1E-09	1.9E-10	2.6E-08	7.3E-08	1.9E-04	1.9E-04
PP granules	6.3E-09	2.6E-10	4.8E-08	1.3E-08	1.3E-09	1.5E-08	8.5E-08	1.0E-05	1.1E-05
Energy for Injection molding	1.8E-09	2.3E-11	7.6E-09	1.9E-09	9.6E-11	1.3E-08	7.3E-08	4.2E-07	5.2E-07
Small parts	3.8E-09	9.8E-11	1.7E-08	3.2E-09	2.0E-10	1.3E-08	1.6E-07	1.6E-05	1.6E-05
White glass	1.0E-09	4.9E-11	4.7E-09	1.2E-09	1.1E-10	4.9E-08	6.6E-08	7.0E-07	8.2E-07
Heat use for jug production	9.5E-10	5.3E-13	8.4E-10	1.9E-10	2.4E-11	1.2E-09	7.8E-09	4.8E-07	4.9E-07
Power cord	2.2E-09	6.8E-11	2.7E-08	3.2E-09	2.8E-10	1.3E-08	8.9E-08	1.7E-05	1.7E-05
Electricity use in assembly	4.3E-09	4.9E-11	6.4E-09	3.7E-09	1.2E-10	2.8E-08	1.5E-07	8.7E-07	1.1E-06
Total	2.3E-08	6.3E-10	1.3E-07	3.0E-08	2.3E-09	1.3E-07	7.0E-07	2.4E-04	2.4E-04
Overall resource productivity, RP, (Unit/yr)									4.2E+03
Value productivity, VP, ($/yr)									8.4E+04
Overall eco-efficiency, EE, (%)									8.4E-02

Table 13.23 Eco-Efficiency for the Production of the Plastic Household Coffee Maker

Subassembly/assembly	Eco-Efficiency, EE (%)								
	Global warming	Ozone depletion	Acidification	Eutrophication	Photochemical ozone formation	Human toxicity	Eco-toxicity	Resource depletion	Overall
Aluminum ingot	7.4E+03	2.4E+05	1.2E+03	1.3E+04	1.4E+05	2.3E+03	4.1E+02	1 0E-01	1.0E-01
PP granules	3.2E+03	7.7E+04	4.2E+02	2.4E+03	2.2E+05	4.5E+03	4.2E+02	4.0E+00	1.9E+00
Energy for Injection molding	11E+03	8.8E+05	2.6E+03	1.8E+04	2.3E+05	3.7E+02	3.4E+02	3.4E+01	9.6E+01
Small parts	5.3E+03	2.1E+05	1.2E+03	1.2E+04	1.3E+05	6.4E+03	2.5E+02	6.0E+00	1.3E+00
White glass	2.0E+04	4.1E+05	4.3E+03	1.8E+04	2.4E+05	4.9E+02	3.5E+02	4.2E+01	2.4E+01
Heat use for jug production	2.1E+03	3.8E+07	2.4E+03	1.2E+05	9.2E+05	6.7E+03	6.2E+02	4.3E+01	4.1E+01
Power cord	9.3E+03	2.9E+05	7.3E+02	9.6E+03	9.7E+04	6.6E+02	6.1E+02	1.0E+00	1.2E+00
Electricity use in assembly	4.7E+02	4.1E+04	3.1E+02	8.9E+02	1.9E+04	1.8E+02	1.6E+02	4.6E+01	1.9E+01

Interpretation

The results from Table 13.22 reveal that resource depletion is the most significant or limiting impact for the manufacturing of the household coffee maker. The contribution of resource depletion accounts for most of the total environmental impact (99.56%) indicated by $\mathrm{EI}_{\mathrm{PR}}$. The production of aluminum ingot contributes 80.41% of the overall environmental impact. This is due to a very limited aluminum reserve in the U.S., discussed previously. As a result, the year 2000, when the basic materials were produced, is the year that the production of the coffee maker poses most of its associated environment impacts (Figure 13.15).

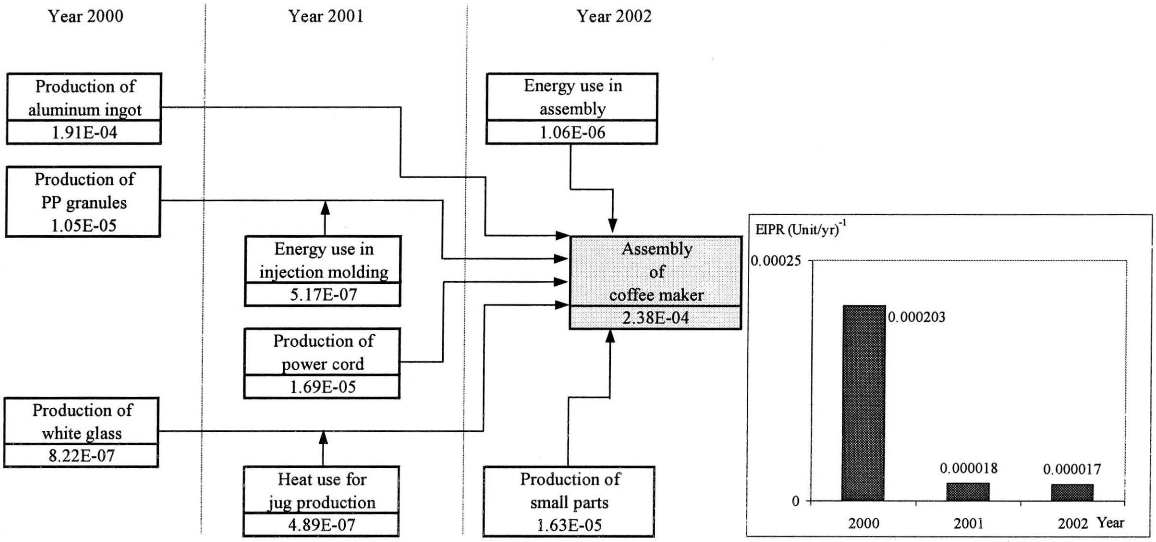

Figure 13.15 $\mathrm{EI}_{\mathrm{PR}}$ contributions due to the production of the household coffee maker.

$\mathrm{EI}_{\mathrm{PR}}$ for individual parts/processes can be viewed graphically in Figure 13.16. Overall $\mathrm{EI}_{\mathrm{PR}}$ contributions of this coffee maker can also be presented as an environmental impact web diagram (EI-web diagram) or compass (EI-compass). This EI-compass is made to the log scale. The most significant impact is, as stated, resource consumption. The following significant impacts are posted clockwise in the web-diagram: acidification, human toxicity, eutrophication, global warming, eco-toxicity, stratospheric ozone depletion, and photochemical ozone formation (photochemical smog) (Figure 13.17).

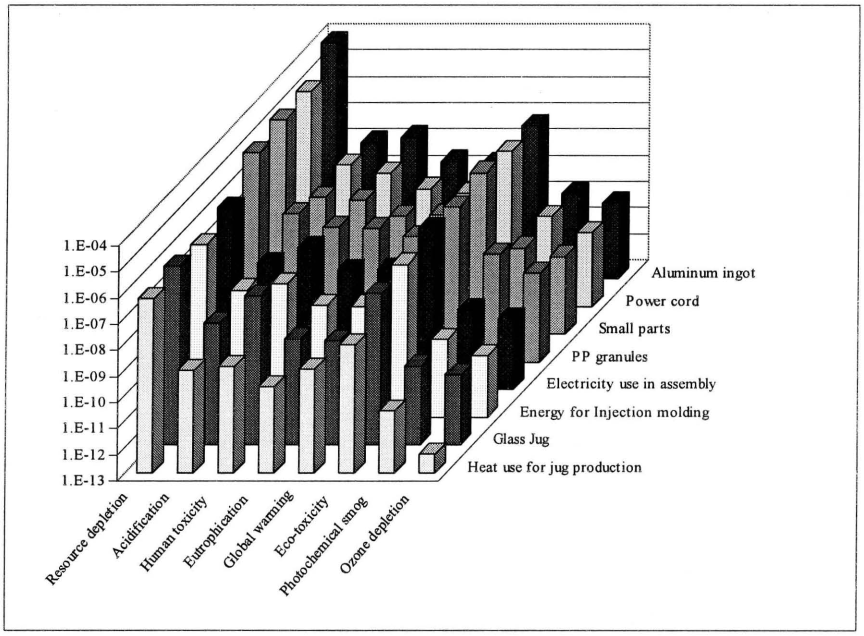

Figure 13.16 EI_{PR} contributions to individual impact categories.

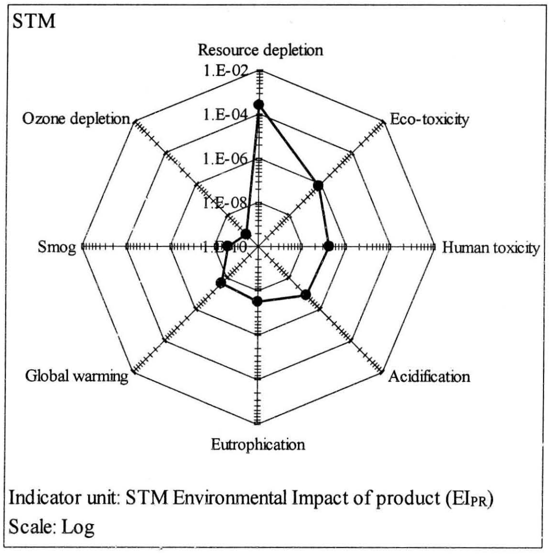

Figure 13.17 EI-Web diagram of the coffee maker assessed by the STM.

The overall Eco-Efficiency of 0.08% indicates that the production of the household coffee maker is about 1,000 times worse than the sustainability level (the criteria for sustainability is EE = 100%). When investigating the environmental impacts of individual raw materials, it is found that the scarcest resource is bauxite (with the smallest EE at 0.1%), which is used in the manufacturing of aluminum. As discussed, the overall Eco-Efficiency does not indicate that all of the impact categories do not meet the sustainability criteria. The sustainability level in terms of Eco-Efficiency of impact categories must be focused individually.

By comparing the Eco-Efficiency of individual impact categories and subcategories (the details are not presented since the calculation spreadsheet is large and occupies a lot of space), it is found that the consumption of fluorspar and copper are also below the sustainability target (2% each). The consumption of fuels and water are also worse than the sustainability target (oil 4%, natural gas 12%, lignite 10%, and water 23%). The only emission-related impact category that does not meet the sustainability level is eco-toxicity with an Eco-Efficiency of 52%.

By using the STM as an impact assessment tool, the environmental impacts posed by the production of the household coffee maker case study can be identified. Furthermore, this case study also presents the interpretation of the results in a few meaningful ways. This interpretation will be a helpful communication tool with the interest groups, as well as help the product designers to improve the environmental performance of a product following the precautionary principle by focusing on the environmental aspects of the manufacturing process that tend not to meet the sustainability target.

13.4.2 LCA for Household Coffee Maker Using Other LCIA Methods

Goal and Scope Definition

LCA for the household coffee maker is evaluated using four different LCIA methods: EI95, EI99, EPS, and EDIP. The purpose of this assessment is to compare the application of these LCIA methods with that of the STM. The comparative analysis is to investigate the environmental impact contributions to the single-score indicators.

Life Cycle Inventory

LCI data are the same as the ones used in the STM implementation. The LCI data was obtained from the SimaPro LCA Software. The functional unit is also the same, which is the production of one household coffee maker.

Life Cycle Impact Assessment

The LCIA using four different methods are evaluated through the SimaPro LCA software. These methods are readily available in the SimaPro LCA software. The results obtained from the evaluations are in a matrix form presenting the single scores of individual impact categories. The results are interpreted, compared, and discussed next.

13.4.3 Comparison and Discussion

Result Comparison-Contribution analysis

The single scores for the household coffee maker are summarized in Table 13.24 and Figure 13.18 for the assessment using the STM, EI95, EI99, EPS, and EDIP. The results for the EPS show that, coincidently, the total ELU (the environmental cost of € 20.2 per unit) is almost the same as the economic value of the typical household coffee maker (arbitrarily selected from the average U.S. market price of $20 as of May 2003).

Table 13.24 Single-Score Indicators for a Unit of Household Coffee Maker Assessed by the STM and Other LCIA Methods

STM	
Impact category	EI_{PR}
Resource depletion	2.4E-04
Acidification	7.0E-07
Human toxicity	1.6E-07
Eutrophication	1.3E-07
Global warming	3.0E-08
Eco-toxicity	2.3E-08
Stratospheric ozone depletion	2.3E-09
Photochemical smog	6.3E-10
Total	2.4E-04

EI95	
Impact category	Millipoint
Heavy metals	9.3E-03
Acidification	5.0E-03
Winter smog	2.2E-03
Carcinogens	1.9E-03
Greenhouse	1.2E-03
Summer smog	1.1E-03
Eutrophication	3.8E-04
Ozone layer	3.7E-04
Pesticides	0
Energy resources	0
Solid waste	0
Total	2.1E-02

EI99	
Impact category	Millipoint
Fossil fuels	6.0E-01
Minerals	1.1E-01
Respiratory damage- inorganics	9.7E-02
Climate change	2.6E-02
Carcinogens	1.6E-02
Ecotoxicity	1.4E-02
Acidification/Eutrophication	1.3E-02
Land use	1.2E-03
Respiratory damage-organics	4.5E-04
Ozone layer	5.3E-05
Radiation	2.6E-05
Total	8.8E-01

EPS 2000	
Impact category	ELU
Depletion of reserves	1.9E+01
Life expectancy	9.1E-01
Severe mobility	2.6E-01
Severe nuisance	6.3E-02
Morbidity	4.7E-02
Nuisance	3.2E-02
Species extinction	7.7E-03
Crop growth capacity	2.9E-03
Soil acidification	8.8E-04
Use of Irrigation water	0
Use of drinking water	0
Fish and meat production	0
Wood growth capacity	0
Total	2.0E+01

EDIP	
Impact category	Person-equivalent
Ecotoxicity water chronic	3.4E-03
Ecotoxicity water acute	3.0E-03
Human toxicity soil	2.4E-03
Human toxicity water	1.6E-03
Global warming	9.2E-04
Human toxicity air	8.2E-04
Acidification	5.9E-04
Ecotoxicity soil chronic	4.1E-04
Ozone depletion	2.9E-04
Eutrophication	1.2E-04
Bulk waste	1.1E-04
Photochemical smog	3.4E-05
Hazardous waste	0
Radioactive waste	0
Slags/ashes	0
Resources	0
Total	1 4E-02

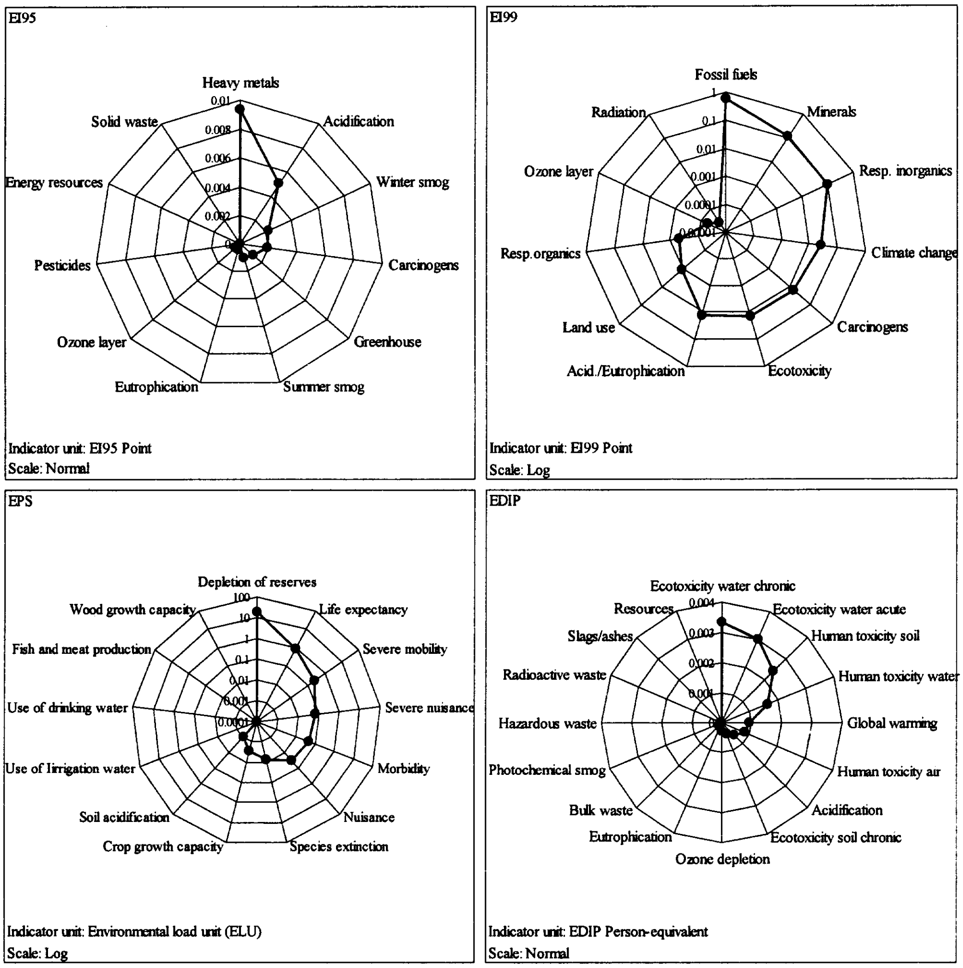

Figure 13.18 EI-Web diagrams of the household coffee maker assessed by other LCIA methods.

Table 13.25 is the summary of the contribution analysis of the household coffee maker assessed by the STM and other LCIA methods. For the STM, the results have been discussed in Subsection 13.4.1. The resource depletion is the primary contributor of the STM, EI99, and EPS. It should be noted that EI95 and EDIP do not take into account the resource depletion (the environmental damage due to resource depletion is evaluated in terms of embedded energy for raw material productions and fuel acquisitions). Hence, there is a possibility that resource depletion would dominate the environmental impact for all LCIA methods should resource depletion be included in EI95 and EDIP. The next two major contributors are the acidification and the human toxicity impacts. The human toxicity impact can be observed in the STM (human toxicity), EI95 (heavy metals, winter smog and carcinogens), EI99 (respiratory system damage due to inorganic substances), EPS (life expectancy and severe mobility), and EDIP (human toxicity). When comparing the contribution analysis that considers only five common impact categories, global warming and acidification are the two major contributors (Table 13.26).

Table 13.25 Environmental Impact Contribution Analysis of the Household Coffee Maker Assessed from Different LCIA Methods

STM		EI95		EI99		EPS		EDIP	
Impact category	%	Impact category	%	Impact category	%	Impact category	%	Impact category	%
Resource depletion	99.56	Heavy metals	68.21	Fossil fuels	68.21	Depletion of reserves	93.45	Ecotox water chronic	24.39
Acidification	0.296	Acidification	12.78	Minerals	12.78	Life expectancy	4.50	Ecotox water acute	22.06
Human toxicity	0.067	Winter smog	11.02	Resp damage- inorg	11.02	Severe mobility	1.29	Human toxicity soil	17.62
Eutrophication	0.054	Carcinogens	2.95	Climate change	2.95	Severe nuisance	0.31	Human toxicity water	11.94
Global warming	0.013	Greenhouse	1.77	Carcinogens	1.77	Morbidity	0.23	Global warming	6.73
Eco-toxicity	0.010	Summer smog	1.63	Ecotoxicity	1.63	Nuisance	0.16	Human toxicity air	5.96
Ozone depletion	0.001	Eutrophication	1.45	Acid/Eutro	1.45	Species extinction	0.04	Acidification	4.29
Photochemical smog	0.000	Ozone layer	0.13	Land use	0.13	Crop growth capacity	0.01	Ecotox soil chronic	2.96
		Pesticides	0.05	Resp. damage-org	0.05	Soil acidification	0.00	Ozone depletion	2.13
		Energy resources	0.01	Ozone layer	0.01	Irrigation water	0.00	Eutrophication	0.87
		Solid waste	0.00	Radiation	0.00	Use of drinking water	0.00	Bulk waste	0.82
						Fish and meat prod	0.00	Photochemical smog	0.24
						Wood growth cap.	0.00	Others	0.00

Table 13.26 Environmental Impact Contribution Analysis for Five Common Impact Categories

STM		EI95		EI99		EDIP	
Impact category	%	Impact category	%	Impact category	%	Impact category	%
Global warming	12.5	Global warming	14.7	Global warming	66.2	Global warming	47.2
Ozone depletion	0.3	Ozone depletion	4.6	Ozone depletion	0.1	Ozone depletion	15.0
Acidification	69.6	Acidification	62.6	Acidification/ Eutrophication	32.5	Acidification	30.1
Eutrophication	16.3	Eutrophication	4.7			Eutrophication	6.1
Photochemical ozone formation	1.2	Photochemical smog	13.3	Photochemical smog	1.2	Photochemical smog	1.7

Discussion

It has been demonstrated in this case study that the STM coupled with the appropriate set of carrying capacities has the capability to assess the environmental impact of a system that has temporal and spatial variations. The temporal and spatial variations in the assessment are seen not only in the environmental impacts, but are also seen in the economic information, i.e., the large-scale economic products and the value-added of the system itself.

The STM implementation of the product level (household coffee maker) is much more complex than that of the process level. This is due to the larger amount of inventory data. There are 438 environmental consumption-related and emission-related constituents associated with the production of the household coffee maker compared to the number of constituents as small as eight items in the electrical energy generation case study. With a large number of constituents, it would be beneficial to develop an STM software tool with an integrated carrying capacity database library.

A limitation that is observed from this case study is the limited number of carrying capacity estimates. As pointed out, there are a large number of environmental emissions and consumptions associated with a complex system. To make the STM application more universal, there should be a number of carrying capacity estimates covering all of the environmental emissions and consumptions. However, it is difficult to prepare the carrying capacity estimates for all of the emissions and consumptions in an inventory database. This is more obvious when considering the carrying capacity evaluation of the human and eco-toxicity impacts. The carrying capacity for toxic substances must be evaluated individually because the human and eco-toxicity are non-homogeneous impact categories[1]. The carrying capacity estimates for a number of toxic substances cannot be evaluated due to the unavailability of information, which in this case is the regulatory level of toxic substances to determine the carrying capacity in human toxicity impact. Likewise, PNEC of toxic substances are needed to determine the carrying capacity in the eco-toxicity impact. Generally, regulatory levels and PNEC are not available for all toxic substances. Sometimes a toxic substance is considered to pose no harm to human health or ecosystem health, therefore its regulatory levels or PNEC are not determined.

Other LCIA methods also share a similar limitation in the characterization step. Chemical substances need to be characterized to a reference chemical by using the equivalency factors in order for the substances to be aggregated[2]. An equivalency factor of a toxic substance is its relative hazard potential compared to the reference chemical of an impact category. In the evaluation of an equivalency factor, a threshold level (NOEC or PNEC) as well as the physical and chemical properties of the toxic substance are needed in environmental modeling (discussed in Chapter 10 and 11). These properties and characteristics are not available for all chemicals reported in the LCI database either. As a result, a number of toxic substances are not accompanied with equivalency factors. This limitation is usually addressed in LCIA methods in two ways, assuming that the substance has the same hazard potential as the reference chemical (equivalency factor =

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[1] For a non-homogeneous impact category, the aggregation of the toxic substances is not allowed because an adverse impact on human health is caused by a specific toxic substance and this substance is not interchangeable with other substances.

[2] By using the carrying capacity estimates in this research, the STM does not need the characterization of toxic substances because, ideally, the toxic substances are considered as the subcategories of the human and eco-toxicity impacts and their individual carrying capacity estimates are available.

1) or the substance does not pose any hazard (equivalency factor = 0). Either way may cause a biased result if the emission of the toxic substance is significant.

In the STM exercise in the household coffee maker case study, a number of carrying capacity values for impact subcategories are not available. For example, about one third of the environmental emissions and consumptions associated with the manufacturing of the household coffee maker in LCI are the emissions of radioactive constituents. At this stage, the carrying capacity for the emission of radioactive constituents has not been evaluated. This limitation may be addressed by assuming that the carrying capacity of these subcategories is unlimited, i.e., the environmental impact (EI) is insignificant. This assumption can cause a biased result as well if the actual carrying capacity of the subcategory is very small (should it be available). As a result, the cause of the unavailability of a carrying capacity estimate should be identified as either because the toxic substance is considered safe or the information is incomplete for the carrying capacity evaluation. Furthermore, an update for the missing carrying capacity values should be made periodically.

Again, in this case study as well as the first two case studies, the human toxicity and eco-toxicity are not significant in the assessment. This may also be due to the limitation of the carrying capacity estimate mentioned earlier that there is still a small number of carrying capacity estimates for chemicals in both human toxicity and eco-toxicity impact categories. As a result, a significant impact may not be observed for some chemicals without carrying capacity values.

However, in this case study as well as the first two case studies, an average value from the carrying capacities available in human toxicity and eco-toxicity is used as the default value for the chemicals for which the threshold levels are not available for carrying capacity estimation. This causes the assessments to be conservatively biased.

It should be emphasized here that the environmental performance evaluation using different LCIA methods result in different results and interpretations. Hence, in the environmental performance evaluation of a system, it is necessary that the method used, limitations, and assumptions made must be clearly stated.

13.5 Carrying Capacity Estimates for Europe

The objectives of estimating the carrying capacity for Europe are twofold: 1) to verify the feasibility of using the STM for other regions of the world based on local conditions, 2) to examine the validity of the techniques, approaches, and methodologies for the carrying capacity estimate in this research. Europe may be a secondary target to which the STM can be applied. Therefore, the European-based carrying capacity is estimated and presented here. Impact categories taken into account are: global warming, stratospheric ozone depletion, acidification, eutrophication, and photochemical ozone formation. The carrying capacity for these impact categories can be estimated using the same techniques and approaches as for the U.S. given the available European data.

To accomplish the second objective, the European-based carrying capacity estimates were compared to two concepts similar to the carrying capacity approach, the EDIP "environment's carrying capacity" and the EI95 "target values". For convenience, the carrying capacity for resource depletion, human toxicity, and eco-toxicity impacts are not evaluated here because of limited information and resources. Furthermore, EI95 does not provide the information to estimate the "target values" for these impact categories.

However, potential sources of reference and approach for the carrying capacity evaluation for these three impact categories are discussed.

13.5.1 Carrying Capacity Estimates

Global Warming Impact

The means for estimating the regional carrying capacity is not presented in this research. The carrying capacity is determined from the most conservative scenario for global greenhouse gas emissions. However, since the carrying capacity of the global warming impact is considered at the global scale, therefore, the carrying capacity of 2.93×10^{13} kg CO_2-eq/yr (year 2000) is also used for Europe.

Stratospheric Ozone Depletion Impact

As in the global warming impact, the means for estimating regional carrying capacity is not presented in this research and the carrying capacity is estimated for the global scale only. Therefore, the global scale carrying capacity of 3.48×10^{8} kg CFC11-eq/yr (year 2000) is also used for Europe.

Acidification

The carrying capacity can be estimated using the critical load approach. The pentile (5%) maximum critical load of 200 eq/ha/yr (3.2 kg S/ha/yr) is selected (Posch et al. 1997). Given an area for Europe of 4.87×10^{6} km^2 (CIA 2003), the carrying capacity can be estimated as 1.56×10^{9} kg S/yr (3.12×10^{9} kg SO_2-eq/yr).

Eutrophication

The carrying capacity can also be estimated using the critical load approach. The pentile critical load of 400 eq/ha/yr (14 kg N/ha/yr) is selected (Posch et al. 1997). Given the area for Europe, a total emission carrying capacity can be estimated as 2.73×10^{9} kg N/yr.

Photochemical Ozone Formation

The carrying capacity can be estimated from critical VOC and NO_x emission rates. However, the U.S. critical VOC and NO_x emission rate was derived from OZIPR model using the U.S. cities' meteorological conditions. The derivation of critical VOC and NO_x emission rates in the same manner for Europe cannot be prepared due to unavailability of information. At this stage the U.S. average critical VOC and NO_x emission rates are used to determine the European-based carrying capacity. By using a simple correlation between U.S. and European-based carrying capacity estimates as expressed in Equation (13.1), the European-based carrying capacities are presented in Table 13.27.

$$\text{Europe CC} = \text{U.S. CC} \times \frac{\text{Europe land area}}{\text{U.S. land area}} \tag{13.1}$$

Table 13.27 European-Based Carrying Capacities for Photochemical Ozone Formation

Primary pollutant	Carrying capacity estimate	
	Old ozone standard as the threshold (1-hr 0.12 ppm)	New ozone standard as the threshold (8-hr 0.08 ppm)
VOC emission (kg C_2H_4-eq/yr)	7.26E+11	5.84E+11
NO_x emission (kg NO_x/yr)	5.66E+11	5.09E+11

It should be noted that there is a difference in threshold levels for ambient ozone in U.S. and Europe. In U.S., NAAQS (National Ambient Air Quality Standards) for ozone is 1-hr 0.12 ppm and 8-hr 0.08 ppm while the European standard is 8-hr 0.11 ppm (RIVM 1998). Use of the carrying capacity value estimated here may be biased conservatively.

Human Toxicity Impact and Eco-Toxicity Impact

The carrying capacity for human toxicity and eco-toxicity impacts can be calculated using the empirical correlation derived from the threshold-oriented technique (Equation (10.5) to Equation (10.7) in Chapter 10). The European-based carrying capacities can also be calculated using Equation (13.1).

However, the carrying capacity for both human toxicity and eco-toxicity impacts are evaluated using the CalTOX model, for which some input parameters are U.S.–based. The use of Equation (13.1) is based on the assumption that the generic conditions for environmental fate and transport of toxic substances are applied for both U.S. and Europe. A more refined evaluation can be carried out using parameters for Europe or employing a European environmental fate model.

Threshold concentrations for Europe may be used instead of the regulatory levels and PNECs used in this research. The European threshold concentrations can be obtained from the international and European toxicity database and other sources (WHO 1999; OECD 2000; EC 2002; ECETOC 2002; ECB 2002; IPCS 2002).

Resource Depletion Impact

The European carrying capacity for resource depletion impact is not presented at this stage due to the unavailability of information (e.g., existing reserves). However, the information on resource reserves for Europe is available commercially (BGS 2001). Upon its availability, the European carrying capacity can be estimated in a further stage using the resource availability/time horizon concept. However, the carrying capacity for the resource consumption in the global scale can be applied for STM implementation as well (Table 12.2, Chapter 12). For water consumption, the carrying capacity for European countries is the freshwater recharge rate that can be obtained from WRI (1999).

13.5.2 Comparison and Discussion

This subsection has presented the European-based carrying capacity estimate for five impact categories: global warming impact, stratospheric ozone depletion impact, acidification impact, eutrophication impact, and photochemical ozone formation impact (Table 13.28). The approaches for other impacts are summarized, but the carrying capacity estimates are not provided.

Table 13.28 European-Based Carrying Capacity Estimated Using the Technique and Approaches Developed in This Research

Impact category	Scale	Carrying capacity	Unit
Global warming	Global	2.93E+13	kg CO_2-eq/yr
Stratospheric ozone depletion	Global	3.48E+08	kg CFC-11-eq/yr
Acidification	Europe	3.12E+09	kg SO_2-eq/yr
Eutrophication	Europe	2.73E+09	kg N-eq/yr
Photochemical ozone formation (1-hr 0.12 ppm)	Europe	7.26E+11 5.66E+11	kg C_2H_4-eq/yr kg NO_x/yr
Resource depletion	Europe	not determined	
Human toxicity	Europe	not determined	
Eco-toxicity	Europe	not determined	

The European-based carrying capacity for some impact categories estimated in this research can be compared to two similar concepts to the carrying capacity, the EDIP' "carrying capacity" and the EI95's "target values". In the EDIP, Hauschild and Wenzel (1998) evaluated a group of "environment's carrying capacity" as the reference potential impact to calculate the normalized environmental potential (see also Subsection 2.3.4) for some impact categories. The reference potential impact was calculated from the carrying capacity normalized by size of population (e.g., kg CO_2-eq/person/year in 1990). In their work, the environmental carrying capacity was evaluated for various sources and methods, e.g., the carrying capacity for the emission of CO_2 and other greenhouse gases was estimated from the absorption capacity of major sinks. However, the carrying capacity estimates in EDIP are basically derived using the damage-oriented approach. Table 13.29 summarizes the EDIP's carrying capacity for four impact categories that are similar in concept to the carrying capacity discussed in this dissertation.

Table 13.29 EDIP's Carrying Capacity
(Source: Hauschild and Wenzel 1998)

Impact category	Method	Scale	Carrying capacity	Unit[1]
Global warming	Natural removal capacity	Global	2.0E+13	kg CO_2-eq/yr
Stratospheric ozone depletion	Natural removal capacity Sustainable level	Global Global	4.4E+07 2.8E+08	kg CFC11-eq/yr kg CFC11-eq/yr
Acidification	Interpolation from pre-damaged level Interpolation from pre-damaged level	Denmark Europe	6.2E+06 2.4E+09	kg SO_2-eq/yr kg SO_2-eq/yr
Nutrient enrichment	Interpolation from pre-damaged level Interpolation from pre-damaged level	Denmark Denmark	3.0E+07 5.7E+05	kg N/yr kg P/yr

EI95's target values for Europe were evaluated by dividing the "normalization values" by the "reduction factors". The normalization values were the existing (1990) anthropogenic emission of pollutants. The emissions were normalized and aggregated within the same impact categories: the greenhouse effect, ozone layer depletion, acidification, eutrophication, heavy metals, carcinogens, winter smog, summer smog, and pesticides. Some extrapolations were made when impact data for some countries were missing (Goedkoop 1998). The extrapolation was based on the country's energy consumption, which reflects the emission pattern. The reduction factors were determined based on several sources of information and criteria, some of which were derived with a subjective judgment (discussed in Subsection 2.5.2). As also pointed out by Goedkoop (1998), some reduction factors contained a high degree of uncertainty. The target values derived for EI95 are presented in Table 13.30.

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[1] CO_2-eq is the same as GWP, CFC11-eq is the same as ODP, and SO_2-eq is the same as AP.

Four values of the estimated European-based carrying capacity can be compared to the EDIP carrying capacity: global warming, ozone depletion, acidification, and nutrient enrichment (eutrophication). Meanwhile, three values of the estimated European-based carrying capacity can be compared to the EI95 target values: acidification, eutrophication, and photochemical ozone formation (Table 13.31).

Table 13.30 Eco-Indicator 95 Target Values
(Source: Goedkoop 1998)

Effect	Scale	Target value	Unit[1]
Greenhouse effect	Europe	2.60E+12	GWP kg/yr
Ozone layer depletion	Europe	4.60E+06	ODP kg/yr
Acidification	Europe	5.60E+09	AP kg/yr
Eutrophication	Europe	3.80E+09	NP kg/yr
Heavy metals	Europe	5.40E+06	Pb-eq kg/yr
Carcinogens	Europe	5.40E+05	PAH-eq kg/yr
Winter smog	Europe	9.40E+09	SO_2-eq kg/yr
Summer smog	Europe	3.56E+09	POCP kg/yr
Pesticides	Europe	1.92E+07	Active ingredient Kg/yr

Table 13.31 Comparison of Some European-Based Carrying Capacity Values Estimated in This Research and the Values in Other Studies.

Impact category	Scale	Carrying capacity in this research	EDIP Carrying capacity	EI95 Target values	Unit
Global warming	Global	2.93E+13	2.0E+13	-	kg CO_2-eq/yr
Stratospheric ozone depletion	Global	3.48E+08	4.4E+07 [2]	-	kg CFC11-eq/yr
			2.8E+08 [3]		
Acidification	Europe	3.12E+09	2.4E+09	5.60E+09	kg SO_2-eq/yr
Eutrophication	Europe	2.73E+09 kg N/yr	4.9E+08 [4]	3.80E+09	kg PO_4^{3-}-eq/yr
			6.8E+07 [5]		
Photochemical ozone formation	Europe	7.26E+11	-	3.56E+09	kg C_2H_4-eq/yr
		5.66E+11	-	-	-

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[1] GWP = Global warming potential (CO_2-equivalent), ODP = Ozone depleting potential (CFC11-eq.), AP = Acidification potential (SO_2-eq.), NP = Nutrification potential (PO_4^{3-}-eq.), PAH = Polyaromatic hydrocarbon, POCP = Photochemical ozone creation potential (C_2H_4-eq.)

[2] Estimated from the natural removal capacity

[3] Estimated from the sustainable level

[4] For nitrogen nutrient

[5] For phosphorus nutrient

The comparison indicates that the European-based carrying capacity values estimated in this research are similar to the EDIP carrying capacity values for global warming (2.93×10^{13} compared to 2.0×10^{13} kg CO_2-eq/yr), stratospheric ozone depletion (3.48×10^{8} compared to 2.8×10^{8} kg CFC11-eq/yr), and acidification (3.12×10^{9} compared to 2.4×10^{9} kg SO_2-eq/yr). For the eutrophication impact, the European-based carrying capacity estimate in this research is 2.73×10^{9} kg N/yr. For comparison purposes, this value can be converted using nutrition potential (0.42 kg PO_4^{3-}/kg N as in Hauschild and Wenzel (1998) to become 1.15 $\times10^{9}$ kg PO_4^{3-}-eq/yr. This carrying capacity is approximately two times that of the carrying capacity for nitrogen nutrient (4.9×10^{8} kg PO_4^{3-}-eq/yr) and 17 times the carrying capacity for phosphorus nutrient (6.8×10^{7} kg PO_4^{3-}-eq/yr) in EDIP. Overall, the European-based carrying capacity values estimated in this research are relatively similar to the EDIP carrying capacity values because both methods are using the same damage-oriented approach.

When compared to the European-based carrying capacity values estimated in this research with the EI95 target values, it was found that the carrying capacity estimates are similar to the target values for acidification and eutrophication impacts even though they were evaluated using different approaches. For the acidification impact, the carrying capacity is 3.12×10^{9} kg SO_2-eq/yr compared to the target value of 5.6×10^{9} kg SO_2-eq/yr. For the eutrophication impact, the carrying capacity is 1.15×10^{9} kg PO_4^{3-}-eq/yr compared to the target value of 3.8×10^{9} kg PO_4^{3-}-eq/yr.

However, the VOC emission carrying capacity for the photochemical ozone formation impact is approximately two orders of magnitude larger compared to the target value (7.26×10^{11} kg C_2H_4-eq/yr for carrying capacity and 3.56×10^{9} kg POCP/yr for target value). This discrepancy between the two figures is a result of the approach used. As previously stated, the EI95 target value was derived from the existing emission rate divided by the reduction factor, which is 2.5 for summer smog (photochemical ozone formation). To verify the validity of the existing VOC emission rate for Europe used in EI95, the existing VOC emission rate is converted to a per unit area basis so it can be compared to that of the U.S. The existing European VOC emission of 8.9×10^9 kg POCP/yr (in EI95) is then divided by the land area for Europe of 4.87×10^6 km^2 to obtain a VOC emission rate of 1,828 kg POCP/km^2/yr or 4,221 kg VOC/km^2/yr. This European VOC emission rate per unit area of 4,221 kg VOC/km^2/yr is about two times greater than the U.S. emission per unit area (1,796 kg VOC/km^2/yr), which is calculated from the U.S. emission rate for the year 1999 (EPA 2000a). This comparison illustrates that the existing European VOC emission rate used in EI95 is reasonable because it is in the same range as that of the U.S. Therefore, it can be concluded that the significant gap between the carrying capacity estimate and the target values is not because of the difference in existing emission rates. The discrepancy may be due to the difference in calculations.

The carrying capacity for photochemical ozone formation impact is carried out in the opposite way to that of the target value. The carrying capacity is derived from the existing emission (for which the threshold level for photochemical ozone is not reached) multiplied by a factor to obtain a maximum allowable emission (for which the threshold level is just reached). In most cases, the carrying capacity is higher than the existing emissions. This procedure is also an application of the threshold-oriented technique. This is contrasted with the EI95 calculations, in which the target values are lower than the existing emissions by applying reduction factors. Equation (13.2) and (13.3) depict the basic concepts of the photochemical ozone formation carrying capacity estimate in this research and the summer smog target value estimate in EI95.

$$CC = \text{Existing Emission} \times \text{Multiplication Factor} \qquad (13.2)$$

$$\text{Target value} = \frac{\text{Existing Emission}}{\text{Reduction Factor}} \qquad (13.3)$$

Basically, the existing emission rates used in the carrying capacity estimate are in the range of 1.2 to 24.3 kg VOC/km^2/hr (Table 9.2, Chapter 9) or 10,512 to 212,868 kg VOC/km^2/yr. This range is much greater than the average U.S. emission rate of 1,796 kg VOC/km^2/yr. However, the emission rates used in the U.S. carrying capacity estimate are the emission rates in the urban areas of ten major cities with high traffic volume and a large number of VOC emissions. Therefore, this high emission rate is assumed to be reasonable for carrying capacity estimation. The application of a multiplication factor is also reasonable, since photochemical ozone is not formed in most of the areas within the range of the existing emissions. Furthermore, the carrying capacity is conservatively biased because it is assumed that nitrogen oxides are readily available for ozone formation reaction (as discussed in Chapter 9). In most areas, the presence of nitrogen oxides is not high enough to cause the formation of photochemical ozone.

For Europe, the existing VOC emissions of 8.9×10^9 kg POCP/yr divided by the reduction factor of 2.5 equals the target value of 3.56×10^9 kg POCP/yr. Using this calculation, the target value is biased considerably more conservatively than the carrying capacity estimate because summer smog does not occur at the existing emission rates, hence the target value is lower.

13.6 STM Implementation Using the Modified Target Values

13.6.1 U.S. Target Values Modified from EI95 Target Values

EI95's target values for seven impact categories were modified by Dickinson et al. (2001) to become a preliminary U.S. carrying capacity (U.S. target values hereafter). These U.S. target values were estimated by normalizing EI95 target values using the European land area, which is 4.43×10^6 km^2, to yield the target values per unit area. Therefore the U.S. target values were obtained by simply multiplying the normalized EI95 target values by the U.S. land area, which is 9.36×10^6 km^2. Table 13.32 presents the U.S. target values modified from EI95 European target values.

The U.S. target values from Table 13.32 can be compared with the U.S. carrying capacity estimates developed in this research for the following impact categories: acidification, eutrophication, and photochemical ozone formation. The comparison may yield the same results as in Subsection 13.5.2 because both comparisons are carried out between the U.S. carrying capacity estimates using the methods developed in this research and the U.S. target values modified from EI95 target values. The U.S. carrying capacity estimates using the methods in this research are similar to the U.S target values for acidification and eutrophication impacts, even though they were evaluated using different approaches (Table 13.33). Similar to the comparison in Subsection 13.5.2, the carrying capacity of VOC emission for photochemical ozone formation impact using the method in this research is approximately two orders of magnitude greater when compared to the summer smog target value. The significant gap between the two figures is the result of the methods used as discussed in the Subsection 13.5.2.

Table 13.32 U.S. Target Values Modified from EI95 Target Values
(Source: Dickinson et al. 2001 and Goedkoop 1998)

Impact category	1990 European emission	Unit	Reduction factor	European target value	U.S. target value	Unit
Greenhouse effect	6.50E+12	GWP kg	2.5	2.60E+12	5.49E+12	GWP kg/yr
Ozone layer depletion	4.60E+08	ODP kg	100	4.60E+06	9.72E+06	ODP kg/yr
Acidification	5.60E+10	AP kg	10	5.60E+09	1.18E+10	AP kg/yr
Eutrophication	1.90E+10	NP kg	5	3.80E+09	8.03E+09	NP kg/yr
Heavy metals	2.70E+07	Pb eq. kg	5	5.40E+06	1.14E+07	Pb eq. kg/yr
Carcinogens	5.40E+06	PAH eq. kg	10	5.40E+05	1.14E+06	PAH eq. kg/yr
Winter smog	4.70E+10	SO_2 eq. kg	5	9.40E+09	1.99E+10	SO_2 eq. kg/yr
Summer smog	8.90E+09	POCP kg	2.5	3.56E+09	7.52E+09	POCP kg/yr
Pesticides	4.80E+08	Act.ing. kg	25	1 92E+07	4.06E+07	Act.ing. kg/yr

Table 13.33 Comparison of U.S. Target Values Modified from EI95 Target Values and U.S. Carrying Capacity Estimated in This Research

Impact	U.S. Target Value (Dickinson et al. 2001)	Unit	Carrying Capacity (This research)	Unit
Acidification	1.2E+10	AP kg/yr	3.04E+10	kg SO_2/yr
Eutrophication	8.0E+09	NP kg/yr	9.96E+08	kg PO_4^{3-}/yr
Summer smog -VOC (Photo. ozone formation)	7.5E+09	POCP kg/yr	1.54E+12	kg CH_4/yr

13.6.2 STM Implementations Using the Modified Target Values

To investigate the application of U.S. target values in the STM, all three case studies are assessed. The U.S. target values discussed in the previous subsection are used in place of the carrying capacity as the reference levels in the STM. The number of impact categories taken into account is the same as that of EI95. These impact categories are: greenhouse effect, ozone layer depletion, acidification, eutrophication, heavy metals, carcinogens, winter smog, summer smog, and pesticides. It should be recalled that resource depletion impact is not considered in the EI95. As a result, a target value for resource depletion is not available.

The LCI data for all three case studies (energy generation sources, basic material productions, and household coffee maker) were also obtained from the SimaPro LCA software. However, a further calculation for characterization is not needed since the characterized impacts can be analyzed by EI95, which is integrated in the SimaPro software. Next, the characterized impacts are assessed by the STM in conjunction with the U.S. target values. The key parameters obtained from the assessment can be compared to those of the assessment using the carrying capacity evaluated in this research. Table 13.34 is the comparison of the assessment for all three case studies.

Table 13.34 Comparison of Key Parameters from the STM Implementations Using the Target Values and the Carrying Capacity for the Case Studies

Energy generation sources				
	Coal	Gas	Lignite	Oil
Rank				
Carrying capacity	1	3	2	4
Target values	4	1	2	3
EI$_{PR}$ (kWhe/yr)$^{-1}$				
Carrying capacity	2.4E-08	4.6E-07	2.2E-07	5.9E-07
Target values	5.8E-13	2.4E-13	4.9E-13	5.6E-13
EE (%)				
Carrying capacity	2.9E+00	1.5E-01	3.2E-01	1.2E-01
Target values	1.2E+05	3.0E+05	1.4E+05	1.5E+05
Major impact				
Carrying capacity	Acidification	Resource depletion	Resource depletion	Resource depletion
Target values	Acidification	Summer smog	Acidification	Acidification
Basic material productions				
	ABS	Aluminum	Copper	Steel
Rank				
Carrying capacity	2	4	3	1
Target values	1	3	4	2
EI$_{PR}$ (kg/yr)$^{-1}$				
Carrying capacity	2.2E-05	1.9E-03	2.6E-04	1.2E-06
Target values	3.3E-12	3.9E-11	9.4E-11	4.1E-12
EE (%)				
Carrying capacity	3.4E-02	8.0E-04	5.9E-03	3.5E-02
Target values	2.3E+05	3.6E+04	1.6E+04	9.7E+04
Major impact				
Carrying capacity	Resource depletion	Resource depletion	Resource depletion	Resource depletion
Target values	Acidification	Carcinogens	Acidification	Heavy metals
Household plastic coffee maker				
EI$_{PR}$ (unit/yr)$^{-1}$				
Carrying capacity	2.4E-04			
Target values	2.0E-11			
EE (%)				
Carrying capacity	8.0E-02			
Target values	9.8E+05			
Major impact				
Carrying capacity	Resource depletion			
Target values	Heavy metals			

For the case study of energy generation sources and basic material productions, the results indicate that the environmental performance of both cases assessed using the target values are ranked in a different way compared to the previous assessment using the carrying capacity developed in this research. By using the target values, gas is the most favorable energy generation source followed by lignite, oil, and coal. This ranking is also different from the one assessed by EI95 (gas, oil, lignite, and coal). The Environmental Impact (EI_{PR}) obtained from the assessment using the target values is five to six orders of magnitude smaller than those of the assessment using the carrying capacity. On the other hand, the Eco-Efficiency (EE) from the assessment using the target values is five to six orders of magnitude greater than those of the assessment using the carrying capacity.

A similar trend is observed from the case study of basic material productions. By using the target values, ABS plastic is the best alternative followed by steel, aluminum, and copper. This ranking is the same as the one assessed by EI95. The Environmental Impact (EI_{PR}) obtained from the assessment using the target values is six to eight orders of magnitude smaller than those of the assessment using the carrying capacity. The Eco-Efficiency (EE) from the assessment using the target values is five to seven orders of magnitude greater than those of the assessment using the carrying capacity.

The results indicate a very sustained environmental performance of both energy generation and basic material production case studies when using the U.S. target values in the assessments. The big difference on the results from both assessments (using target values and the carrying capacity) is probably caused by the difference of impact categories taken into account. It should be recalled that the target value for resource depletion is not available while the assessment using the carrying capacity reveals that resource depletion impact is the most critical impact for most alternatives in both case studies. In the absence of the resource depletion impact, the primary contribution to the environmental impact is a shift to the following critical impact, which is probably acidification for both cases.

A similar situation is also observed in the household coffee maker case study, where the results indicate that the Environmental Impact (EI_{PR}) obtained from the assessment using the target values is seven orders of magnitude smaller than that of the assessment using the carrying capacity. The Eco-Efficiency (EE) from the assessment using the target values is seven orders of magnitude greater than that of the assessment using the carrying capacity. The assessment using the target values indicates a sustainable environmental performance while the result obtained from the assessment using the carrying capacity indicates that the system is not sustainable. The critical impacts of both assessments are also different. Resource depletion is the most critical impact for the assessment using the carrying capacity compared to heavy metals in the assessment using the target values.

The results from this application emphasize the importance of the impact categories that are taken into consideration. As illustrated, an absence/presence of a key impact category may direct the result in the opposite way. In addition, the indicator values can be changed several orders of magnitude due to the change in the number of impact categories as well as the carrying capacity values. It is obvious that the completeness of impact categories taken into account and the accuracy of carrying capacity values play an important role in order to address the problem of uncertainty associated with value judgment of the STM application.

13.7 Summary

This chapter demonstrated the application of the U.S. carrying capacity in conjunction with the STM in three case studies following the LCA framework. The object of the application are the electrical energy generation sources representing the process level, the basic material productions representing the supply-line level, and a household plastic coffee maker representing the product level with temporal and spatial variations in the production process. By using the STM in conjunction with the carrying capacity estimates, superiority among alternatives in terms of environmental performance can be identified. The critical impact associated within a system can also be identified. The environmental performance evaluations of the case studies have been compared to those evaluated by EI95, EI99, EPS, and EDIP. The case study of the coffee maker is an attempt to demonstrate the capability of the STM coupled with an appropriate set of carrying capacities to assess the environmental impact of a system that has temporal and spatial variations.

A limitation that is observed from the coffee maker case study is the limited number of carrying capacity values, which is due to the unavailability of some necessary information. To promote the universal use of the STM procedure, carrying capacity values covering all of the environmental emissions and consumptions in the LCI database should be made available as much as possible. Further development of carrying capacity evaluation is surely possible, especially for human and eco-toxicity impacts. In all three case studies, the human toxicity and eco-toxicity are not significant in the assessment. This may be due to the small number of carrying capacity estimates for chemicals in both human toxicity and eco-toxicity impact categories.

By comparing the results of the case studies using the STM to those using the other LCIA methods, it was found that the number of impact categories taken into consideration plays a significant role. The trends obtained from the STM exercises are similar to those obtained from other LCIA methods when only the common impact categories are used and the results are different when using all of the impact categories available. The impact category that is significant is the resource depletion impact, more specifically, the resource availability. It has been demonstrated that impact in terms of resource availability is the major contribution to the overall impact in most case studies and, as a result, it is too important to be omitted in an LCA assessment. This can also lead to a conclusion that the STM in association with the carrying capacity estimated in this research is a resource depletion-driving metric. The STM in conjunction with the carrying capacity estimates will be useful for an LCA of a resource-intensive system.

This chapter also demonstrated the application of the techniques and approaches developed in this research to evaluate the carrying capacity for Europe. It can be concluded that the techniques and approaches for carrying capacity evaluation developed in this research can be applied for regions other than U.S. provided that similar scientific information is available.

An examination is conducted to verify the importance of selected impact categories and accuracy of carrying capacity values. This examination is made by applying a set of preliminary U.S. carrying capacities (so-called U.S. target values) estimated from EI95 target values in an STM exercise. The results are compared to those using the U.S. carrying capacity developed in this research. It was found that a complete list of impact categories and the accuracy of carrying capacity values are the key factors that can increase the credibility and applicability of the STM.

The carrying capacity estimates presented in this research are based on the available scientific techniques, approaches, and information. The carrying capacity estimates can be modified and revised periodically. This should be done to ensure that the STM will be a generic environmental performance metric. Feedback from users will provide the limitations and the modifications needed. Future research on the STM and the carrying capacity development may focus on additional impact categories and the accuracy of the carrying capacity estimates.

There are a number of applications that can result from the carrying capacities evaluated in this research. Furthermore, this carrying capacity approach is also a reliable alternative that can be used as a sustainability metric as well as for LCA purposes. By using an appropriate set of carrying capacity estimates, the STM can be applied in the LCA to become a framework for using the precautionary principle approach. The application of the carrying capacity and the STM in this research is hopefully a useful contribution to LCA development.

CHAPTER 14

SENSITIVITY AND UNCERTAINTY ANALYSIS

14.1 Overview

Sensitivity analysis is a procedure for estimating the effects on the outcome of a study of the chosen methods and data as a result of changes in some of the key variables involved in the study (ISO 1998a). A sensitivity analysis quantifies the effects of a change of one parameter on a variable (Binder et al. 1997; Bjorklund 2002). The purpose of the sensitivity analysis in this chapter is to refine the critical assumption that the advance of carrying capacity estimates is significant in the performance of the STM. Essentially, this exercise is used to evaluate the impact on the performance of the STM of a marginal change in the carrying capacity estimates. Two sets of numerical sensitivity analyses are examined for the case studies assessed in Chapter 13. The first set is a one-way sensitivity analysis applied for all three case studies. This analysis is used to investigate the impact caused by a marginal change in the carrying capacity estimate of individual impact categories on the STM's overall absolute indicator, the EE. An EE result that is sensitive to marginal changes in carrying capacity values would demonstrate that the accuracy of carrying capacity estimates plays an important role in the STM performance.

The second set is a ratio sensitivity analysis applied for the case studies of electrical energy generation sources and basic material production. This analysis is used to investigate the impact on ranking of alternatives in terms of environmental performance caused by a marginal change in the carrying capacity value for individual impact categories. The results of this analysis would also emphasize the significance of advanced carrying capacity estimates.

Similarly, the carrying capacity estimates cannot be validated without an uncertainty analysis. Uncertainty analysis, in the latter part of this chapter, presents causes and possible sources of uncertainty that are associated with the carrying capacity estimates. Qualitative uncertainty is assessed on the impact category basis. The conclusions of both the sensitivity and the uncertainty analysis are also presented.

14.2 Sensitivity Analysis

The quality of an LCA is only as good as the quality of information upon which it is based (EPA 1995b). Data quality assessment can be applied to the primary and secondary data in both the inventory analysis and the impact assessment steps. Sensitivity analysis, as a method for data quality assessment, is used in this research to investigate the impact on the performance of the STM caused by a marginal change in carrying capacity estimates. According to Bjorklund (2002), sensitivity analysis methods that could be used in the LCA context are: a tornado diagram, a one-way sensitivity analysis, a scenario analysis, a factorial design and multivariate analysis, a ratio sensitivity analysis, and a critical error factor. One-way sensitivity analysis and ratio sensitivity analysis are two methods that are suitable for the purposes of this investigation.

14.2.1 One-Way Sensitivity Analysis

The STM is comprised of a set of mathematical calculations functioning in the form of simple mathematical operations. A carrying capacity is employed to calculate the Impact Reference levels (I_R) in one of these calculations. The STM indicators are then calculated from the I_R. Therefore, a simple sensitivity analysis can be made without any demonstration of numerical calculation if an assessment consists of only a single impact category. In this case the STM indicators will be varied in the same ratio as the marginal change in the carrying capacity estimate of the only impact category. For example, if the carrying capacity estimate of the only impact category is increased by 10%, the Environmental Impact (EI) will be decreased by 10% while the Resource Productivity (RP), the Value Productivity (VP), and the Eco-Efficiency (EE) will be increased by 10% as well, and vice versa for the decreasing of the carrying capacity estimate. This rationale is also valid if all of the carrying capacity estimates in all impact categories are increased or decreased by the same percent should the assessment consist of multiple impact categories. However, the impact due to the increase in individual carrying capacity estimates cannot be analyzed without a demonstration using numerical calculation because the number of impact subcategories is not limited (e.g., human toxicity and eco-toxicity impacts can consist of multiple impact subcategories due to the number of chemical releases). In this case, a one-way sensitivity analysis needs to be carried out using a numerical set from a case study.

One-way or single system sensitivity analysis determines the importance of an individual input parameter for the overall model results (EPA 1995b). The sensitivity of an individual parameter relative to the model results is determined by calculating the amount an individual parameter would need to change, so that the model results are changed by a given ratio or percentage. The change in the model results can also be calculated from the change in an individual parameter by a given ratio or percentage.

First, the case study of energy generation sources (Section 13.2, Chapter 13) is revisited. A one-way sensitivity analysis for four electrical energy generation sources (coal, gas, lignite, oil) is carried out individually. Individual carrying capacity estimates are varied from ratios between 0.0001 and 10,000 times the estimated values of the carrying capacity (base case carrying capacity). For instance, a 0.0001×base case CC means 0.01 kg/yr compared to the base case carrying capacity of 100 kg/yr. Analogously, a 0.1×base case EE means 10% compared to the base case EE of 100%. The overall Eco-Efficiency (EE) is selected as the surrogate indicator for sensitivity analysis, noting that the change in RP and VP will be in the same ratio as that of EE. A changed EE is also presented proportionally to the base case EE.

The results are presented in Figure 14.1 for the analyses related to electrical energy generation from coal, gas, lignite, and oil. From these analyses, the relationship between the carrying capacity estimates and EE are observed as S-shape curves. A sharply inclining, or a steep, S-curve indicates a greater sensitivity of EE, which is related to the marginal change in the carrying capacity estimate. In other words, the sensitivity of the carrying capacity is due to the weight of an impact. EE is more sensitive to marginal change in the carrying capacity estimate of the impact that dominates the STM results than to those impacts that are not significant.

For example, a marginal change in the carrying capacity estimate for the resource depletion impact results in a change in EE in a higher ratio than other impacts in all energy generation using sources. This is because resource depletion is the most critical impact in these energy generation sources. For coal, EE does not vary significantly if the carrying capacity estimate is greater than ten times (i.e., one order of magnitude greater) of the base case carrying capacity estimate for the resource depletion impact. Also for coal, impacts that dominate the STM results next to the resource depletion impact are acidification, global warming, eutrophication, human toxicity, photochemical ozone formation, and eco-toxicity respectively. It should be recalled that none of the energy generation sources affect the stratospheric ozone depletion impact in the STM implementation; hence, sensitivity analysis for this impact is not carried out here.

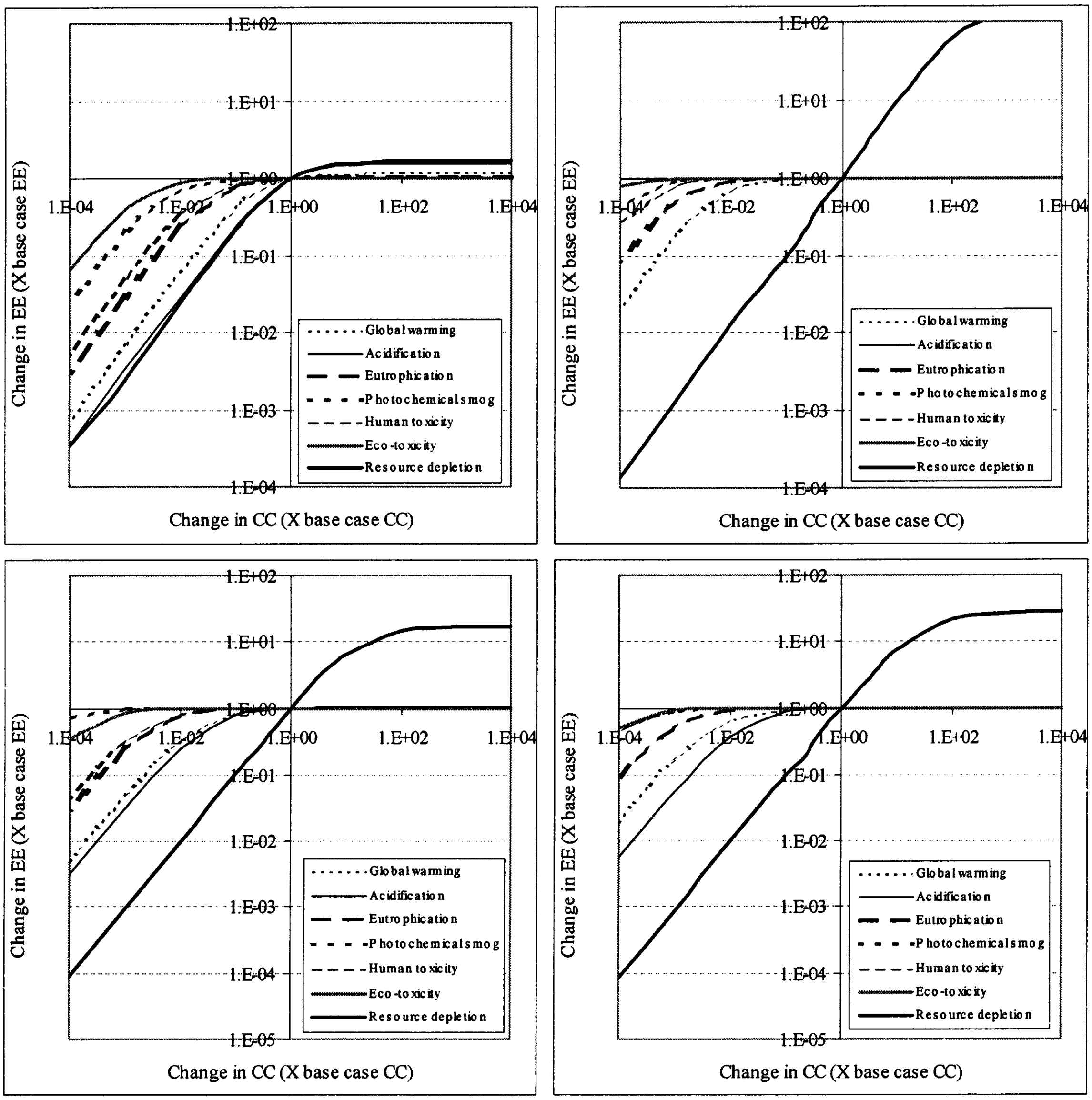

Figure 14.1 Effects of marginal change in carrying capacity on Eco-Efficiency (EE) for electrical energy generation using coal (top left panel), gas (top right panel), lignite (bottom left panel), and oil (bottom right panel).

Eco-toxicity is the impact that is less significant next to ozone depletion. The significance of the marginal change in carrying capacity estimate for eco-toxicity impact is not observed (flat line) when the carrying capacity is varied by less than two orders of magnitude smaller than the base case carrying capacity. The carrying capacity for eco-toxicity must be lowered by at least two orders of magnitude in order to alter EE.

For energy generation sources other than coal, the curves for resource depletion obviously show a steeper "S" than other impacts (Figure 14.1). These curves indicate that resource depletion has a much more significant impact than the others. The asymptote of the resource depletion curve cannot be seen for energy generation using gas (Figure 14.1 top right panel) but it is shown separately in Figure 14.2, which presents only the resource depletion curves for all energy generation sources.

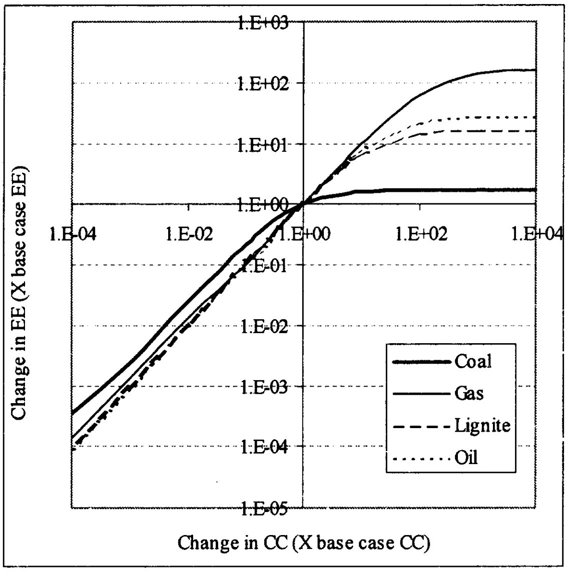

Figure 14.2 Effects of marginal change in carrying capacity of resource depletion on Eco-Efficiency. (Energy generation sources)

From this figure, the asymptotes of the resource depletion curves for coal, gas, lignite, and oil are reached at 1.67, 155, 16.75, and 27.8 times the base case EE respectively. The flat part (asymptote) of these curves implies that the change in EE will be constant regardless of any increasing carrying capacity estimates. It should be noted that decreasing of the carrying capacity affects the overall EE more than does an increase. A smaller carrying capacity value results in a larger EE value. One-way sensitivity analysis is also extended to the case studies of basic material production and the household plastic coffee maker (Figure 14.3, Figure 14.4, and Figure 14.5).

14.2.2 Ratio Sensitivity Analysis

The one-way sensitivity analysis provides results in terms of changes in the STM indicators caused by changes in the carrying capacity estimates within the assessment of a product, an alternative, or a system. Sensitivity analysis can also be extended to investigate the reverse ranking between a pair of alternatives due to a marginal change in carrying capacity estimates. This can be carried out using a ratio sensitivity analysis. Basically, ratio sensitivity analysis is applicable for comparative assessment (EPA 1995b; Bjorklund 2002). Rather than varying individual parameters one at a time to determine the impact on model results, two alternatives can be compared by a ratio, which is calculated to determine the change that would be needed in the input parameters to reverse the ranking (EPA 1995b).

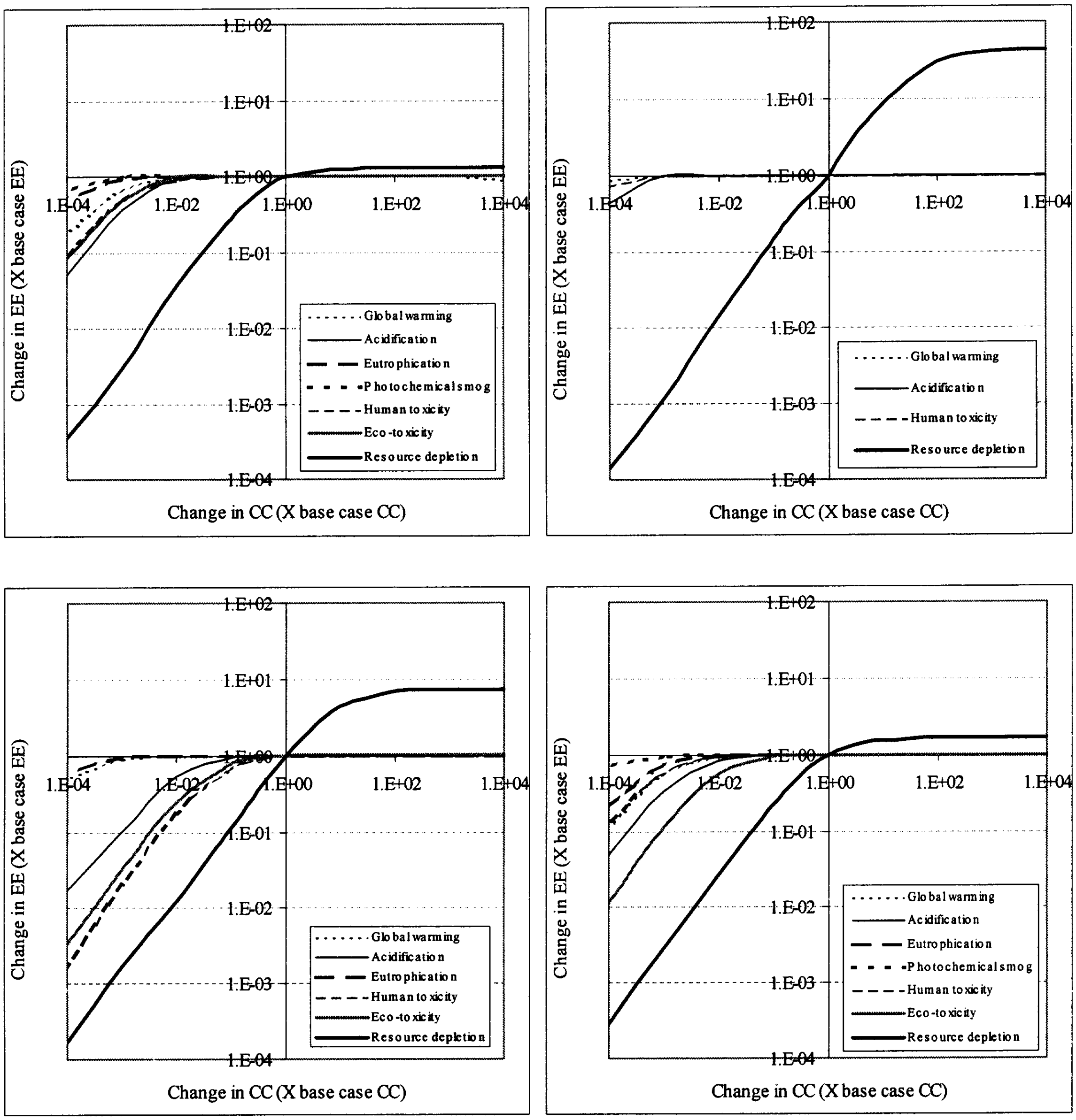

Figure 14.3 Effects of marginal change in carrying capacity on Eco-Efficiency for the productions of ABS plastic (top left panel), aluminum (top right panel), copper (bottom left panel), and steel (bottom right panel).

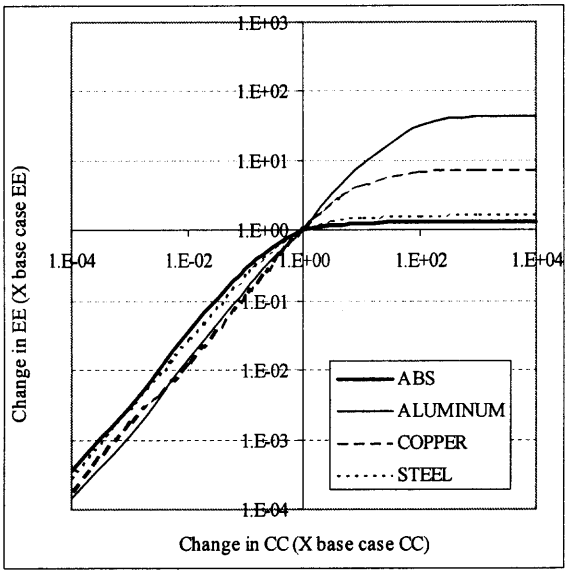

Figure 14.4 Effects of marginal change in carrying capacity of resource depletion on Eco-Efficiency. (Basic material production)

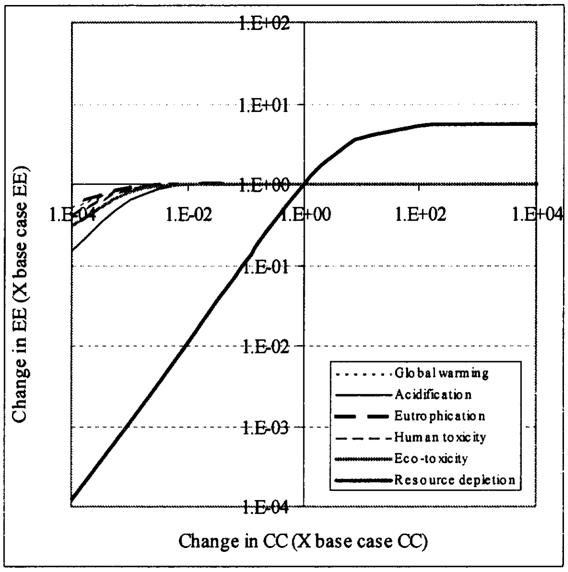

Figure 14.5 Effects of marginal change in carrying capacity on Eco-Efficiency for the production of a household plastic coffee maker.

A ratio sensitivity analysis is used for the case study of energy generation sources to investigate the marginal change in the carrying capacity estimates that would be needed to reverse the ranking between pairs of alternatives. This is accomplished by comparing the EE of two alternatives (ratio) that are changed due to a marginal change in the carrying capacity estimates for individual impact categories. The ratio between the EE of a better environmental performance alternative to the EE of a worse alternative must be greater than one (the higher the EE, the better the environmental performance). Reverse ranking is obtained when this ratio is less than one.

The results of the ratio sensitivity analysis for the case study of electrical energy generation sources are presented in Table 14.1. This table presents only the pairs of alternatives for which the ranking can be reversed by a marginal change in carrying capacity estimates. The ranking between the other pairs is not affected by a change in carrying capacity estimates within the designated range (0.0001 to 10,000 times or four orders of magnitude smaller to four orders of magnitude greater than the base case value). The sensitivity of the carrying capacity estimate can be seen from this table. For example, a marginal change in the acidification carrying capacity estimate as low as 21 times smaller than the base case value can alter the ranking between gas and lignite. As expected, a marginal change in the carrying capacity estimate (in the designated range) for ozone depletion impact does not reverse the rankings between any pair. This is because ozone-depleting substances are not released from the electrical energy generation sources (from the LCI). A change in the carrying capacity estimate for eco-toxicity impact also plays a small role. There are only a few toxic chemicals, with very low emissions, that are classified to the eco-toxicity impact. To reverse the ranking between gas and lignite, the decrease in eco-toxicity carrying capacity estimate for almost four orders of magnitude (8254 times lower) from the base case value is required. In this exercise, the analysis for resource depletion is not carried out because it is not appropriate to make a comparison of the basic processes that are based totally on different resource uses. In the comparison at the process level (as well as the supply line level), a marginal change in the major resource depletion carrying capacity of an alternative will not affect the STM scores of other alternatives that do not consume the same resource.

Table 14.1 Impacts of a Marginal Change in Carrying Capacity Estimates on the Ranking between Alternatives for the Case Study of Energy Generation Sources

Reduction factor that reverses the ranking (New CC = Base case CC/Reduction factor)	Pair of energy sources that the rank is reversed
Global warming impact	
553	Coal-Gas
1540	Coal-Oil
119	Gas-Lignite
452	Lignite-Oil
Stratospheric ozone depletion impact	No change
Acidification impact	
77	Coal-Gas
21	Gas-Lignite
Eutrophication impact	
2330	Coal-Gas
2783	Coal-Lignite
4938	Coal-Oil
1701	Gas-Lignite
5854	Lignite-Oil
Photochemical ozone formation impact	
3682	Coal-Lignite
Human toxicity	
1701	Coal-Gas
1874	Coal-Oil
1624	Gas-Oil
848	Lignite-Oil
Eco-toxicity	
8254	Gas-lignite
Resource depletion	No comparison

The ratio sensitivity analysis is also extended to the case study of basic material production (Table 14.2). Similar conclusions can be drawn that a marginal change in the carrying capacity estimate of the impacts with a very low emission (such as ozone depletion impact) plays an insignificant role in the analysis. On the other hand, a very small change in some carrying capacity estimates can alter the ranking between a pair with very similar EE scores. For example, a marginal change in acidification carrying capacity estimate as low as seven times smaller than the base case value can alter the ranking between ABS and steel. In this case, an error in the carrying capacity value less than one order of magnitude can yield the result in a different way.

Table 14.2 Impacts of a Marginal Change in Carrying Capacity Estimates on the Ranking between Alternatives for the Case Study of Basic Material Productions

Reduction factor that reverses the ranking (New CC = Base case CC/Reduction factor)	Pair of basic materials that the rank is reversed
Global warming impact 22	ABS/Steel
Stratospheric ozone depletion impact	No change
Acidification impact 7 622	ABS/Steel ABS/Copper
Eutrophication impact 22	ABS/Steel
Photochemical ozone formation impact 215	ABS/Steel
Human toxicity 115	ABS/Copper
Eco-toxicity 145 3329	Aluminum/Copper Aluminum/Steel
Resource depletion	No comparison

14.2.3 Summary and Conclusion

Two sets of sensitivity analysis are carried out, namely, one-way and ratio sensitivity analyses using the results of the case study of energy generation sources (Chapter 13). The results from the sensitivity analyses are expected to demonstrate the role of the carrying capacity estimates in the STM implementation.

From the one-way sensitivity analysis, an order of magnitude change in the carrying capacity estimates can vary the EE value by a magnitude up to ten times. The results also indicate that a marginal change in the limiting carrying capacity (the impact that dominates the overall EI) results in a change in the STM scores (EE in this case) that is more noticeable than a change in the carrying capacity for other impacts. The results from the ratio sensitivity analysis show that a change in carrying capacity estimates can alter the pair-wise comparison in terms of environmental performance between alternatives. This is obvious where a pair of alternatives have similar EE scores or where an impact is the primary contributor to the overall impact. However, results of a few case studies do not imply that the STM indicators of other implementations are sensitive to the carrying capacity estimates in the same manner. There are other factors such as the LCI data and the number of impact categories selected that can also affect the STM indicators.

The results from the sensitivity analyses agree with the fifth hypothesis established for this research. A conclusion that can be drawn from this finding is that the sophistication of the carrying capacity estimates also plays an important role in the STM implementation (as discussed in Chapter 13, another factor that is significant in the STM implementation is the number of impact categories taken into account). A rough, or less sophisticated, carrying capacity may lead to biased and misleading results, e.g., the superiority in terms of environmental performance among product alternatives. This proves that the accuracy and validity of the STM implementation relies heavily on the sophistication of the carrying capacity estimates that are used. A set of refined carrying capacity estimates is therefore needed in order to minimize any bias. The significance of the advance of the carrying capacity estimates can be established by the demonstration in the sensitivity analyses.

As pointed out, the soundness of carrying capacity estimates is essential in order to validate their use for a broad spectrum of applications. The following section introduces the causes and potential sources of uncertainty that are associated with the carrying capacity estimates in this research. This will be of benefit to the next step in the development of a more refined set of carrying capacity estimates.

14.3 Uncertainty Analysis

The reliability of the LCA is affected by several factors, one of which is the uncertainty due to data quality and methodological choices. An uncertainty analysis evaluates the contribution of the uncertainty of different parameters to the total uncertainty of results. There has been a lack of consensus about which methodology should be used to deal with uncertainty in LCA practice (Bjorklund 2002). Most of the uncertainty analyses for the LCI were focused on the qualitative methods because less quantitative tools have already been developed (Huijbregts et al. 2001c; Bjorklund 2002). For the LCA context, uncertainty analyses are targeted as an aid to improve the data quality in the LCA inventory. However, the uncertainty analysis in this research serves a slightly different role; its purposes are:

1. To increase confidence in the carrying capacity estimates;
2. To provide the information necessary for a justification of the reliability of the STM implementation in conjunction with the carrying capacity estimated in this research;
3. To better understand the importance of the data sources and models used in evaluating the carrying capacity estimates; and
4. To determine where data quality resources should be focused.

By using the information provided in this assessment, the level of uncertainty can be reduced by using the methods shown in the next step of the research should a more refined carrying capacity be required.

14.3.1 Qualitative Uncertainty Analysis

Firstly suggested by Huijbregts (1998), the uncertainty and variability in LCA can be distinguished into several types: parameter uncertainty, model uncertainty, uncertainty due to choices, spatial variability, temporal variability, and variability between objects/sources. A more complete list and definition of types and sources of uncertainty in conventional LCA are discussed by Bjorklund (2002) as illustrated in Table 14.3.

According to EPA (1995b), any data error may be addressed by both quantitative and qualitative techniques. The quantitative analysis addresses how the uncertainties associated with the input parameters affect the uncertainty of the overall results of the model. Basically, statistical and mathematical methods are popular for performing the quantitative uncertainty analysis. Other methods can also be applied including that of expert judgments (Huijbregtes 1998) and the "rules of thumb" (Finnveden and Lindfors 1998). A quantitative uncertainty analysis is appropriate when the numerical uncertainty ranges of the input variables are available. Otherwise, when less numerical data is available, a qualitative uncertainty analysis can be used instead (Bjorklund 2002).

A qualitative uncertainty analysis is applied to this research because of the lack of uncertainty information associated with data used in calculating the carrying capacity estimates. Most of the data used here are secondary data that were derived from other research, studies, literature, databases, and regulatory and standard establishing processes. These secondary data are mostly selected as discrete or single point data without associated uncertainties being provided in the terms of probability density function. Therefore, there is no known quantitative uncertainty range and errors given as a probability distribution. It should be anticipated here that the point estimates without uncertainty may be unreasonably overestimated.

Table 14.3 Types and Sources of Uncertainty (Source: Bjorklund 2002)

Data inaccuracy: Data inaccuracy concerns the empirical accuracy of measurements that are used to derive the numerical parameter values. Measurements can be subject to random error, which results from imperfections in the measuring instrument and observational techniques, or systematic error, which results from an inherent flaw or bias in the data collection or measurement process.

Data gaps: Missing parameter values may leave the model with data gaps.

Unrepresentative data: Data gaps may be avoided by using unrepresentative data, typically data from similar processes, but of unrepresentative age, geographical origin, or technical performance.

Model uncertainty: Model uncertainty is due to simplifications of aspects that cannot be modeled within the LCA structure, such as temporal and spatial characteristics lost by aggregation, linear instead of non-linear models, or derivation of characterization factors.

Uncertainty due to choices: Choices are unavoidable in LCA. Because there is often not one single correct choice, there is uncertainty in choice, for instance, of allocation rules, functional unit, system boundaries, characterization method, weighting method, marginal or average data, or technology level.

Spatial variability: Variability stems from inherent fluctuations in the real world. Although there are natural variations between different geographical sites, environmental interventions are usually summed up in the impact assessment, regardless of the spatial context. Examples of these factors that vary over space are background concentration and human population density.

Temporal variability: Variations over time are relevant in both the inventory and impact assessment, as processes and factors in the receiving environment vary naturally over short and long time scales. Examples are process emissions, wind speed, and temperature. Another aspect is the chosen time horizon to integrate potential effects, which, for instance, applies to global warming potentials (GWP), photochemical ozone creation potentials (POCP), and emissions from landfills.

Variability between sources and objects: Variability also appears between sources of the inventoried system (e.g., inherent variations in comparable technical processes), objects that determine the impact on the environment (e.g., human characteristics such as body weight or sensitivity to toxic substances), and preferences that determine the weighting of impacts.

Epistemological uncertainty: Epistemological uncertainty is caused by lack of knowledge on system behavior. It affects all phases of LCA. By nature, it is seldom acknowledged, and is very difficult to assess. A certain type of epistemological uncertainty arises when future systems are modeled, because the future is inherently uncertain.

Mistakes: Sheer mistakes are also a source of uncertainty. As in the case with epistemological uncertainty, mistakes are seldom acknowledged and are very difficult to assess.

Estimation of uncertainty: Estimation of all types of uncertainty is in itself a source of uncertainty.

However, it is beyond the scope of this research to investigate the quantitative uncertainty analyses of the primary data; the uncertainty associated with the development of the secondary data may be available within the sources of the data. Furthermore, it is difficult and not feasible to perform the combined uncertainty analysis for every impact category, especially for toxicity impacts, which may expand to an indefinite number of impact subcategories. The combined uncertainty analysis includes the inherent uncertainties within the development of the secondary data from the primary data and uncertainties associated with applying the secondary data to the carrying capacity estimates. Figure 14.6 illustrates an example of the combined uncertainty analysis of applying a chemical (human toxicity impact) as a baseline reference to determine the overall STM scores.

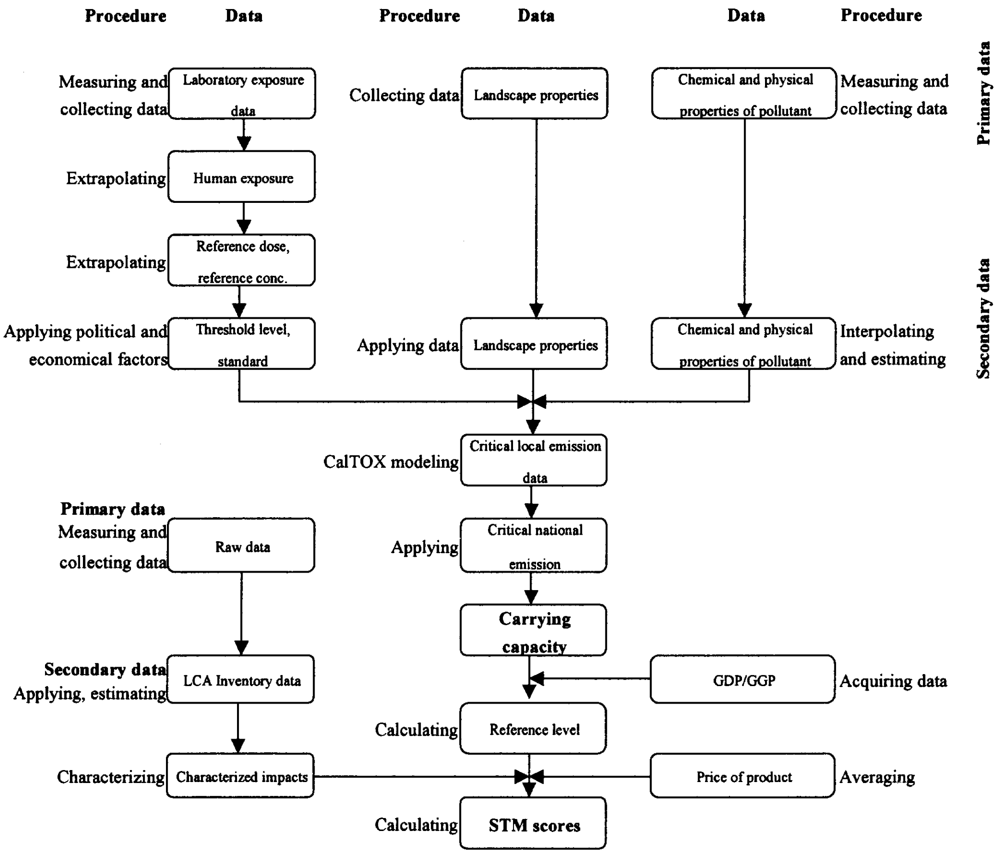

Figure 14.6 Diagram of uncertainty associated with procedures of data evaluation.

The uncertainty analysis presented in this chapter is associated with the development of the carrying capacity as well as the application of the STM in a typical environmental performance evaluation. The uncertainty analysis is expressed as the quality index matrix (high to low uncertainty) for sources and causes of uncertainty within components of the carrying capacity estimates and types of uncertainty as categorized as in Table 14.3. An example of the uncertainty analysis in terms of quality index matrix for the human and eco-toxicity impacts is presented in Table 14.4 (low uncertainty is favorable). The matrices for other impact categories are presented in Table F.1 to Table F.7 in Appendix F.

From Table 14.3, the first three components of the carrying capacity evaluation for human toxicity impact (toxicity data, landscape properties, and chemical properties) involves with the development of the secondary data from the primary or raw data. The secondary data from these components are used in the carrying capacity evaluation. As stated, the uncertainty associated with the development of the secondary data may be available within the sources of the primary data. It is beyond the scope of this research to investigate the uncertainty analyses associated with the data development. Therefore, the uncertainties associated with these components are indicated in the table as the ones that are inherent within the sources of data.

The uncertainties containing in the next component, the carrying capacity estimate, are caused by the steps taken in this research. There are various types of uncertainty involved with the carrying capacity estimate. For example, the model uncertainty is considered high because the CalTOX model, which is used in the carrying capacity estimate, is a screening model and there are some assumptions and rough estimates in the CalTOX calculations.

Table 14.4 Uncertainties Associated with the Carrying Capacity Evaluation for Human Toxicity Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
Toxicity data (RfC, RfD, standards, target concentrations)	Collecting and measuring data	Data inaccuracy	Inherent within sources	Medium
		Data gaps		
		Unrepresentative data		
	Extrapolating	Model uncertainty	Inherent within sources	Low
		Uncertainty due to choices		
		Variability between sources and objects		
Landscape properties	Collecting and measuring data	Data inaccuracy	Inherent within sources	Low
		Data gaps		
		Unrepresentative data		
		Spatial variability		
		Temporal variability		
		Variability between sources and objects		
Chemical properties	Collecting and measuring data	Data inaccuracy	Inherent within sources	Low
		Data gaps		
		Unrepresentative data		
	Interpolating and estimating	Model uncertainty	Inherent within sources	Low
		Temporal variability		
		Uncertainty due to choices		
		Variability between sources and objects		
		Epistemological uncertainty		
Carrying capacity estimate	CC estimate using CalTOX	Model uncertainty	High	Low
		Uncertainty due to choices	High	
		Spatial variability	High	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
	CC extrapolating for other chemicals	Unrepresentative data	High	Low
		Uncertainty due to choices	Medium	
		Temporal variability	High	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
STM implementation	Data acquiring	Uncertainty due to choices	High	Low
		Data inaccuracy	High	Low

The uncertainty due to choices is considered high because there may be other models or methodologies for carrying capacity estimate that are better or more appropriate than the one used in this research. The spatial uncertainty is considered high because the landscape data used in CalTOX simulation is locally-specific but the average ones of the U.S. in this research. The uncertainties in other procedures and components are also considered in a similar manner on a case-by-case basis. For STM implementation, LCI data are often collected systematically.

The reliability of data sources is based on the sources themselves. High reliability is given for a source that is considered as a general consensus, e.g., international agreements or regulatory information. Low reliability is given for a source that is less rigorous or does not have a strong supporting background.

It should be noted that uncertainty is inherent in every step of the development and data acquiring processes. This assessment provides only the key uncertainties that should be focused on. The degree of uncertainty in the matrices is determined following the criteria suggested by Weidema (1998), Rousseaux et al. (2001), and Huijbregts et al. (2001e). The qualitative uncertainty analysis presented in this chapter will facilitate further research in reducing the errors and uncertainty by (after EPA 1995b):

- Careful planning and executing of the carrying capacity development processes;
- Review of the data development processes;
- Focusing data quality efforts on data values that have the greatest influence on the carrying capacity estimates; and
- Obtaining as large an amount of data as possible to increase the reliability of the estimates and to decrease its variability.

14.3.2 Summary and Conclusion

This section has provided an uncertainty analysis for the carrying capacity estimates. The uncertainty analysis is expressed in terms of a qualitative uncertainty analysis. A set of data quality index matrices is presented to show the possible causes and potential sources of uncertainty associated with the steps in calculating the carrying capacity estimates. In the matrices, the degree of uncertainty is a result of the steps taken in order to calculate the carrying capacity estimates. These matrices also identify the steps that should be focused on in order to develop a more refined set of carrying capacity estimates for further studies.

The uncertainty analysis presented in this chapter serves as a means to: 1) increase confidence in the carrying capacity estimates, 2) provide information to bolster the reliability of the STM implementation in conjunction with the carrying capacity estimates, 3) understand the importance of data sources and models used in the carrying capacity estimates, and 4) indicate the gaps in where data quality resources to take into account to reduce errors and uncertainties in future studies.

CHAPTER 15

SUMMARY, DISCUSSION, AND CONCLUSIONS

15.1 Summary and Discussion

15.1.1 Carrying Capacity Estimates

1. Global Warming Impact

A time-dependent carrying capacity for GHG emission is estimated from the mitigation scenario that stabilizes the atmospheric carbon dioxide concentration at 380 ppmv. The emission levels in this mitigation scenario may not be realistic. However, it illustrates a possible pathway of achieving the most conservative option that considers the natural absorption capacity to minimize the future climate change. At this emission projection, the temperature will be stabilized at +0.9 °C (2060) relative to the 1990 level and the atmospheric CO_2 concentrations will reach the stabilization level within a few years from now. The carrying capacity is estimated annually up to the year 2100.

Because of the importance of the global warming risk potential, a large number of investigations are expected to be conducted in the near future. These new findings will obviously influence the view of the IPCC and other scientists. For example, the new findings may lead to a more stringent upcoming greenhouse gas limitation and mitigation policy, which is currently debatable due to the scientific uncertainties. More refined climate models may also lead to the revision of greenhouse gas emission and mitigation scenarios. The damage assessment may be refined and a consensus on threshold level may be identified. As a result of these new findings, the carrying capacity may need to be refined accordingly. Future IPCC assessment reports are a source of information that will be useful for the refinement of the carrying capacity.

One factor that may be considered for this carrying capacity estimate is any future international protocol on GHG limitation. The emission level established by any such international agreement may not account for the natural absorption capacity, but it would reflect the political, social, and practical basis that is acceptable to the LCA community.

2. Stratospheric Ozone Depletion Impact

The carrying capacity is estimated by assuming that ozone depletion is solely an effect of ODS emissions. If the ODS emissions are limited to the pre-1980 level, the ozone layer will be allowed to recover to the pre-1980 level as well. The Montreal Protocol and its amendments and adjustments stipulates that ODSs be scheduled for production cuts, emission cuts, freezes, and eventual phasing out. Analogous to the carrying capacity for global warming impact, the time-dependent carrying capacity for stratospheric ozone depletion impact is adapted from the amounts projected by the Montreal Protocol. In this research, the carrying capacity is estimated for up to the year 2100.

The projected emission of ODS may not be the theoretical "sustainable level" because it takes into account several practical factors. The carrying capacity for the emission of ODS should be revised accordingly in line with other future findings. Recently, UNEP and WMO have been cooperating in addressing the ozone depletion issue. As a result, scientific assessment reports have been published and updated periodically (by NOAA, NASA, UNEP, WMO, and EC). Future research supported by these organizations and agencies as well as the political issues related to ozone emission limitation are sources that will be useful for the refinement of the carrying capacity.

3. Acidification Impact

The carrying capacity for acidification impact is estimated from the U.S. target levels according to Title IV of the Clean Air Act Amendments of 1990. Title IV requires the reduction of 10 million short tons per year of SO_2 and 2 million short tons per year of NO_x by 2010 compared to 1980 levels. Target levels for national SO_2 and NO_x emissions are estimated at 16 and 25 million short tons per year in 2010, respectively. It is expected that the emission projection according to Title IV of CAAA of 1990 would result in the mitigation of impacts to various degrees for the eastern U.S., where acidification is a serious environmental issue. For instance, sulfur deposition in some sensitive areas would be reduced by 30-50% in 2010 compared to the 1985 levels, the life span of sensitive cultural materials would be expanded by 30-40%, and the number of lakes unsuitable for aquatic biota would be reduced by approximately 10%.

Acidification is considered as a regional impact. Region-specific carrying capacities are estimated by applying region-specific factors. The region-specific factors are weighted according to the seriousness of the acidification problem among regions using the current field data of acid wet deposition as the indicator. By applying the region-specific factors, the regions susceptible to acidification have lower carrying capacities than the more tolerant ones.

Site-dependent characterization factors (site factors) are also needed in order to characterize an emission of acidifying substances from a specific location in the U.S. Site factors are basically the fate factors that take into account the transport of a pollutant under the site-specific atmospheric pathways and chemistry processes. A higher site factor indicates more deposition of pollutant and more contribution to acidification impact on North America. By applying the site factors and chemical characterization factors, the overall characterization of acidifying substances can be calculated.

The carrying capacity for acidification impact can be refined to reflect changes in policy on acid deposition control and new scientific findings. Another means to refine the carrying capacity is the use of a long-range transport model coupled with a soil-water chemistry model. This approach is likely to be more accurate and sophisticated than simply using the target levels; but the use of this sophisticated approach is less feasible because it is very complex, costly, and time-consuming. However, it may be employed in the future should a more refined carrying capacity be needed.

4. Eutrophication Impact

The carrying capacity for eutrophication impact is estimated using the critical load concept. The critical loads for eutrophication are adopted from TMDL reports from 27 states available online. Generally, TMDLs are reported for impaired waters only. The carrying capacity estimate in this research is derived from the available TMDL reports assuming that the impaired waters are representative for all other waters. Since eutrophication impact is considered to affect an area as large as regional and continental scales, only the national level carrying capacity for phosphorus and nitrogen nutrients are determined. The U.S. carrying capacity for eutrophication impact is determined for both phosphorus and nitrogen nutrients.

From the LCA perspective, several burdens from nutrient emissions may be aggregated due to the assumption that P and N compounds have an additive effect to the eutrophication impact. In this research, however, cross-species aggregation between P and N is not recommended. This is due to the spatial and temporal variations of the N:P ratio in various water bodies. Furthermore, the synergistic and additive effects due to the

P and N loadings seems to be less logical, especially for the STM, where the impact assessment is based on a damage-oriented approach rather than an emission-oriented approach used in other LCIA methods. As a result, the carrying capacity estimates for P and N are not aggregated and they are counted as impact subcategories.

The carrying capacity for eutrophication impact can be refined when better methodologies and new scientific findings become available. Further improvement includes the evaluation of locally-specific carrying capacities. A locally-specific carrying capacity can be determined from the source-receptor or dose-response relationship which is the result of water pollution modeling. Another means to refine the carrying capacity is to update TMDL information. Although TMDL development has been progressing since 1972, not many TMDLs have been assessed until the past few years. Recently, legal actions have forced EPA and states to assess more TMDLs. A large number of TMDLs is thus expected to be developed in the next 10 to15 years.

5. Photochemical Ozone Formation Impact

The carrying capacities for VOC and NO_x emissions are estimated for the U.S. following the threshold-oriented method developed in this research. The carrying capacity is calculated from the average critical VOC and NO_x emission rates of ten U.S. metro cities. The evaluation of carrying capacity for photochemical ozone formation impact employs the OZIPR model to simulate the critical VOC and NO_x emission rates, i.e., the emissions that cause the atmospheric ozone concentration just reaching the threshold level. The NAAQS for ozone (old 1-hr 0.12 ppm and new 8-hr 0.08 ppm) is selected as the threshold level for photochemical ozone formation.

The carrying capacity estimated in this research is conservatively biased because it is assumed that NO_x and VOC concentrations are in excess of the level required to cause a critical (maximum) ozone concentration for both VOC-sensitive and NO_x-sensitive regimes, respectively. Practically, in some areas with high emission of VOC, NO_x may not be available at a level high enough to form a photochemical reaction with all of the VOC and vice versa. If a precursor is only available in small amounts, this will level up the critical emission of the other precursor so that the maximum ozone formation is reached (causing a higher carrying capacity, which will result in a less conservative estimate). Furthermore, critical emission of a precursor cannot be reached in the case of a very low concentration of the other precursor; therefore it is assumed that there is always the other precursor in excess for the greatest ozone formation. Another assumption that causes the carrying capacity to be conservatively biased is that the ozone formation is carried over to night-time in the same manner as in daytime. Without this assumption, the carrying capacity cannot be reached because the ozone formation will not occur in the absence of sunlight, which will imply unlimited VOC or NO_x emission at night-time for the photochemical formation impact.

The conversion of a VOC to its ethylene equivalent can be carried out using either MIR or POCP values. However, the MIR is recommended by this research because: it was developed using U.S. atmospheric conditions, its scenarios represent NO_x conditions where organic emissions have the greatest effect on ozone formation, and its values are available for approximately 700 VOCs. The conversion factor of a “typical VOC” to C_2H_4-equivalent is estimated to be 0.481 kg VOC/kg C_2H_4 using MIR values. This conversion factor is higher than that used in EI95.

The proposed carrying capacity estimate using OZIPR should be used in the U.S. only since the meteorological and emission conditions vary spatially. To estimate the carrying capacity for other parts of the world, corresponding data may be needed as the

OZIPR input. Otherwise, other appropriate photochemical models may be used. A more sophisticated model will certainly improve the accuracy of the carrying capacity estimate.

The carrying capacity estimate can be revised and modified when the input parameters for photochemical modeling are refined as well as the change in reference threshold ozone concentration when the new regulatory level is promulgated or a new threshold level is selected. The carrying capacity estimate may also be refined should the input data for the OZIPR are prepared for other cities. However, the revision of the carrying capacity following the method presented here may be difficult and time consuming because there are a number of documents and information to be reviewed. In this case, a less difficult and simpler approach may be used.

6. Human Toxicity Impact

The carrying capacity for human toxicity impact is estimated for 89 chemicals using the threshold-oriented technique in conjunction with CalTOX. In this approach, the chemical-specific emission-concentration relationship (the partitioning factor) is estimated using CalTOX, an integrated multimedia environmental model. By using CalTOX, concentrations of a chemical in target media (air, surface water, and surface soil) resulting from a unit emission of the pollutant are obtained. As the linear function of emission-concentration relationship is assumed, the carrying capacity can be determined from the emission of the pollutant that results in the concentration in a target medium not exceeding the desirable threshold level. The threshold levels for air, surface water, and surface soil are taken into account. The target threshold levels are adopted mainly from regulatory standards (e.g., NAAQS, MCL) and the toxicity database for risk assessment (e.g., IRIS and MRL). The carrying capacity is estimated for those three types of environmental media resulting from both air and water emissions. The uncertainty associated with the carrying capacity estimate in the threshold-oriented technique using CalTOX is the result of several factors including the selected threshold levels, specific physical and chemical properties of the pollutants, and the assumptions made from the landscape information in CalTOX.

There is a difficulty in providing the carrying capacity for all chemicals or pollutants, not only because the data are not available, but also because it is costly to do so. Therefore, an empirical approach may be used to estimate the carrying capacity for chemicals not assessed using the threshold-oriented technique in conjunction with CalTOX. The empirical approach can be used for the carrying capacity estimate based on the fact that the carrying capacity is relatively proportional to the target threshold levels. The approach also assumes average partitioning factors derived from the data set of 78 organic chemicals. By using the empirical approach, the carrying capacity can be estimated directly from a target threshold level.

A further improvement of the carrying capacity for human toxicity impact is certainly possible. For example, the carrying capacity for a chemical can be added or revised when the target threshold level is updated. A more recent target threshold may be due to changes in the regulatory levels and standards and updates of the toxicity databases. The revision of the carrying capacity can be accomplished by using the CalTOX simulation or by using the empirical approach. Further improvement of the accuracy of carrying capacity estimates may also be achieved by the use of a more sophisticated and accurate environmental fate and transport model. An estimate of the carrying capacity that takes into account the spatial variation can be carried out using locally-specific modeling.

7. Eco-Toxicity Impact

The carrying capacity for eco-toxicity impact is estimated using the threshold-oriented technique developed in this research. The empirical carrying capacity estimate for eco-toxicity impact is based on two components: the assessment of environmental transport of toxic chemicals and the assessment of threshold levels. The assessment of environmental transport of toxic chemicals is adopted from the one that has been carried out in the evaluation of human toxicity impact. The partitions of toxic chemicals in the environmental media as a result of a unit of emission are expressed in terms of partitioning factors, which were calculated using CalTOX, an integrated multimedia environmental model.

For the assessment of the threshold levels, it is difficult to select an appropriate set of threshold levels. This is because there are multiple effects on ecosystems caused by individual substances and there is insufficient supporting information to select a single entity to represent the functioning of ecosystem as a whole. In this research, the predicted-no-effect-concentrations (PNECs) are used as a surrogate for the generic assessment endpoint to protect the ecosystem as a whole. The PNEC is defined as the highest environmental concentration expected to cause no effects, acute or chronic, on the structure or functioning of ecosystems. PNECs are therefore utilized as the target threshold levels.

The U.S. carrying capacity for eco-toxicity impact have been estimated using the empirical equations derived for human toxicity impact. However, the carrying capacity is available for only 131 chemicals for aquatic eco-toxicity impact due to the limited number of reports on PNECs. In the future, the carrying capacity for more chemicals can be determined in the same manner based upon the increased availability of PNECs or other endpoint indicators for ecological health. PNECs and other indicators, which are feasible as the threshold levels, may be evaluated from several toxicology databases.

The next step in estimating eco-toxicity carrying capacity may also be the development of a consensus methodology or approach that can evaluate generic assessment endpoints/indicators at the ecosystem-level. Additional aspects include the specification of areas of protection in ecosystems. A refinement also could include the use of a more sophisticated and accurate environmental fate and transport model. Furthermore, a carrying capacity estimate that takes into account local conditions could be carried out using locally specific modeling.

8. Resource Depletion Impact

The carrying capacity for fossil fuels, minerals, and freshwater consumption is estimated for the resource depletion impact. For minerals, the carrying capacity is related to virgin minerals only. The carrying capacity estimates for resource depletion are in the category of economic carrying capacity because they are associated with value-added activities.

For non-renewable resources, such as fossil fuels and minerals, the carrying capacity can be determined using the reserve-time horizon technique developed in this research. This technique calculates the maximum allowable yearly consumption of existing resources for a specified time horizon. The carrying capacity decreases as reserves are depleted each year to assure a continuous supply for the specified time horizon. For the carrying capacity estimate in this research, the time horizon of 200 years is used for consumption of fossil fuels and minerals. This time horizon is selected on the basis that existing world coal reserves can last for approximately 200 years at today's consumption rate. Coal is the most used and the most abundant fossil fuel in the world's current mix of power sources. Should supporting literature on the time horizon for resources become available, a more appropriate time scale may be examined and applied. The time horizon use in this technique can be altered to a more appropriate figure depending on the purpose of an LCA study and the judgment of the practitioner. An advantage of applying the reserve-time horizon technique to estimate the carrying capacity is that the existing amount of reserves is flexible. The reserves are depleted each year by human consumption but, on the other hand, they can be expanded due to mining exploration and improved technology.

The carrying capacity for the consumption of renewable resources such as freshwater is simply determined from their replenishment rate. The renewable rate for freshwater, which is calculated as precipitation left after losses and natural uses, is used as the carrying capacity for both groundwater and surface water. The carrying capacity for water consumption is evaluated in terms of water availability only. The carrying capacity of pollutant discharge to water resources is addressed in eutrophication impact, human toxicity, and eco-toxicity impacts. For the U.S., the locally specific carrying capacity estimates are provided.

The carrying capacity estimate in this research can be refined by updating the information on remaining reserves of non-renewable resources. The carrying capacity estimate for resource depletion can also be further advanced by inclusion of other resources such as biotic resources, biodiversity, and land use.

15.1.2 Limitations of the Carrying Capacity Estimates

1. Not all relevant impacts are evaluated

A critical point of the carrying capacity development was the selection of the impact categories. The impact categories selected in this research are normally considered in LCA as recommended by the ISO. To make the environmental performance evaluation using the STM more universal, all relevant environmental aspects and potential impacts should be included. However, that will need a very large amount of effort. Use of the STM in conjunction with the carrying capacity as carried out in this research must always state the impact categories used as one of the criteria in the evaluation. All users should be aware of this basis. A more universal estimate, with more relevant environmental aspects, may alter the result of the evaluation using the currently available impact categories and their associated carrying capacity estimates.

An example of this limitation is the absence of nuclear waste management and land use impact. Nuclear waste management and land use are ones of the primary environmental concerns in LCIA as well as other environmental assessments. The carrying capacity for nuclear waste management has not been completed because it is difficult to determine the management capacity that can be used as the surrogate for the carrying capacity. Land use is not taken into account in the STM at this stage because a typical LCA inventory does not typically include land use as an environmental aspect and it is difficult to determine the land use available for different types of environmental management.

Another example is the absence of waste management. Waste management is a major concern in the U.S. especially in its environmental performance at a corporate level. However, this aspect may be addressed by tracking the waste flow and translating the waste streams into the impacts of the available impact categories.

2. A limited number of toxic chemicals in human and eco-toxicity impacts

The number of toxic chemicals considered in human toxicity impact is limited to the top one hundred of those chemicals that are the most commonly released into the environment. Sometimes the LCI provides data for chemicals other than those addressed in this research. In this case, the carrying capacity of the chemicals can be determined from other available references for threshold levels using the empirical approach developed in this research.

Likewise, the carrying capacity for eco-toxicity is available for only 132 chemicals, which are derived from the available PNECs. In the future, the carrying capacity for more chemicals can be determined in the same manner. The target threshold levels or PNECs may be evaluated from several toxicology databases.

3. The environmental performance evaluation is input-oriented

It is observed from Chapter 13 that resource depletion is obviously the major contributing impact for the environmental performance evaluation of all case studies using the STM. In some cases, although eco-efficiency of resource depletion is the only impact that is below the threshold sustainability level, the overall eco-efficiency of the evaluation is also well below the sustainability level. This input-oriented evaluation may be due to the conservative carrying capacity of resource impletion impact or the intrinsic characteristics of the STM calculations which may exaggerate the major impact. Therefore a study for a new approach to determine a less conservative carrying capacity for resource depletion impact may need to be carried out. The calculations in the STM may need to be modified to minimize any overemphasis. For an environmental performance evaluation or an LCA that is output-oriented, one may choose to consider only the output-related impact categories at this time. A characteristic of the STM is that it allows a limited number of impact categories to be added. The number of impact categories considered depends on the goal and scope of the evaluation or assessment. For an evaluation or assessment that is input-oriented, one may use the carrying capacity developed in this research or select a more appropriate value that suits the goal and scope.

4. Improvement Opportunities

Approaches used, models used, and data quality are the major aspects that could be improved in order to refine the carrying capacity estimates. This research found troublesome the attempt to use the same approach to evaluate the carrying capacity for output-related impact categories. This is because the approach used relies on the robustness and the availability of environmental models and their associated input data. In some impact categories, it is difficult to decide what would be the most appropriate models. Some models are robust but they surely need more resources while some models are less sophisticated but they may deliver results in which less confidence can be placed. Another difficulty is that data is not always available for the selected models. To cope with these difficulties, a new approach may be developed for the evaluation of carrying capacity. A new approach or methodology may or may not need the environmental models and their associated data.

In the selection of environmental models used in this research, factors taken into account are the availability of input data and the resources needed in order to implement them. More sophisticated models will surely gain confidence in the carrying capacity estimates. Quality of data is also an important issue. The data applied in the carrying capacity development are the most up-to-date, yet some uncertainty due to the quality of data and the models is present.

Another aspect that may need improvement is the background assumptions used in the carrying capacity estimates. These assumptions, e.g., calculation assumptions and scientific assumptions, have been discussed in the chapters involving carrying capacity estimates. Most assumptions are made based on up-to-date scientific bases. In the future, more appropriate assumptions may be applied if more supporting information is available. The carrying capacity can then be recalculated. In fact, this should be seen as an evolving process.

15.1.3 Characteristics of the STM

1. The STM combines the economic and environmental considerations

The STM takes into account economic considerations by application of the economic value of a product or service. The application of the economic value may also be extended to social considerations, which may be quantified in terms of monetary value of social perception of the willingness-to-pay to protect or prevent environmental damages and social value of the product or service.

2. The STM deals with temporal and spatial variations

The STM attempts to take into account the temporal and spatial variations as well. Mostly, the aspects and life cycle stages of a system are temporal and spatial variations. The impacts demonstrate spatial variation when local or regional scales are considered and there are temporal variations when the aspects are not influenced simultaneously. Temporal and spatial variations could be dealt with in the STM calculations where EI is the aggregation of EI from several aspects or life cycle stages of a system.

As demonstrated in the case study of the production of household coffee maker, the overall EI is the sum of EI from different life cycle stages of the product, which are temporal and spatial distributions.

3. The STM measures the environmental impacts related to sustainability

In the STM, the significance of the environmental impact is not only identified, it is also expressed as a group of indicators related to the sustainability target. In other words, the sustainability target is used as the ruler to measure environmental performance of systems. This makes the STM a tool for absolute measure, with which the environmental performance or the level of environmental stewardship of a product can be identified without any further comparison with others. The evaluation of environmental impacts as related to the sustainability target may also be considered as an approach toward the precautionary principle in the examination of alternatives in the decision-making process.

4. The STM assesses environmental impacts without weighting procedure

Even though other sophisticated LCIA methods are capable for a full stage implementation, the subjectivity is still presence in the weighting factors where they are derived from the value systems such as expert panels and governmental policies. These value-based weighing factors are controversial and may decrease the credibility of the entire LCA process. The weighting procedure is used traditionally in LCIA to identify the relative importance among various impact categories. With the STM, weighting is avoided because the environmental impacts are expressed based on the earth's carrying capacity. Exceeding the earth's carrying capacity is unsustainable regardless of the specific impact category. The STM is considered as an endpoint assessment, i.e., the environmental damage is assessed relative to the sustainability target.

5. The STM is flexible for the selection of impact categories

An initial step in LCA is the goal and scope definition. At this step, the number of impact categories is selected An LCA should include all relevant environmental aspects and impacts. The impact category selection may also be made based on the availability of information in addition to the goal and scope definition. Most of the LCIA methods have a pre-defined and limited number of impact categories. A method that is flexible in the selection of impact category is more preferable in accordance with LCA context.

The impact assessment of the STM is simple and straightforward. The impact categories are not attached to the calculation processes; therefore, the STM is flexible and not limited to the selection of impact categories. The characterized impact potentials from other methods can be further assessed in the STM.

6. The STM provides environmental condition assessment

An environmental condition indicator (ECI) provides the information on environmental health and conditions at different scales. This type of indicator links specific business activities or emissions to environmental impacts, i.e., establishes a relationship between pollutants and impacts. According to the National Academy of Engineering (NAE 1999), this type of indicator is of greatest interest to industry and stakeholders because it may be used to estimate environmental performance toward the sustainability of human activities. Indicators in the STM may be considered as an ECI because they are derived from environmental conditions. In other words, the sustainability is the ultimate target for improvement or maintenance of environmental conditions.

15.2 Conclusions

15.2.1 Evaluation of Environmental Carrying Capacity

The first hypothesis is examined by the evaluation of the U.S. carrying capacity estimates and it is also the primary task of this research. Impact categories taken into account are: global warming impact, stratospheric ozone depletion impact, acidification impact, eutrophication impact, photochemical ozone formation impact, human toxicity impact, eco-toxicity impact, and resource depletion. These impact categories are compatible with the ones recommended by ISO 14040 series. The appropriate scales are applied: global scale for global warming and stratospheric ozone depletion impacts and regional scale (U.S.) for other impacts. The regional and local sensitivity of receiving environments is considered in the carrying capacity for acidification and water consumption impacts. A method that takes into account the spatial distributions of emissions is provided in the characterization procedure for acidification, eutrophication, and photochemical ozone formation impacts. Once determined, the carrying capacity estimates are compared to those in the literature. The second hypothesis was assessed at the summary of the chapters relating to the evaluation of carrying capacity.

Generally, the carrying capacity estimates for output-related impact categories are based on the threshold-oriented technique using threshold concentrations in environmental compartments. Most of the threshold concentrations are adopted from the consensus target levels such as national standards, national databases, and international agreements. By using these levels, more confidence and reliability is given to the STM by reason of the fact that these levels have been derived based upon a strong scientific basis. In addition, these consensus levels are also accepted as being safe. A carrying capacity is basically determined from the emission that causes the predicted environmental concentrations not exceeding the consensus target levels. This can be accomplished using appropriate environmental fate and transport models.

The methodologies for carrying capacity evaluation developed in this research has been examined further by evaluating the feasibility to determine the carrying capacity in other regions. A set of European-based carrying capacity estimates was evaluated. It can be concluded from the results that it is feasible to establish a set of carrying capacity estimates for Europe as well as for other regions providing that some adjustments are made. For Europe, the carrying capacity for global warming, stratospheric ozone depletion, acidification, eutrophication, and photochemical ozone formation impacts can be compared with two similar concepts of the carrying capacity derived from information in other existing studies. The carrying capacity for global warming, stratospheric ozone depletion, acidification, and eutrophication impacts show an agreement with the "target values" derived from the "normalization values" (existing emissions) coupled with the "reduction factors" evaluated in Eco-Indicator 95 (Goedkoop 1998) and the "European "environment's carrying capacity " evaluated in EDIP (Hauschild and Wenzel 1998).

15.2.2 Performance of the STM in Conjunction with Carrying Capacity

The third and the fifth hypotheses of this research are examined here while the fourth hypothesis has already been assessed in Section 15.1. For the third hypothesis, the case studies in this research were conducted to examine the applicability of the STM in conjunction with the carrying capacity LCA to identify superiority among alternatives in the same fashion as do other LCIA methods and to identify the trend of impact contribution. The case studies demonstrate that the STM in conjunction with the carrying capacity developed in this research is readily available for a range of LCA practice.

The results from the case study of electrical energy generation sources suggest that energy generation using coal is the most favorable environmental performance according to the assessment using the STM. The following favorable sources are lignite, gas, and oil respectively. This trend is different from the results obtained from the assessments using other LCIA methods. The results from the case study of basic material productions indicate that steel is the most favorable environmental performer. The following materials in order of favorability are ABS, copper, and aluminum respectively. This trend is also different from the results obtained from the assessments using other LCIA methods. For both case studies, resource depletion is the major impact for most of alternatives assessed by the STM.

Since the ranking of alternatives assessed by the STM for both case studies is different from other LCIA methods, an investigation was further conducted to verify the discrepancy by limiting the impact categories for the STM as well as other LCIA methods to the common ones. The results indicate that there is an agreement in the trends when using only the common impact categories (global warming, ozone depletion, acidification, eutrophication, and photochemical ozone formation). Therefore, it can be concluded that the number of impact categories taken into account may be significant to the outcome of an environmental performance evaluation. The difference in weightings of the impact categories used in LCIA methods may also play an important role in the evaluation as well.

The case study of the household plastic coffee maker has demonstrated that the STM coupled with the appropriate set of carrying capacities may have the capability to assess the environmental impact of a system that has temporal and spatial variations. The temporal and spatial variations in the assessment are not only present in the environmental impacts, but they are also present in the economic information, i.e., the large-scale economic products and the value-added of the system itself.

The results from the case study of the household coffee maker production indicate that resource depletion is the primary environmental impact. The following three major impacts are eco-toxicity, human toxicity, and acidification respectively. The evaluation using other LCIA methods also indicates a similar result; that resource-related impact is the primary contribution to the overall environmental impact.

The bottom line from the examination of the case studies is that there is no clear answer as to which would be the best method. Selecting an appropriate metric depends on the preferences of the users based on the goal and scope definition of the LCA. However, this research has demonstrated that the STM in association with the carrying capacity estimates is a resource depletion-driving metric. The STM in conjunction with the carrying capacity estimates will be useful for an LCA of a resource-intensive system. Furthermore, the STM is also suitable for an LCA that takes into account the temporal and spatial variations along the life cycle stages of the system.

For the fifth hypothesis, the results show that the STM single scores are sensitive to the different carrying capacity values. Sensitivity analyses have been conducted to investigate the sensitivity of the STM scores related to individual carrying capacity estimates. The results demonstrate that the STM scores are sensitive to the change in carrying capacity estimates in particular circumstances, e.g., the STM scores are sensitive related to the limiting carrying capacity. The limiting carrying capacity is the one that dominates the overall Environmental Impact (EI) score. The effects from the changes in minor impacts are barely observable.

A ratio sensitivity analysis has been carried out using the case studies of electrical energy generation sources and the production of basic materials, to investigate the impact on the STM performance in terms of reverse ranking due to the change in individual carrying capacity estimates. The result demonstrates that the ranking of alternatives can be altered due to a marginal change in carrying capacity values.

The above results imply that accurate carrying capacity estimates are required in order to make the STM valid for a wide range of applications. Furthermore, the impact categories taken into account should cover as many impact categories as is practicable. These two aspects can be addressed by the improvement of the carrying capacity estimates using more sophisticated approaches and information in further studies.

15.3 Scholarly Contributions

The eventual goal of this research is to provide a practical basis for achieving a sustainable society by using a credible environmental performance evaluation tool for a product, a service, or a system. In the attempt to accomplish this goal, this research has made the following unique scholarly contributions:

- Develop a threshold-oriented technique to estimate the emission-related environmental carrying capacity;
- Develop a reserve-time horizon technique to estimate the consumption-related environmental carrying capacity;
- Formulate and validate a set of U.S.-based carrying capacity estimates; and
- Evaluate the environmental performance of selected case studies utilizing the STM based on carrying capacity estimates.

15.4 Recommendations for Further Studies

Further advances based on this research are certainly possible. Further needs include the areas of data quality and methodology development. The carrying capacities for more chemicals in human toxicity and eco-toxicity impacts are required to expand the database. Methodologies to determine the generic assessment endpoints are needed for the eco-toxicity carrying capacity development.

Other impacts that could be taken into consideration are land use impact, impact of radioactive wastes, waste management impact, and social and human value impact. The application of weighing techniques may be used in order to take into account the social, human value, and economic impacts.

A study to develop new approaches for resource depletion carrying capacity and to modify the STM calculations should be carried out. This may produce a less input-oriented performance evaluation using the STM. Further study may be extended to an evaluation of carrying capacity that is based not only on the natural absorbing capability, but is also influenced by human activities, which may either expedite or delay the natural absorbing capability. There may also be some other ways to apply carrying capacity estimates in this research for other purposes, e.g., policy making and regulatory decision-making. The carrying capacity estimates for parts of the world other than Europe or North America may be developed to make use of the STM more universal.

Lastly, to make the STM implementation more practical, a software tool that integrates the LCI, the STM calculations, and the carrying capacity database as a supporting reference source may be developed. This software tool will minimize time use in the entire LCA procedures. Furthermore, the carrying capacity database may be modifiable by users to make the STM flexible for a broad spectrum of LCA applications.

APPENDIX A

REVIEW OF ENVIRONMENTAL PERFORMANCE ASSESSMENT FRAMEWORKS AND APPROACHES

1. The Importance of Environmental Performance Metrics

Environmental performance metrics are a method for measuring and gauging the progress toward environmental goals and related business strategies (Characklis and Richards 1999). Environmental performance metrics are also being used as a communication tool for a variety of stakeholders including internal decision makers, regulators, public, and customers. Internally, environmental performance metrics can be used as supporting information for decision-making. Externally, environmental performance metrics are commonly found in the environmental reporting of businesses to inform their stakeholders. According to Line et al. (2002), about half of the top 100 global companies produced environmental and social reports in 2001.

O'Reilly and et al. (2000) and Raynolds (1997) also pointed out that the benefits of the environmental performance evaluation are that they:

- help to track the environmental performance of an organization;
- help to measure the organization's eco-efficiency;
- help with the identification of significant aspects and the quantification of objectives and targets in an organization;
- help to track costs and revenues associated with environmental activities and programs;
- help with meaningful and useful reporting and allowing for the demonstration of strong performance;
- provide a clear focus for the organization's environmental management efforts;
- provide an indication of the environmental risk faced by the organization;

- provide information for the comparison of division performance;
- provide a comparison tool of the organization's performance with the other similar organizations;
- motivate and promote awareness for different parties within an organization;
- provide a structure to demonstrate the support for the environmental objectives of stakeholders and customers; and
- impart information to regulators and shareholders.

2. Basic Construction of Environmental Performance Metrics

Environmental performance metrics can be classified into groups and categories according to several criteria. For example, ISO 14031 classifies environmental performance metrics according to their utility into three categories: operational performance indicators (OPIs), management performance indicators (MPIs), and environmental condition indicators (ECIs) (ISO 1999 and O'Reilly et al. 2000). Loew and Kottmann (1996 as cited in Olsthoorn et al. 2001) classified environmental performance indicators into different levels according to environmental protection areas, system boundaries, and levels of analysis/presentation (Table A.1). As pointed out by AIChE (2001), basic environmental performance indicators/metrics can be material intensity, energy intensity, water use, pollutant release, human health, and ecotoxicity. Other criteria include performance management, application within an environmental management system, reliability of data collection, and the internal- or external-based purposes (Olsthoorn et al. 2001).

A comparative assessment between products/services can be carried out using environmental-related indicators. Environmental performance indicators can be expressed as absolute or relative indicators (as in the example shown in Table A.2), as indexed indicators (percentage with respect to total), as aggregated indicators (figures of the same unit are summed), and weighted evaluations (conversion factors are used to depict figures of varying importance) (ISO 1999).

Table A.1 Classification of Environmental Performance Indicators
(Source: Olsthoorn et al. 2001)

Criteria	Indicators
Environmental protection areas	Energy Transport Emissions Waste Packaging Production Stock-keeping Water management
System boundaries	Site/company Process Product
Levels of analysis/representation	Material and energy flow level Flow quantities from different boundaries Polluter level Cause of energy and material flows Cost level Energy and material flow-induced costs Effect level Environmental impacts/effects

Table A.2 Examples of Environmental Performance Indicators
(Source: Jasch 2000)

Indicator	Absolute	Relative
Production output (PO) in kg, items, etc.		
Raw material consumption	kg	kg/PO
Energy consumption	kWh	kWh/PO
Water consumption	m^3	m^3/PO
Total wastes	kg	kg/PO
Waste qualities	kg	In % of total wastes
Wastewater	m^3	m^3/PO
Air emissions	kg	kg/PO

Environmental performance indicators are highly diversified based on the approach or the framework used in the environmental assessment or environmental reporting. Recently, many businesses have adopted the concept of the sustainable development as their primary value (Schwarz et al. 2002). Consequently, the sustainability concept has been put into practice by identifying the indicators or metrics measuring the environmental performance as to its sustainability. And since the sustainability concept consists of three dimensions (triple bottom line), therefore, environmental performance metrics toward sustainability or sustainability metrics can then be defined as the metrics that are designed to consolidate key measures of environmental, economic, and social performance.

There are a number of environmental performance evaluation frameworks available for reporting environmental performance indicators nowadays. However, there is as yet no standard or perception of what indicators must be included in an environmental performance assessment. This is because environmental performance assessment or environmental reporting is mainly voluntarily. Furthermore, the availability of data also plays an important role in the process of indicator selection for the assessment. As summarized by NAE (1998) and Olsthoorn et al. (2001), the areas that need to be refined in order to make the use of environmental performance assessment more homogeneous are:

- The need for more standardization and practicable;
- Measurement of sustainability;
- Life cycle assessment framework; and
- A narrower but deeper analysis of environmental impacts.

Among the frameworks that have been developed as a standard framework for presenting environmental performance metrics toward sustainability, the ones that are worth noting include the Eco-Efficiency (WBCSD 2000), the GRI's Sustainability Report (GRI 2002), the ISO 14031 (ISO 1999), and the Ecological Footprint (Wackernagel et al. 1999). Life cycle consideration can be incorporated into the environmental performance evaluation by following the SETAC' LCA (Society of Toxicology and Chemistry's Life Cycle Assessment) framework (SETAC 1991). A narrow and deep analysis of environmental impacts can be implemented through the life cycle impact assessment (LCIA) step in LCA.

3. Eco-Efficiency

The World Business Council for Sustainable Development (WBCSD 2000), states that a basic business contribution to sustainable development is eco-efficiency, which is:

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

According to Raynolds (1997), a basic form of eco-efficiency environmental performance indicators is the link between resource inputs and/or pollution outputs and units of products or services. In other words, the higher eco-efficiency implies the creating of more goods and services with less use of resource, waste, and pollution.

WBCSD identified that business can use the following seven procedures to improve eco-efficiency: reduce material intensity, reduce energy intensity, reduce dispersion of toxic substances, enhance recyclability, maximize use of renewable resources, extend product durability, and increase service intensity. Eco-efficiency combines the two eco-dimensions of economy and ecology to relate product or service value to environmental influence. The measuring of eco-efficiency is widely used as the following ratio:

$$\textit{Eco-Efficiency} = \frac{\textit{Product or Service Value}}{\textit{Environmental Influence}} \qquad (A.1)$$

WBCSD (2000) proposes that an environmental report should include the following elements: organization profile, value profile, environmental profile, eco-efficiency ratios, and methodological information. The applicable consensus core indicators for value profile include net sales or quantity of goods or services produced or provided to customers. The applicable consensus core indicators for environmental profile include energy consumption, material consumption, water consumption, greenhouse gas emissions, and ozone depleting substance emissions. Other potential indicators include additional financial value indicators for value profile and acidification emissions and total wastes for environmental profile. However, the indicators may not be limited to these core and other potential indicators. Other business specific indicators may also be reported as well. Table A.3 presents some indicators proposed by WBCSD.

The eco-efficiency framework has established a concept of how to relate an environmental impact to value added of a product/service. The comparative assessment among products, processes, or services may be implemented using eco-efficiency ratios.

'Eco-efficiency' seems to be one of the most widely used indicators for environmental reporting these days. WBCSD has developed the basic elements that are required in an environmental report including the eco-efficiency. This is an attempt to standardize the environmental report. However, scientific backgrounds and technical procedures of how to work from the first step to complete the report are not provided. It is left to businesses/organizations to use any appropriate methodologies to evaluate individual indicators.

Table A.3 Eco-Efficiency Indicators Proposed by WBCSD
(Source: WBCSD 2000)

Generally Applicable Indicators	Unit
Value Indicators	
Quantity	As appropriate number/mass
Net sales	Currency
Environmental Influence Indicators	
Energy consumption	Gigajoules
Material consumption	Metric tons
Water consumption	Cubic meters
Ozone depleting substance emissions	Metric tons of CFC-11 equivalent
Greenhouse gas emissions	Metric tons of CO_2 equivalent
Potential Applicable Indicators	
Value Indicators	
Net profit/earnings/income	Currency
Environmental Influence Indicators	
Acidification emissions to air	Metric tons of SO_2 equivalent
Total wastes	Metric tons
Examples of Business Specific Indicators	
Value Indicators	
Profit	Currency
Gross margin	Currency
Value added	Currency
Environmental Influence Indicator	
Priority heavy metal emissions to surface water	Metric tons of Cu equivalent
Waste to landfill	Metric tons
Waste to incineration	Metric tons
Photochemical oxidant creation	Metric tons of Ethylene equivalent
Eutrophication emissions to surface water	Metric tons of Phosphorus equivalent
COD to surface water	Metric tons of Oxygen equivalent
Packaging	Metric tons
GHG emissions from purchased electricity	Metric tons of CO_2 equivalent

4. GRI's Sustainability Report

The Global Reporting Initiative (GRI) was convened in 1997 by the Coalition for Environmental Responsibility Economics (CERES) in partnership with the United Nations Environmental Programme (UNEP). The purpose of the establishment of the GRI was to standardize the sustainability reporting practices. GRI released the first version of Sustainability Reporting Guidelines in 2000 and the second version in 2002. These guidelines demonstrate the framework of how to organize a sustainability report. The GRI's Sustainability Report is to be used in the organization level. The performance indicators listed in the guidelines cover all three dimensions of sustainability. The guidelines specify that the content of a GRI-based report should include (GRI 2002):

- Vision and strategy;
- Profile;
- Governance structure and management systems;
- GRI content index; and
- Performance indicators

The major part of a GRI-based report is performance indicators. The performance indicators are grouped in terms of the three dimensions of the sustainability: economic performance indicators, environmental performance indicators, and social performance indicators. Within these three categories, indicators are grouped into sub-categories and aspects totaling 13 economic performance indicators, 35 environmental performance indicators, and 49 social performance indicators (Table A.4). However, GRI states that sometimes these three categories may not totally capture the performance of an organization for a number of reasons (GRI 2002). Therefore, a fourth dimension of performance indicators, the integrated indicators, is established. The integrated indicators may be system indicators or cross-cutting indicators. System indicators relate the activity of an organization to the larger economic, environmental, and social systems. Systems indicators also relate a performance to the limit or capacity of the system. In other words, systems indicators provide the degree of which an organization's performance influences the performance of a larger system. Cross-cutting indicators directly relate two or more dimensions of economic, environmental, and social performance as a ratio. Therefore, the eco-efficiency measures (Product or service value/Environmental influence) are the best example of cross-cutting indicators. GRI has a very comprehensive list of indicators toward sustainability. However, the technical protocols are still under development. These protocols will provide the details of how to estimate individual indicators.

Table A.4 Performance Indicators of the GRI
(Source: GRI 2002)

Economic Performance Indicators	
Core Indicators	Additional Indicators
Customers	
Monetary flow indicator - Net sales - Geographic breakdown of markets	
Suppliers	
Monetary flow indicator - Cost of all goods, materials, services purchased - Percentage of contracts that were paid	- Supplier breakdown by organization and country
Employees	
Monetary flow indicator - Total payroll and benefits	
Providers of Capital	
Monetary flow indicator - Distributions to providers of capital - Increase/decrease in retained earnings	
Public Sector	
Monetary flow indicator - Total sum of taxes - Subsidies received - Donations	- Total spent on non-core business infrastructure development
Indirect Economic Impacts	
	- The organization's indirect economic impacts

Table A.4 Performance Indicators of the GRI (Continued)

Environmental Performance Indicators	
Core Indicators	Additional Indicators
Materials	
- Total material use - Percentage of materials used that are wastes	
Energy	
- Direct energy use segmented by primary source - Indirect energy use	- Initiatives to use renewable energy sources and to increase energy efficiency - Energy consumption footprint - Other indirect energy use and implications
Water	
- Total water use	- Water sources and ecosystems affected by water use - Annual withdrawal of ground/surface water - Total recycling and reuse of water
Biodiversity	
- Location and size of land used in biodiversity-rich areas - Description of major impacts on biodiversity	- Total of land used for production activities - Amount of impermeable surface as a percentage of land purchased or leased - Impacts of activities and operations on protected and sensitive areas - Changes to natural habitats resulting from activities - Objectives, programs, and targets for protecting and restoring native ecosystems - Number of species with habitats in areas affected by operations - Business units currently operating or planning operations in protected areas
Emissions, Effluents, and Waste	
- Greenhouse gas emissions - Ozone-depleting substance use and emissions - Air emissions by type - Total amount of waste by type and destination - Discharges to water by type - Spills of chemicals, oils, and fuels	- Other indirect GHG emissions - Hazardous waste management - Water resources affected by discharges of water and runoff
Suppliers	
	- Performance of suppliers relative to environmental components of programs and procedures
Products and Services	
- Environmental impacts of products and services - Recycling and reuse percentage and claim - Incidents and fines of compliance violation	
Transport	
	- Environmental impacts of transportation
Overall	
	- Total environmental expenditures by type

Table A.4 Performance Indicators of the GRI (Continued)

Social Performance Indicators: Labor Practices and Decent Work	
Employment	
- Breakdown of workforce - Employment creation	- Employee benefits
Labor/Management Relations	
- Percentage of employees represented by trade union organization - Policy and procedures involving information with employees	- Provision for worker representation in decision-Making
Health and Safety	
- Recording on accidents and diseases - Description of formal joint health and safety committee - Standard injury, lost day, and absentee rates - Description of policies or programs on HIV/AIDS	- Evidence of substantial compliance with health and safety guidelines - Description of formal agreements with trade unions
Training and Education	
- Average hours of training per year per employee	- Description of programs to support the continued employability - Specific policies and programs for skills management
Diversity and Opportunity	
- Description of equal opportunity policies - Composition of senior management	
Social Performance Indicators: Human Rights	
Strategy and Management	
- Description of issues to deals with aspects of human rights - Evidence of consideration of human rights impacts - Description of policies to address human rights aspects	- Employee training on policies concerning aspects of human rights
Non-discrimination	
- Description of global policy preventing discrimination	
Freedom of Association and Collective Bargaining	
- Description of freedom of association policies	
Child Labor	
- Description of policy excluding child labor	
Forced and Compulsory Labor	
- Description of policy preventing forced and compulsory labor	
Disciplinary Practices	
	- Description of practices including human rights issues - Description of non-retaliation policy and effective, confidential employee grievance system
Security Practices	
	- Human right training for security personnel
Indigenous Rights	
	- Description of policies to address the needs for indigenous people - Description of jointly managed community grievance mechanisms/authority - Share of operating revenues from the area of operations that are redistributed to local communities

Table A.4 Performance Indicators of the GRI (Continued)

Social Performance Indicators: Society	
Community	
- Description of policies to manage impacts on communities	- Awards received to social, ethical, and environmental performance
Bribery and Corruption	
- Description of the policy addressing bribery and corruption	
Political Contributions	
- Description of policy for managing political lobbying and contributions	- Amount of money paid to politics
Competition and Pricing	
	- Court decisions regarding cases pertaining to anti-trust and monopoly regulations - Description of policy for preventing anti-competitive behavior
Social Performance Indicators: Product Responsibility	
Customer Health and Safety	
- Description of policy for preserving customer health and safety during use of products/services	- Violations concerning health and safety of customers - Complaints about health and safety of products - Voluntary compliance
Products and Services	
- Description of policy related to product information and labeling	- Violations concerning product information and labeling - Description of policy concerning customer satisfaction
Advertising	
	- Description of policy related to advertising - Number of breaches of advertising and marketing regulations
Respect for Privacy	
- Description of policy for consumer privacy	- Number of complaints regarding breaches of consumer policy

5. ISO 14031

ISO 14031 is a standard which has been developed to compliment the ISO 14000 series (ISO 1999). This standard focuses on the development and application of environmental performance indicators that may be used by an organization for environmental management and environmental reporting. ISO 14031 identifies key performance indicators into two categories of environmental performance indicators and another category of environmental condition indicators.

Operational Performance Indicators (OPIs)

Operational performance indicators or operational metrics generally measure the potential environmental burden in terms of inputs and outputs of energy, raw materials, and waste streams. The operational metrics can be separated into four subcategories (Ditz and Ranganathan 1997 as cited in NAE 1999): material use, energy consumption, non-product output, and pollutant releases.

Management Performance Indicators (MPIs)

Management performance indicators or management metrics provide indicators in terms of allocation of funds and labor, implementation of environmental programs and policies, environment-related legal expenses, environmental remediation activities, and the status of environmental information systems (NAE 1999). These types of metrics are used by management to provide information which forms the basis for decision-making in several areas/fields or to improve the business's environmental performance.

Environmental Condition Indicators (ECIs)

Environmental condition indicators or environmental condition metrics provide information on environmental health and conditions at a local, regional, national, or global level. These metrics link specific business activities or emissions to environmental impacts, in other words, they establish a relationship between pollutants and impacts. Examples of these metrics are the atmospheric ozone concentration, global temperature, and environmental concentrations of pollutants. According to the National Academy of Engineering (NAE 1999), this type of metric is of greatest interest to industry and stakeholders because they may be used to estimate the environmental performance as regards the sustainability of human activities.

ISO 14031 provides the outline of the environmental performance evaluation process following the general steps of ISO 14001 series (Plan-Do-Check-Act):

- Planning environmental performance evaluation (Plan);
- Developing and using data and information (Do);
    - Collecting data
    - Analyzing & converting data
    - Assessing information
    - Reporting and communication; and
- Reviewing and improving environmental performance evaluation (Check & Act).

Examples of operational performance indicators suggested by ISO 14031 are summarized in Table A.5. Selecting indicators is at the discretion of the organization and the standard only specifies that these indicators include the following features:

- Comparability: the indicators must be comparable and reflect changes in environmental performance;
- Target-oriented: the indicators must act toward goals;
- Balanced: the indicators must reflect environmental performance in a concise manner, stating problems as well as benefits in a balance manner;
- Continuity: the indicators must be derived by the same criteria through corresponding time series and units;
- Frequency: the indicators must be derived frequently enough so that action can be taken in due time; and
- Comprehensibility: the indicators must meet the requirements of the users and must be understandable.

Table A.5 Examples of MPI and OPI

Management Performance Indicators
Implementation of policies and programs - Number of achieved objectives and goals - Number of organizational units achieving environmental objectives and goals - Degree of implementation of specified codes of management or operating practice
Conformity - Degree of compliance with regulations - Number of non-compliances - Number of or costs attributable to fines and penalties
Financial performance - Costs that are associated with a product's or process environmental aspects - Return on investment for environmental improvement projects - Savings achieved through reductions in resource usage, prevention of pollution or waste recycling
Community relations - Number of inquiries or comments about environmentally related matters - Number of press reports on the organization's environmental performance - Number of environmental educational programs of materials provided for the community
Operational Performance Indicators
Materials - Quantity of materials used per unit of product - Quantity of processed, recycled or reused materials - Quantity of water per unit of product
Energy - Quantity of energy used per year or per unit of product - Quantity of energy used per service or customer - Quantity of each type of energy used
Services supporting the organization's operations - Amount of hazardous materials used by contracted service providers - Amount of cleaning agents used by contracted service providers - Amount or type of wastes generated by contracted service providers
Physical facilities and equipment; supply and delivery - Average fuel consumption of vehicle fleet - Total land area used for production purposes - Number of business trips by mode of transportation
Products - Number of products introduced in the market with reduced hazardous properties - Number of products which can be reused or recycled - Rate of defective products
Services provided by the organization - Amount of cleaning agent used per square meter - Amount of fuel consumption - Quantity of licenses sold for improved processes
Wastes - Quantity of waste per year or per unit of product - Total waste for disposal - Quantity of hazardous, recyclable or reusable waste produced per year
Emissions - Quantity of specific emissions per year - Quantity of specific emissions per unit of product - Quantity of waste energy released to air
Effluents to land or water - Quantity of specific material discharged per year - Quantity of specific material discharged to water per unit of product - Quantity of material sent to landfill per unit of product
Other emissions - Noise measured at a certain location - Quantity of radiation released - Amount of heat, vibration of light emitted

Although ISO 14031 is a standard, it does not stipulate minimum performance standards or specific reporting requirements. However, this disadvantage can also be viewed as an advantage in that the standard is flexible enough to be applied to a wide range of organizations and scenarios, especially for the small and medium-sized enterprises because it is cost effective. Since it is just a guideline however, scientific backgrounds and technical procedures are not provided.

6. Ecological Footprint

Unlike other frameworks for environmental performance assessment reviewed here, the Ecological Footprint is a measuring tool for sustainability of ecological resources (Wackernagel 1999). This method uses area requirements (footprint area) for environmental aspects as the aggregated indicator to measure sustainability. The environmental aspects such as human consumption are translated into areas of productive land that are required to provide resources and to assimilate waste products. The area requirement of the ecological footprint is expressed on the basis of per capita per year.

The ecological footprint tracks the energy and resources throughput and translates them into areas of biological production that are required to produce these inputs. For instance, the footprint of a crop production is calculated from the yield of the crop per unit area, the footprint of fossil fuel consumption is calculated from the absorption capacity of carbon dioxide through planting trees, the footprint of material consumption is calculated from the embodied energy.

This method also compares the consumption of resources and energy to the ecological capacity available in the country, which is the concept of sustainability (in other words, its sustainability). When the required amount is greater than the total area where the people live, the sustainability is not met. Indicator such as area requirements is a simple way to communicate the impact of anthropogenic activities to the audience.

The calculation of a national footprint is derived from national consumption in biophysical terms, using agricultural or biological yield figures to translate the consumption into areas of biological production and, finally, aggregating the results into a total footprint area. The statistical data used to derive the conversion factors for a national footprint of consumption and supply capacity are obtained from several sources. Table A.6 presents the impact categories that are taken into consideration for estimating national footprint.

Table A.6 Impact Categories in Footprint Calculations
(Source: ICLEI 2003)

Category	Unit
Foods	kg/ha
Timber	m^3/ha
Other crops	kg/ha
Energy balance	Gj/ha/yr
- Fossil fuels	Gj/ha/yr
- Nuclear energy	Gj/ha/yr
- Hydro energy	Gj/ha/yr
- Wood energy	Gj/ha/yr
- Embodied energy in imported goods	Gj/ton

The ecological footprint is among the most popular metrics for environmental reports at the national level. This is attributed by its ability to present technical information such as sustainability in a simple and straightforward manner for 152 nations as documented in the "Living Planet Report" (WWF 2002). In this report, the estimated footprints are that of cropland, grazing land, forest, fishing ground, energy, and water withdrawal. There is also an application of the ecological footprint at the local scale in Barrett and Scott (2001). In a modified calculation of the ecological footprint for energy for an energy-planning project, the substitution of fossil fuel by potential renewable energy can also be described (Stoglehner 2003).

The advantage of the ecological footprint is that the measure is translated in terms of an aggregated indicator. By using the area requirements, the comparative assessment for the sustainability between nations/demographic areas can be carried out. The simple criteria of the sustainability are twofold: the finite land area (bio-capacity) and the population density. Up to date, the ecological footprint provides the footprint calculations for 152 countries.

The drawback of the ecological footprint is its limited impact categories. The environmental impacts that are left out are fresh water use, disposal of wastes and contaminants. For the consumption impact, the resources taken into account are for basic human needs (food, energy, and limited number of goods and supplies). This limitation is due to unavailability of data. Because of this limitation, it is difficult to apply the ecological footprint concept to the supply line level.

7. Life Cycle Assessment

Life cycle assessment (LCA) is a framework for evaluating the life cycle environmental impacts of a product or process. U.S. EPA (1993) defines LCA as follows:

> "A concept and methodology to evaluate the environmental effects of a product or activity holistically, by analyzing the whole life cycle for a particular product, process, or activity."

According to Curran (2000), there have been several efforts to develop LCA methodology since the 1970's. In the 1990's, the Society of Environmental Toxicology and Chemistry (SETAC) in North America and the EPA sponsored projects to develop and promote a consensus on a framework for conducting life cycle assessments (SETAC 1991). Similar efforts have been undertaken by other international organizations such as SETAC-Europe and the International Organization for Standardization (ISO). As a result, a consensus among these agencies has been achieved on an overall LCA framework and methodology. However, the main problem with LCA is the lack of uniform international standards in carrying out the details of an assessment. Because LCA is increasingly practiced especially in Europe, ISO has adopted standards and guidelines for conducting LCA (Table A.7). The International Organization for Standardization has formalized the 14040 series following the LCA framework proposed by SETAC.

Table A.7 ISO 14040 Series
(Source: ISO 2002)

Designation	Year	Title
ISO 14040	1997	Environmental management-Life cycle assessment-Principles and framework
ISO 14041	1998	Environmental management-Life cycle assessment-Goal and scope definition and inventory analysis
ISO 14042	2000	Environmental management-Life cycle assessment-Life cycle impact assessment
ISO 14043	2000	Environmental management-Life cycle assessment-Life cycle interpretation
ISO/TR 14047	To be determined	Environmental management-Life cycle assessment-Examples of application of ISO 14042
ISO/TR 14048	2002	Environmental management-Life cycle assessment-Data documentation format
ISO/TR 14049	2000	Environmental management-Life cycle assessment-Examples of application of ISO 14041 to goal and scope definition and inventory analysis

Life cycle assessment is considered a "cradle-to-grave" approach. "Cradle-to-grave" begins, for example, with the mining of raw materials from their ore to create products and services and ends when the residual materials are sent to its ultimate disposal site. Life cycle assessment evaluates all stages of a product's life and estimates the cumulative environmental stressors for individual impact categories. The LCA conceptual framework proposed by SETAC consists of four major parts or components: goal and scope definition, life cycle inventory, life cycle impact assessment, and life cycle interpretation (SETAC 1991; EPA 1993; ISO 1998a). Figure A.1 illustrates the interrelation among these four LCA components.

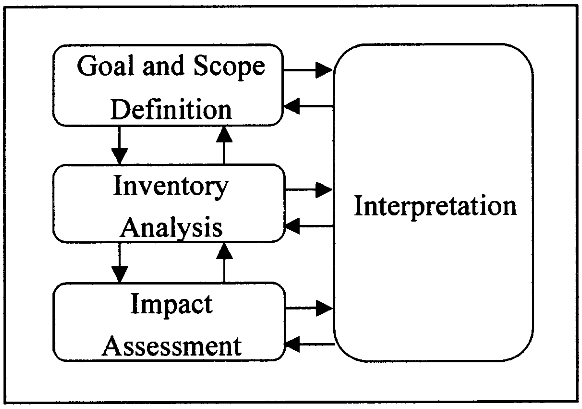

Figure A.1 Life cycle assessment framework.
(Source: Fava 1993)

Goal and scope definition defines the purpose and boundary of the study. Life cycle inventory (LCI) is the process used to quantify the energy and raw material inputs and environmental outputs associated with each stage of the product's life cycle. Life cycle impact assessment (LCIA) applies the results from the inventory to assess the impact on the environment and human health. Interpretation is the evaluation of the alternatives in the study and summary of the opportunities to reduce the environmental impacts during the life cycle stages. Among the four components of LCA, LCIA is considered as the most important (Curran 2000).

Life cycle impact assessment builds on the LCI, which lists the emissions and resource consumption throughout the different stages of a product. Life cycle impact assessment is the evaluation of potential human health and environmental impacts of resource consumption and environmental releases identified during the LCI. The impact assessment addresses ecological and human health effects as well as resource depletion. Furthermore, LCIA establishes a linkage between a product or process and its potential environmental impacts. In theory, LCIA converts the results of an LCI to a set of common impact categories such as global warming, ozone depletion, and acidification.

An LCIA also provides a systematic procedure for classifying, characterizing, and weighting the environmental impacts. During classification, the environmental stressors are categorized to different impact categories. In the characterization, the stressors in each impact category are aggregated and compared to form a single category indicator. In some advanced LCIA methods, category indicators are weighted and combined to form a measure of overall environmental performance of the product (single indicator or single-score indicator). A typical concept of LCIA requires that information about loadings (environmental emissions) and resource uses are in the form of a numerical indicator or index for each impact category. These indicators or scores can then be used as the basis for comparison of products' environmental performance.

The simplest form of an LCA result and report is presented in terms of environmental burden (or environmental impact) characterized by the same functional unit as is used for individual impact categories. A functional unit can be defined as the product or a set of products that provide the same service (Hofstetter et al. 2000).

An example of an LCA result, the environment burden (EB), of an impact category can be calculated using a simple formula (ICI 1997):

$$EB = (W_a \times PF_a) + (W_b \times PF_b) + (W_c \times PF_c) + \ldots \qquad (A.2)$$

Where W denotes the weight of substance emission (a, b, c, ...) and PF is the specific potency factor. The potency factor, or equivalency factor, can be the potential of a chemical to cause damage to the environment in a specific category. An impact score for an impact category can then be generally written as (Olsen and Hauschild 1998):

$$Impact\ Score_{category} = \sum_{Subst} Equivalency\ factor_{category,\ subst} \times Emission_{substance} \qquad (A.3)$$

Conventional LCA is a multiple indicator system because the environmental performance is reported corresponding to the number of environmental impacts or environmental aspects associated with the business's activities. The conventional LCA assesses the environmental impact in terms of the potential of emissions to cause environmental damage. A comparative assessment of a conventional LCA is based on the "less-is-better" concept (the one posing less environmental stressors is the one with better environmental performance). The main limitation of the conventional LCA is that the environmental impacts cannot be added together to obtain an overall environmental impact unless the weighting factors are provided. Available weighting techniques are based primarily on value or expert judgment. Other limitations include the lack of temporal and spatial considerations, data gap and data quality, missing an environmental condition indicator, a limited number of environmental impacts taken into consideration, uncertainty analysis, and necessity of further development (Finneden 2000).

APPENDIX B

THE STM METHODOLOGY

The STM Methodology[1]

This environmental metric framework is "universal" in the sense that it possesses numerous necessary attributes, including: applicability to the full product lifecycle; compatibility with EMS and LCA method, databases, and standards; means for communicating with both suppliers and customers; and the suitability as a standard. It can be applied to any business, product or service. RP, used as a cumulative measure of environmental impact, offers an alternative to the supply line LCA. EE extends RP, based on the value provided by the business, to provide an absolute indicator for sustainability. The method can be integrated with DFE procedures and CAD systems, and guides investment decisions by providing criteria for project selection. A brief summary and simple example of the method is included also in a previous IEEE ISEE paper on competitive advantage and sustainability[2]. A case study application for a specific product is given in another related paper[3].

Environmental impact (EI) resulting from an activity such as manufacturing a product or providing a service, is quantified by adding normalized environmental aspect

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[1] From Dickinson, D.A., Mosovsky, J.A., and Morabito, J. (2001). Sustainability: An Evaluation & Target Method for Businesses. Summary & Reference Levels. Lucent Technologies: Bell Laboratories Technical Memorandum. May 2001.

[2] Mosovsky, J.A., Dickinson, D.A., and Morabito, J.M. (2000). "Creating Competitive Advantage Through Resource Productivity, Eco-Efficiency, and Sustainability in the Supply Chain". Proceedings of the 2000 IEEE International Symposium on Electronics and the Environment. San Francisco CA, May 2000.

[3] Mosovsky, J.A., Dispenza, J., Dickinson, D., Morabito, J., Caudill, R., and Alli, N. et al. (2001). "Assessing Product Design Alternatives With Respect to Environmental Performance and Sustainability: A Case Study for Circuit Pack Faceplates". Proceedings of the 2001 IEEE international Symposium on Electronics and the Environment. Denver CO, May 2001.

levels to obtain an aggregated and dimensionless quantity (for selected scope: supply line, factory, customer use, full life cycle, etc):

$$EI = A_1/A_{R1} + A_2/A_{R2} + A_3/A_{R3} + \ldots + A_N/A_{RN} \qquad (B.1)$$

Where $A_{1,2,3,\ldots,N}$ = each Aspect Level, e.g., kWh/year for energy, lb/year for consumption or waste

$A_{R1,R2,R3,\ldots,RN}$ = the Aspect Reference Level indicating the level at which the aspect would have a significant environmental impact, i.e., the level at which it would become non-sustainable for a "reference firm" of given size (specified in terms of its rate of value generation, V_R).

Resource Productivity (RP) and Value Productivity (VP), the production rate and value generation achieved per unit of environmental impact (or resulting in a unit of life cycle impact), are then:

$$RP = P/EI \quad \text{and} \quad VP = V/EI \qquad (B.2)$$

Where: P = production rate (product units/year)

V = value creation, e.g., revenue (market value of the product or service) or "value added", depending on the scope of EI ($/year).

RP and VP serve as relative indicators of environmental impact that allows direct comparison, such as between products or alternative product designs. When determined for energy, raw materials, supplies, etc., RP also serves as the Aspect Reference Level for these consumption "aspects".

Eco-Efficiency (EE) then is defined as:

$$EE = \beta/EI \qquad (B.3)$$

Where: β = V/V_R, i.e., the value ratio (dimensionless)

V_R = the Value Reference Level corresponding to the Aspect Reference Levels ($/year).

EE serves as an absolute indicator of sustainability. It is essentially the actual rate of value generation, V, normalized to the rate that is sustainable at the actual level of environmental impact ($V_R \times EI$). It can be viewed also as the ratio of sustainable to actual impact. The free market value can be further adjusted, if desired, for various social considerations, e.g., essentials, vs. luxuries.

Production rate and revenue may not yet be estimated or may be very uncertain when a product design is still in progress. Also, LCA databases typically do not present data in terms of rates. EI can be expressed alternatively in terms of per-product-unit quantities by dividing by production rate P to obtain EI_{pr}, the environmental impact per unit production rate:

$$EI_{pr} = EI/P = (A_1^*/A_{R1} + A_2^*/A_{R2} + A_3^*/A_{R3} + \ldots + A_N^*/A_{RN}) \qquad (B.4)$$

Then, combining Equations 2,3, and 4:

$$RP = 1/EI_{pr} \quad \text{and} \quad VP = Price \times RP \qquad (B.5)$$

$$EE = VP/V_R \qquad (B.6)$$

Where: A^* = A/P, the aspect quantity per product unit (e.g., kWh/unit)

Price = V/P, the revenue per product unit (market price) or value added per unit ($/unit).

For each aspect, the reference level A_R depends on the environmental impacts (or "effects") created by the aspect, the associated carrying capacities, and the relationship of V_R to total economic output. For each impact, the "natural carrying capacity", NCC, minus the natural burden, NB, is the "economic carrying capacity", ECC, i.e., that portion available to industry and commerce to support the needs of society. For each impact, part of ECC can be associated with the reference firm in the same proportion as the ratio of V_R to total global or regional economic output, depending on the geographical scale of the impact (see also Figure A.1). This is the "Impact Reference Level", I_R:

$$I_R\ (Global) \quad = \quad ECC\ (Global) \times V_R/GGP \qquad (B.7)$$

or

$$I_R\ (Regional) \quad = \quad ECC\ (Region) \times V_R/GDP \qquad (B.8)$$

Where GGP and GDP are total global and regional (e.g., national) economic product ($/year), respectively. (Regional Impact Reference Levels can be adjusted further for specific local issues based on local conditions, e.g., air and water quality or rainfall compared to the regional average.) The Impact Reference Level I_R is defined as the maximum level at which the impact is sustainable for the reference firm, i.e., the level at which its environmental impact is the same in proportion to carrying capacity as the contribution of its value generation V_R to total economic output.

The Impact Reference Levels are combined to produce the Aspect Reference Level. Each Impact Reference Level first is converted to an Aspect Equivalence Level, A_{EQ} (e.g., for the electrical energy consumption aspect and the global warming impact, A_{EQ} would be the rate of energy generation that produces CO_2 at the rate I_R). Then for each aspect the following equation must be satisfied:

$$A/A_R \quad = \quad A/A_{EQ1} + A/A_{EQ2} + A/A_{EQ3} + \ldots + A/A_{EQn} \qquad (B.9)$$

Where: A = the Aspect Level

A_R = the Aspect Reference Level to be determined

$A_{EQ1,2,\ldots,n}$ = the Aspect Equivalence Level for each impact associated with the aspect.

The Aspect Reference Level is then

$$A_R = \frac{1}{(1/A_{EQ1} + 1/A_{EQ2} + 1/A_{EQ3} + \ldots + 1/A_{EQn})}. \qquad (B.10)$$

If A_{EQ} for any one of the impacts is much smaller than the others, this identifies the primary or "limiting" impact. This A_{EQ} can be denoted A_L. Then $A_R = A_L$.

Using the above approach, EI as shown in (1) is really the sum of the normalized Impact Levels. In practice, operating data is collected and maintained and objectives generally will be set on the basis of aspects (e.g., goals to reduce energy or material consumption), so the above approach allows direct use of such data. However, LCA databases typically present LCA inventory data in terms of impacts. Since impact reference levels have been determined in the above approach, the elements EI, RP, and EE also can be calculated using such data when necessary. These elements also can be calculated for each of the aspects or impacts individually, rather than aggregated.

Resource Productivity, RP, depends on the V_R selected and the associated A_R quantities, but this is consistent with its use as a relative indicator only. The choice is arbitrary as long as the same V_R and A_R are used within a given comparison. Further, since each A_R is linearly proportional to V_R, Eco-Efficiency, EE is independent of V_R. This is consistent with its use as an absolute indicator. Most importantly, EE ≥ 1 (100%) indicates that the value provided by the product meets or exceeds that necessary given the level of environmental impact it causes, i.e., the product is sustainable.

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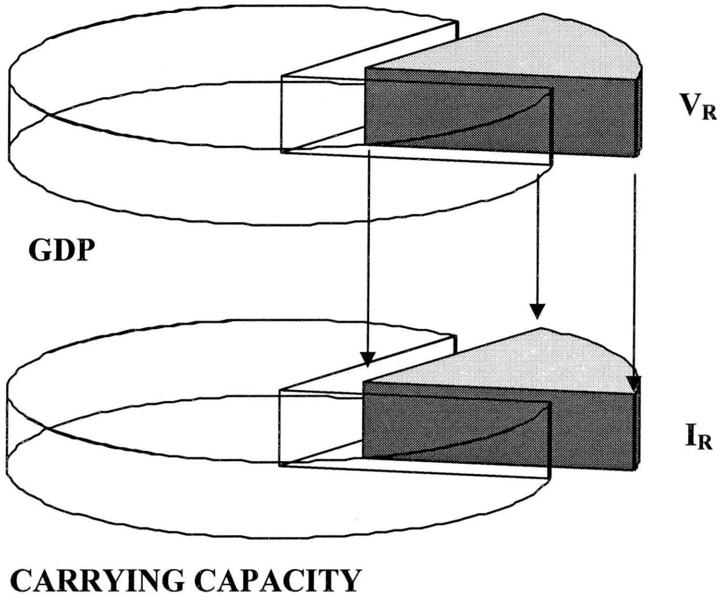

Figure B.1 Linking value generation and environmental impact as a target for sustainability.

APPENDIX C

CHARACTERIZATION FACTORS AND SITE FACTORS

This appendix summarizes the characterization factors for global warming impact (Table C.1), ozone depletion impact (Table C.2), acidification impact (Table C.3(a)), eutrophication impact (Table C.4(a)), and photochemical ozone formation impact (Table C.5(a) and C.5(b)). Site factors for the U.S. states are also provided for acidification impact (Table C.3(b)), eutrophication impact (Table C.4(b)), and photochemical ozone formation impact (Table C.5(c)).

Table C.1 Global Warming Potentials, GWP (100-year time horizon)
(Source: WMO 2003)

Common Name	Chemical Formula	Lifetime (Years)	Global Warming Potential for 100 years time horizon (kg CO_2/kg GHG)
Carbon dioxide	CO_2		1
Methane	CH_4	12.0	23(1)
Nitrous oxide	N_2O	114	300
Chlorofluorocarbon			
CFC-11	CCl_3F	45	4,680
CFC-12	CCl_2F_2	100	10,720
CFC-13	$CClF_3$	640	14,190
CFC-113	CCl_2FCClF_2	85	6,030
CFC-114	$CClF_2CClF_2$	300	9,880
CFC-115	$CClF_2CF_3$	1,700	7,250
Hydrochlorofluorocarbons			
HCFC-21	$CHCl_2F$	1.7	148
HCFC-22	$CHClF_2$	12.0	1,780
HCFC-123	$CHCl_2CF_3$	1.3	76
HCFC-124	$CHClFCF_3$	5.8	599
HCFC-141b	CH_3CCl_2F	9.3	713
HCFC-142b	CH_3CClF_2	17.9	2,270
HCFC-225ca	$CHCl_2CF_2CF_3$	1.9	120
HCFC-225cb	$CHClFCF_2CClF_2$	5.8	586
Hydrofluorocarbons			
HFC-23	CHF_3	270	12,240
HFC-32	CH_2F_2	4.9	543
HFC-41	CH_3F	2.4	90
HFC-125	CHF_2CF_3	29	3,450
HFC-134	CHF_2CHF_2	9.6	1,090
HFC-134a	CH_2FCF_3	14.0	1,320
HFC-143	CH_2FCHF_2	3.5	347
HFC-143a	CH_3CF_3	52	4,400
HFC-152	CH_2FCH_2F	0.60	52
HFC-152a	CH_3CHF_2	1.4	122
HFC-227ea	CF_3CHFCF_3	34.2	3,660
HFC-236cb	$CH_2FCF_2CF_3$	13.6	1,320
HFC-236ea	CHF_2CHFCF_3	10.7	1,350
HFC-236fa	$CF_3CH_2CF_3$	240	9,650
HFC-245ca	$CH_2FCF_2CHF_2$	6.2	682
HFC-245fa	$CHF_2CH_2CF_3$	7.6	1,020
HFC-365mfc	$CH_3CF_2CH_2CF_3$	8.6	782
HFC-43-10mee	$CF_3CHFCHFCF_2CF_3$	15.9	1,610
Chlorocarbons			
Methyl cholroform	CH_3CCl_3	5.0	144
Carbon tetrachloride	CCl_4	26	1,380
Methyl chloride	CH_3Cl	1.3	17
Bromocarbons			
Methyl bromide	CH_3Br	0.7	5
Bromodifluoromethane	$CHBrF_2$	5.8	397
Halon-1211	$CBrClF_2$	16	1,860
Halon-1301	$CBrF_3$	65	7,030
Halon-2402	$CBrF_2CBrF_2$	20	1,620

Table C.1 Global Warming Potentials, GWP (Continued)

Common Name	Chemical Formula	Lifetime (Years)	Global Warming Potential for 100 years time horizon (kg CO_2/kg GHG)
Fully fluorinated species			
Sulfur hexafluoride	SF_6	3,200	22,450
Trifluoromethylsulfurpentafluoride	SF_5CF_3	800	17,500
FC-14	CF_4	50,000	5,820
FC-116	C_2F_6	10,000	12,010
FC-218	C_3F_8	2,600	8,690
FC-31-10	C_4F_{10}	2,600	8,710
FC-318	c-C_4F_8	3,200	10,090
FC-41-12	C_5F_{12}	4,100	9,010
FC-51-14	C_6F_{14}	3,200	9,140
Halogenated alcohols and ethers			
	$(CF_3)_2CFOCH_3$	3.4	338
	$(CF_3)_2CHOH$	2.0	214
HFE-125	CHF_2OCF_3	136	14,670
HFE-134	CHF_2OCHF_2	26	6,220
HFE-143a	CH_3OCF_3	4.3	744
HFCE-235da2	$CHF_2OCHClCF_3$	2.6	343
HFE-245cb2	$CH_3OCF_2CF_3$	5.1	697
HFE-245fa2	$CHF_2OCH_2CF_3$	4.9	649
HFE-254cb2	$CH_3OCF_2CHF_2$	2.6	353
HFE-346mcc3	$CH_3OCF_2CF_2CF_3$	5.2	566
HFE-356pcf3	$CHF_2OCH_2CF_2CHF_2$	3.6	494
HFE-374pc2 [i]	$CH_3CH_2OCF_2CHF_2$	5	548
HFE-7100	$C_4F_9OCH_3$	5	397
HFE-7200	$C_4F_9OC_2H_5$	0.77	56
H-Galden 1040x [j]	$CHF_2OCF_2OC_2F_4OCHF_2$	6.3	1,840
HFE-236ca12	$CHF_2OCF_2OCHF_2$	12.1	2,780
HFE-338pcc13	$CHF_2OCF_2CF_2OCHF_2$	6.2	1,480
Others			
Nitrogen trifluoride	NF_3	740	10,970
Perfluorocyclopropane	c-C_3F_6	>1,000	>17,070
HFE-227ea	$CF_3CHFOCF_3$	11.0	1,520
HFE-236ea2	$CHF_2OCHFCF_3$	5.8	973
HFE-236fa	$CF_3CH_2OCF_3$	3.7	480
HFE-245fa1	$CHF_2CH_2OCF_3$	2.2	282
HFE-329mcc2	$CF_3CF_2OCF_2CHF_2$	6.8	904
HFE-338mcf2	$CF_3CF_2OCH_2CF_3$	4.3	543
HFE-347mcf2	$CF_3CF_2OCH_2CHF_2$	2.8	368
HFE-356mec3	$CH_3OCF_2CHFCF_3$	0.94	99
HFE-356pcc3	$CH_3OCF_2CF_2CHF_2$	0.93	108
HFE-356pcf2	$CHF_2CH_2OCF_2CHF_2$	2.0	260
	$(CF_3)_2CHOCHF_2$	3.1	373
	$-(CF_2)_4CH(OH)-$	0.85	71

Table C.2 Ozone Depletion Potentials, ODP
(Source: UNEP 2000b)

Formula	Chemical name	ODP(1)
Annex A Group I		
CFC-11	trichlorofluoromethane	1
CFC-12	dichlorodifluoromethane	1
CFC-113	1,1,1-trichlorotrifluoroethane	0.8
CFC-114	dichlorotetrafluoroethane	1
CFC-115	monochloropentafluoroethane	0.6
Annex A Group II		
Halon-1211	bromochlorodifluoromethane	3
Halon-1301	bromotrifluoromethane	10
Halon-2402	dibromotetrafluoroethane	6
Annex B Group I		
CFC-13	Chlorotrifluoromethane	1
CFC-111	Pentachlorofluoroethane	1
CFC-112	Tetrachlorodifluoroethane	1
CFC-211	Heptachlorofluoropropane	1
CFC-212	Hexachlorodifluoropropane	1
CFC-213	Pentachlorotrifluoropropane	1
CFC-214	Tetrachlorotetrafluoropropane	1
CFC-215	Trichloropentafluoropropane	1
CFC-216	Dichlorohexafluoropropane	1
CFC-217	Chloroheptafluoropropane	1
Annex B Group II		
CCl_4	Carbon tetrachloride	1.1
Annex B Group III		
$C_2H_3Cl_3$(2)	1,1,1-trichloroethane (methyl chloroform)	0.1
Formula	Chemical name	ODP(3)
Annex C Group I		
$CHFCl_2$	HCFC-21	0.04
CHF_2Cl	HCFC-22	0.055
CH_2FCl	HCFC-31	0.02
C_2HFCl_4	HCFC-121(4)	0.01-0.04
$C_2HF_2Cl_3$	HCFC-122(4)	0.02-0.08
$C_2HF_3Cl_2$	HCFC-123	0.02-0.06
$CHCl_2CF_3$	HCFC-123	0.02
C_2HF_4Cl	HCFC-124	0.02-0.04
$CHFClCF_3$	HCFC-124(4)	0.022
$C_2H_2FCl_3$	HCFC-131	0.007-0.05
$C_2H_2F_2Cl_2$	HCFC-132	0.008-0.05
$C_2H_2F_3Cl$	HCFC-133	0.02-0.06
$C_2H_3FCl_2$	HCFC-141	0.005-0.07
CH_3CFCl_2	HCFC-141b(4)	0.11
$C_2H_3F_2Cl$	HCFC-142	0.008-0.07
CH_3CF_2Cl	HCFC-142b	0.065
C_2H_4FCl	HCFC-151	0.003-0.005
C_3HFCl_6	HCFC-221	0.015-0.07
$C_3HF_2Cl_5$	HCFC-222	0.01-0.09

Table C.2 Ozone Depletion Potentials, ODP (Continued)

Formula	Chemical name	ODP(3)
$C_3HF_3Cl_4$	HCFC-223	0.01-0.08
$C_3HF_4Cl_3$	HCFC-224	0.01-0.09
$C_3HF_5Cl_2$	HCFC-225	0.02-0.07
$CF_3CF_2CHCl_2$	HCFC-225ca(4)	0.025
CF_2ClCF_2CHClF	HCFC-225cb(4)	0.033
C_3HF_6Cl	HCFC-226	0.02-0.10
$C_3H_2Cl_5$	HCFC-231	0.05-0.09
$C_3H_2F_2Cl_4$	HCFC-232	0.008-0.10
$C_3H_2F_3Cl_3$	HCFC-233	0.007-0.23
$C_3H_2F_4Cl_2$	HCFC-234	0.01-0.28
$C_3H_2F_5Cl$	HCFC-235	0.03.-0.52
$C_3H_3FCl_4$	HCFC-241	0.004-0.09
$C_3H_3F_2Cl_3$	HCFC-242	0.005-0.13
$C_3H_3F_3Cl_2$	HCFC-243	0.007-0.12
$C_3H_3F_4Cl$	HCFC-244	0.009-0.14
$C_3H_4FCl_3$	HCFC-251	0.001-0.01
$C_3H_4F_2Cl_2$	HCFC-252	0.005-0.04
$C_3H_4F_3Cl$	HCFC-253	0.003-0.03
$C_3H_5FCl_2$	HCFC-261	0.002-0.02
$C_3H_5F_2Cl$	HCFC-262	0.002-0.02
C_3H_6FCl	HCFC-271	0.001-0.03
Annex C Group II		
$CHFBr_2$		1
CHF_2Br (HBFC-22B1)		0.74
CH_2FBr		0.73
C_2HFBr_4		0.3-0.8
C_2HF_2Br3		0.5-1.8
$C_2HF_3Br_2$		0.4-1.6
C_2HF_4Br		0.7-1.2
$C_2H_2FBr_3$		0.1-1.1
$C_2H_2F_2Br_2$		0.2-1.5
$C_2H_2F_3Br$		0.7-1.6
$C_2H_3FBr_2$		0.1-1.7
$C_2H_3F_2Br$		0.2-1.1
C_2H_4Br		0.07-1.1
C_3HFBr_6		0.3-1.5
$C_3HF_2Br_5$		0.2-1.9
$C_3HF_3Br_4$		0.3-1.8
$C_3HF_4Br_3$		0.5.2.2
$C_3HF_5Br_2$		0.9-2.0
C_3HF_6Br		0.7-3.3
$C_3H_2FBr_5$		0.1-1.9
$C_3H_2F_3Br_3$		0.2-5.6
$C_3H_2F_4Br_2$		0.3-7.5
$C_3H_2F_5Br$		0.9-1.4
$C_3H_3FBr_4$		0.08-1.9
$C_3H_3F_2Br_3$		0.1-3.1

Table C.2 Ozone Depletion Potentials, ODP (Continued)

Formula	Chemical name	ODP⁽³⁾
$C_3H_3F_3Br_2$		0.1-2.5
$C_3H_3F_4Br$		0.3-4.4
$C_3H_4FBr_3$		0.03-0.3
$C_3H_4F_2Br_2$		0.1-1.0
$C_3H_4F_3Br$		0.07-0.8
$C_3H_5FBr_2$		0.04-0.4
$C_3H_5F_2Br$		0.07-0.8
C_3H_6FBr		0.02-0.7
Annex E Group I		
CH_3Br	methyl bromide	0.6

(1) The ozone depleting potentials are estimates based on existing knowledge and will be reviewed and revised periodically.

(2) This formula does not refer to 1,1,2-trichloroethane.

(3) Where a range of ODPs is indicated, the highest value in that range shall be used for purposes of the Protocol. The ODPs listed as a single value have been determined from calculations based on laboratory measurements. Those listed as a range are based on estimates and are less certain. The range pertains to an isomeric group. The upper value is the estimate of the ODP of the isomer with the highest ODP and the lower value is the estimate of the ODP of the isomer with the lowest ODP.

(4) Identifies the most commercially viable substances with ODP values listed against them to be used for the purposes of the Protocol.

Table C.3(a) Acidification Potentials
(Source Norris 2002; EPA 2003a)

Acidification substance (AS)	Acidification potential (AP), kg SO_2/kg AS
SO	1.00
SO_2	1.00
SO_3	1.00
SO_x	1.00
NO	1.07
NO_2	0.70
NO_x	0.70
NH_3	1.88
HCl	0.88
HF	1.60

Table C.3(b) Site Factors for Characterization of Acidifying Substances
(Source Norris 2002; EPA 2003a)

State	SF_{Acid}	State	SF_{Acid}	State	SF_{Acid}
AL	1.21	MD	0.55	SC	0.79
AK	1.00(1)	MA	0.25	SD	0.99
AZ	1.29	MI	0.78	TN	1.14
AR	1.40	MN	1.04	TX	1.51
CA	1.15	MS	1.10	UT	1.55
CO	1.45	MO	1.26	VT	0.50
CT	0.33	MT	1.29	VA	0.72
DE	0.35	NE	1.26	WA	1.22
DC	0.78	NV	1.36	WV	0.69
FL	0.81	NH	0.28	WI	0.97
GA	0.96	NJ	0.41	WY	1.41
HI	1.00(1)	NM	1.49	Regional	
ID	1.52	NY	0.49	NE	0.48
IL	1.19	NC	0.72	MW	1.02
IN	1.13	ND	0.99	S	1.03
IA	1.21	OH	0.81	W	1.34
KS	1.39	OK	1.48	East of Mississippi	0.85
KY	1.09	OR	1.26	West of Mississippi	1.25
LA	1.33	PA	0.59		
ME	0.34	RI	0.27	U.S. Average	1.00

(1) Use U.S. average.

Table C.4(a) Eutrophication Potentials

(Source: Norris 2002; EPA 2003a)

Phosphorus nutrient		
Nutrient	Eutrophication potential (EP-P)	
	(kg P/kg Nutrient)	(kg PO_4^{3-}/kg Nutrient)
Airborne		
P	1.00	3.06
Waterborne		
P	1.00	3.06
PO_4^{3-}	0.33	1.00
BOD	0.007	0.02
COD	0.007	0.02
Nitrogen nutrient		
Nutrient	Eutrophication potential (EP-N) (kg N/kg Nutrient)	
Airborne		
NO	0.47	
NO_x	0.30	
NO_2	0.30	
N_2O	0.64	
NH_3	0.82	
Waterborne		
N	1.00	
NO_3^-	0.23	
NH_4^+	0.78	
BOD	0.05	
COD	0.05	

Table C.4(b) Site-Specific Airborne Transport Factor for Characterization of Nutrients

(Source: Norris 2002; EPA 2003a)

State	$SF_{Eutro,a}$	State	$SF_{Eutro,a}$	State	$SF_{Eutro,a}$
AL	0.093	MD	0.062	SC	0.060
AK	0.069(1)	MA	0.039	SD	0.049
AZ	0.038	MI	0.059	TN	0.097
AR	0.102	MN	0.061	TX	0.066
CA	0.061	MS	0.101	UT	0.060
CO	0.057	MO	0.086	VT	0.061
CT	0.047	MT	0.055	VA	0.062
DE	0.037	NE	0.099	WA	0.082
DC	0.068	NV	0.092	WV	0.069
FL	0.049	NH	0.054	WI	0.066
GA	0.075	NJ	0.052	WY	0.059
HI	0.069(1)	NM	0.048	Regional	
ID	0.088	NY	0.056	NE	0.055
IL	0.080	NC	0.062	MW	0.071
IN	0.083	ND	0.042	S	0.074
IA	0.073	OH	0.072	W	0.061
KS	0.065	OK	0.069	East of Mississippi	0.070
KY	0.087	OR	0.094	West of Mississippi	0.067
LA	0.090	PA	0.064		
ME	0.043	RI	0.033	U.S. Average	0.069

(1) Use U.S. average

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact
(Source: Carter 2000)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
1	CO	Carbon Monoxide	630-08-0	0.058
2	METHANE	Methane	74-82-8	0.014
3	ETHANE	Ethane	74-84-0	0.306
4	PROPANE	Propane	74-98-6	0.560
5	N-C4	n-Butane	106-97-8	1.329
6	N-C5	n-Pentane	109-66-0	1.544
7	N-C6	n-Hexane	110-54-3	1.445
8	N-C7	n-Heptane	142-82-5	1.278
9	N-C8	n-Octane	111-65-9	1.109
10	N-C9	n-Nonane	111-84-2	0.960
11	N-C10	n-Decane	124-18-5	0.832
12	N-C11	n-Undecane	1120-21-4	0.741
13	N-C12	n-Dodecane	112-40-3	0.662
14	N-C13	n-Tridecane	629-50-5	0.620
15	N-C14	n-Tetradecane	629-59-4	0.585
16	N-C15	n-Pentadecane	629-62-9	0.555
17	N-C16	n-C16	544-76-3	0.526
18	N-C17	n-C17	629-78-7	0.495
19	N-C18	n-C18	593-45-3	0.468
20	N-C19	n-C19	629-92-5	0.444
21	N-C20	n-C20	112-95-8	0.422
22	N-C21	n-C21	629-94-7	0.402
23	N-C22	n-C22	629-97-0	0.384
24	2-ME-C3	Isobutane	75-28-5	1.346
25	2-ME-C4	Iso-Pentane	78-78-4	1.675
26	22-DM-C3	Neopentane	463-82-1	0.695
27	BR-C5	Branched C5 Alkanes		1.675
28	22-DM-C4	2,2-Dimethyl Butane	75-83-2	1.334
29	23-DM-C4	2,3-Dimethyl Butane	79-29-8	1.136
30	2-ME-C5	2-Methyl Pentane	107-83-5	1.797
31	3-ME-C5	3-Methylpentane	96-14-0	2.070
32	BR-C6	Branched C6 Alkanes	96-14-0	1.535
33	223TM-C4	2,2,3-Trimethyl Butane	464-06-2	1.322
34	22-DM-C5	2,2-Dimethyl Pentane	590-35-2	1.218
35	23-DM-C5	2,3-Dimethyl Pentane	565-59-3	1.545
36	24-DM-C5	2,4-Dimethyl Pentane	108-08-7	1.646
37	2-ME-C6	2-Methyl Hexane	591-76-4	1.373
38	33-DM-C5	3,3-Dimethyl Pentane	562-49-2	1.320
39	3-ME-C6	3-Methyl Hexane	589-34-4	1.854
40	BR-C7	Branched C7 Alkanes	108-08-7	1.630
41	2233M-C4	2,2,3,3-Tetramethyl Butane	594-82-1	0.444
42	224TM-C5	2,2,4-Trimethyl Pentane	540-84-1	1.439
43	22-DM-C6	2,2-Dimethyl Hexane	590-73-8	1.132
44	234TM-C5	2,3,4-Trimethyl Pentane	565-75-3	1.226
45	23-DM-C6	2,3-Dimethyl Hexane	584-94-1	1.336
46	24-DM-C6	2,4-Dimethyl Hexane	589-43-5	1.807
47	25-DM-C6	2,5-Dimethyl Hexane	592-13-2	1.682
48	2-ME-C7	2-Methyl Heptane	592-27-8	1.200
49	3-ME-C7	3-Methyl Heptane	589-81-1	1.344
50	4-ME-C7	4-Methyl Heptane	589-53-7	1.479

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
51	BR-C8	Branched C8 Alkanes		1.573
52	225TM-C6	2,2,5-Trimethyl Hexane	3522-94-9	1.326
53	235TM-C6	2,3,5-Trimethyl Hexane	1069-53-0	1.327
54	24-DM-C7	2,4-Dimethyl Heptane	2213-23-2	1.487
55	2-ME-C8	2-Methyl Octane	3221-61-2	0.956
56	33-DE-C5	3,3-Diethyl Pentane	1067-20-5	1.346
57	35-DM-C7	3,5-Dimethyl Heptane	926-82-9	1.633
58	4-ET-C7	4-Ethyl Heptane	2216-32-2	1.441
59	4-ME-C8	4-Methyl Octane	2216-34-4	1.077
60	BR-C9	Branched C9 Alkanes		1.252
61	24-DM-C8	2,4-Dimethyl Octane	4032-94-4	1.095
62	26DM-C8	2,6-Dimethyl Octane	2051-30-1	1.269
63	2-ME-C9	2-Methyl Nonane	871-83-0	0.854
64	34-DE-C6	3,4-Diethyl Hexane		1.194
65	3-ME-C9	3-Methyl Nonane		0.888
66	4-ME-C9	4-Methyl Nonane		0.988
67	4-PR-C7	4-Propyl Heptane		1.243
68	BR-C10	Branched C10 Alkanes	2051-30-1	1.095
69	26DM-C9	2,6-Dimethyl Nonane		0.948
70	35-DE-C7	3,5-Diethyl Heptane		1.208
71	3-ME-C10	3-Methyl Decane		0.765
72	4-ME-C10	4-Methyl Decane		0.806
73	BR-C11	Branched C11 alkanes	6975-98-0	0.866
74	2346TMC7	2,3,4,6-Tetramethyl Heptane		1.259
75	36-DE-C8	2,6-Diethyl Octane		1.097
76	36DM-C10	3,6-Dimethyl Decane		0.883
77	3-ME-C11	3-Methyl Undecane		0.699
78	5-ME-C11	5-Methyl Undecane		0.717
79	BR-C12	Branched C12 Alkanes		0.796
80	2357TMC8	2,3,5,7-Tetramethyl Octane		1.061
81	36DM-C11	3,6-Dimethyl Undecane		0.818
82	37-DE-C9	3,7-Diethyl Nonane		1.074
83	3-ME-C12	3-Methyl Dodecane		0.639
84	5-ME-C12	5-Methyl Dodecane		0.643
85	BR-C13	Branched C13 Alkanes		0.729
86	2468TMC9	2,4,6,8-Tetramethyl Nonane		0.937
87	236M4IC7	2,3,6-Trimethyl 4-Isopropyl Heptane		1.241
88	37DM-C12	3,7-Dimethyl Dodecane		0.743
89	38DE-C10	3,8-Diethyl Decane		0.681
90	3-ME-C13	3-Methyl Tridecane		0.567
91	6-ME-C13	6-Methyl Tridecane		0.617
92	BR-C14	Branched C14 Alkanes		0.668
93	24568MC9	2,4,5,6,8-Pentamethyl Nonane		1.111
94	2M35IPC7	2-Methyl 3,5-Diisopropyl Heptane		0.782
95	37DM-C13	3,7-Dimethyl Tridecane		0.652
96	39DE-C11	3,9-Diethyl Undecane		0.622
97	3-ME-C14	3-Methyl Tetradecane		0.535
98	6-ME-C14	6-Methyl Tetradecane		0.571
99	BR-C15	Branched C15 Alkanes		0.602
100	268M4IC9	2,6,8-Trimethyl 4-Isopropyl Nonane		0.764

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
101	3-ME-C15	3-Methyl Pentadecane		0.505
102	48DM-C14	4,8-Dimethyl Tetradecane		0.576
103	7-ME-C15	7-Methyl Pentadecane		0.515
104	BR-C16	Branched C16 Alkanes		0.543
105	27M35IC8	2,7-Dimethyl 3,5-Diisopropyl Heptane		0.692
106	BR-C17	Branched C17 Alkanes		0.512
107	BR-C18	Branched C18 Alkanes		0.483
108	CYCC3	Cyclopropane		0.103
109	CYCC4	Cyclobutane		1.046
110	CYCC5	Cyclopentane	287-92-3	2.689
111	CYCC6	Cyclohexane	110-82-7	1.458
112	IPR-CC3	Isopropyl Cyclopropane		1.516
113	ME-CYCC5	Methylcyclopentane	96-37-7	2.423
114	CYC-C6	C6 Cycloalkanes		1.458
115	13DMCYC5	1,3-Dimethyl Cyclopentane	2532-58-3	2.150
116	CYCC7	Cycloheptane		2.256
117	ET-CYCC5	Ethyl Cyclopentane		2.272
118	ME-CYCC6	Methylcyclohexane	108-87-2	1.991
119	CYC-C7	C7 Cycloalkanes		1.991
120	13DMCYC6	1,3-Dimethyl Cyclohexane	638-04-0	1.717
121	CYCC8	Cyclooctane		1.727
122	ET-CYCC6	Ethylcyclohexane	1678-91-7	1.748
123	PR-CYCC5	Propyl Cyclopentane		1.908
124	CYC-C8	C8 Cycloalkanes		1.748
125	BCYC-C9	C9 Bicycloalkanes		1.574
126	113MCYC6	1,1,3-Trimethyl Cyclohexane		1.362
127	1E4MCYC6	1-Ethyl-4-Methyl Cyclohexane		1.622
128	C3-CYCC6	Propyl Cyclohexane		1.476
129	CYC-C9	C9 Cycloalkanes		1.549
130	BCYC-C10	C10 Bicycloalkanes	91-17-8	1.291
131	13DECYC6	1,3-Diethyl-Cyclohexane		1.337
132	14DECYC6	1,4-Diethyl-Cyclohexane		1.493
133	1M3IPCY6	1-Methyl-3-Isopropyl Cyclohexane		1.254
134	C4-CYCC6	Butyl Cyclohexane		1.076
135	CYC-C10	C10 Cycloalkanes		1.272
136	BCYC-C11	C11 Bicycloalkanes		1.007
137	13E5MCC6	1,3-Diethyl-5-Methyl Cyclohexane		1.115
138	1E2PCYC6	1-Ethyl-2-Propyl Cyclohexane		0.954
139	C5-CYCC6	Pentyl Cyclohexane		0.916
140	CYC-C11	C11 Cycloalkanes		0.994
141	CYC-C11	C11 Cycloalkanes		0.994
142	BCYC-C12	C12 Bicycloalkanes		0.879
143	CYC-C12	C12 Cycloalkanes		0.868
144	135ECYC6	1,3,5-Triethyl Cyclohexane		1.060
145	1M4C5CY6	1-Methyl-4-Pentyl Cyclohexane		0.805
146	C6-CYCC6	Hexyl Cyclohexane		0.743
147	BCYC-C13	C13 Bicycloalkanes		0.791
148	13E5PCC6	1,3-Diethyl-5-Pentyl Cyclohexane		0.989
149	1M2C6CC6	1-Methyl-2-Hexyl-Cyclohexane		0.702
150	C7-CYCC6	Heptyl Cyclohexane		0.658

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
151	CYC-C13	C13 Cycloalkanes		0.782
152	BCYC-C14	C14 Bicycloalkanes		0.713
153	13P5ECC6	1,3-Dipropyl-5-EthylCyclohexane		0.937
154	1M4C7CC6	1-Methyl-4-Heptyl Cyclohexane		0.582
155	C8-CYCC6	Octyl Cyclohexane		0.601
156	CYC-C14	C14 Cycloalkanes		0.705
157	BCYC-C15	C15 Bicycloalkanes		0.688
158	135PCYC6	1,3,5-Tripropyl Cyclohexane		0.903
159	1M2C8CC6	1-Methyl-2-Octyl Cyclohexane		0.601
160	C9-CYCC6	Nonyl Cyclohexane		0.544
161	CYC-C15	C15 Cycloalkanes		0.681
162	13P5BCC6	1,3-Propyl-5-Butyl Cyclohexane		0.772
163	1M4C9CY6	1-Methyl-4-Nonyl Cyclohexane		0.550
164	C10CYCC6	Decyl Cyclohexane		0.503
165	CYC-C16	C16 Cycloalkanes		0.607
166	ETHENE	**Ethene**	74-85-1	9.078
167	PROPENE	Propene	115-07-1	11.576
168	1-BUTENE	1-Butene	106-98-9	10.290
169	C4-OLE1	C4 Terminal Alkenes		10.290
170	1-PENTEN	1-Pentene	109-67-1	7.788
171	3M-1-BUT	3-Methyl-1-Butene	563-45-1	6.990
172	C5-OLE1	C5 Terminal Alkenes		7.788
173	1-HEXENE	1-Hexene	592-41-6	6.167
174	33M1-BUT	3,3-Dimethyl-1-Butene	558-37-2	6.065
175	3M1-C5E	3-Methyl-1-Pentene	760-20-3	6.220
176	4M1-C5E	4-Methyl-1-Pentene	691-37-2	6.262
177	C6-OLE1	C6 Terminal Alkenes		6.167
178	1-HEPTEN	1-Heptene	592-76-7	4.555
179	1-OCTENE	1-Octene	111-66-0	3.454
180	C8-OLE1	C8 Terminal Alkenes		3.454
181	1-C9E	1-Nonene	124-11-8	2.766
182	C9-OLE1	C9 Terminal Alkenes		2.766
183	1-C10E	1-Decene		2.285
184	C10-OLE1	C10 Terminal Alkenes		2.285
185	1-C11E	1-Undecene		1.952
186	C11-OLE1	C11 Terminal Alkenes		1.952
187	C12-OLE1	C12 Terminal Alkenes		1.724
188	1-C12E	1-Dodecene		1.723
189	1-C13E	1-Tridecene		1.547
190	C13-OLE1	C13 Terminal Alkenes		1.547
191	1-C14E	1-Tetradecene		1.412
192	C14-OLE1	C14 Terminal Alkenes		1.412
193	1-C15E	1-Pentadecene		1.373
194	C15-OLE1	C15 Terminal Alkenes		1.373
195	ISOBUTEN	Isobutene	115-11-7	6.355
196	2M-1-BUT	2-Methyl-1-Butene	563-46-2	6.507
197	23M1-BUT	23-Dimethyl-1-Butene		4.773
198	2E1-BUT	2-Ethyl-1-Butene		5.045
199	2M1-C5E	2-Methyl-1-Pentene	763-29-1	5.178
200	233M1BUT	2,3,3-trimethyl-1-Butene		4.614

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
201	C7-OLE1	C7 Terminal Alkenes		4.555
202	3M2I1C4E	3-Methyl-2-Isopropyl-1-Butene		3.284
203	C-2-BUTE	cis-2-Butene	590-18-1	13.226
204	T-2-BUTE	trans-2-Butene	624-64-6	13.912
205	C4-OLE2	C4 Internal Alkenes		13.568
206	2M-2-BUT	2-Methyl-2-Butene	513-35-9	14.447
207	C-2-PENT	cis-2-Pentene	627-20-3	10.236
208	T-2-PENT	trans-2-Pentene	646-04-8	10.231
209	2-C5-OLE	2-Pentenes		10.232
210	C5-OLE2	C5 Internal Alkenes	646-04-8	10.232
211	23M2-BUT	2,3-Dimethyl-2-Butene		13.324
212	2M-2-C5E	2-Methyl-2-Pentene	625-27-4	12.287
213	C-2-C6E	Cis-2-Hexene	7688-21-3	8.436
214	C-3-C6E	Cis-3-Hexene		8.215
215	C3M2-C5E	Cis-3-Methyl-2-Hexene		13.377
216	T3M2-C5E	Trans 3-Methyl-2-Hexene		14.169
217	T4M2-C5E	Trans 4-Methyl-2-Hexene		7.885
218	T-2-C6E	Trans-2-Hexene	4050-45-7	8.436
219	T-3-C6E	Trans-3-Hexene		8.161
220	2-C6-OLE	2-Hexenes	592-43-8	8.436
221	C6-OLE2	C6 Internal Alkenes		8.436
222	23M2-C5E	2,3-Dimethyl-2-Hexene		10.408
223	C-3-C7E	Cis-3-Heptene		6.960
224	T44M2C5E	Trans 4,4-dimethyl-2-Pentene		6.996
225	T-2-C7E	Trans-2-Heptene		7.333
226	T-3-C7E	Trans-3-Heptene		6.960
227	2-C7-OLE	2-Heptenes		6.960
228	C7-OLE2	C7 Internal Alkenes		6.960
229	C-4-C8E	Cis-4-Octene		5.940
230	T22M3C6E	Trans 2,2-Dimethyl 3-Hexene		5.979
231	T25M3C6E	Trans 2,5-Dimethyl 3-Hexene		5.438
232	T-3-C8E	Trans-3-Octene		6.128
233	T-4-C8E	Trans-4-Octene		5.911
234	3-C8-OLE	3-Octenes		6.128
235	C8-OLE2	C8 Internal Alkenes		5.911
236	244M2C5E	2,4,4-trimethyl-2-Pentene	107-40-4	5.842
237	3-C9-OLE	3-Nonenes		5.303
238	C9-OLE2	C9 Internal Alkenes		5.303
239	T-4-C9E	Trans-4-Nonene		5.220
240	34E2-C6E	3,4-Diethyl-2-Hexene		3.945
241	C-5-C10E	Cis-5-Decene		4.888
242	T-4-C10E	Trans-4-Decene		4.493
243	3C10-OLE	C10 3-Alkenes		4.493
244	C10-OLE2	C10 Internal Alkenes		4.493
245	T-5-C11E	Trans-5-Undecene		4.233
246	3C11-OLE	C11 3-Alkenes		4.233
247	C11-OLE2	C11 Internal Alkenes		4.233
248	2C12-OLE	C12 2-Alkenes		3.752
249	3C12-OLE	C12 3-Alkenes		3.752
250	C12-OLE2	C12 Internal Alkenes		3.752

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
251	T-5-C12E	Trans-5-Dodecene		3.752
252	T-5-C13E	Trans-5-Tridecene		3.384
253	3C13-OLE	C13 3-Alkenes		3.384
254	C13-OLE2	C13 Internal Alkenes		3.384
255	T-5-C14E	Trans-5-Tetradecene		3.076
256	3C14-OLE	C14 3-Alkenes		3.076
257	C14-OLE2	C14 Internal Alkenes		3.076
258	T-5-C15E	Trans-5-Pentadecene		2.824
259	3C15-OLE	C15 3-Alkenes		2.824
260	C15-OLE2	C15 Internal Alkenes		2.824
261	C4-OLE	C4 Alkenes		11.929
262	C5-OLE	C5 Alkenes		9.011
263	C6-OLE	C6 Alkenes		6.875
264	C7-OLE	C7 Alkenes		5.756
265	C8-OLE	C8 Alkenes		4.683
266	C9-OLE	C9 Alkenes		4.035
267	C10-OLE	C10 Alkenes		3.390
268	C11-OLE	C11 Alkenes		3.091
269	C12-OLE	C12 Alkenes		2.738
270	C13-OLE	C13 Alkenes		2.465
271	C14-OLE	C14 Alkenes		2.244
272	C15-OLE	C15 Alkenes		2.099
273	CYC-PNTE	Cyclopentene	142-29-0	7.388
274	1M-CC5E	1-Methyl cyclopentene		13.947
275	CYC-HEXE	Cyclohexene	110-83-8	5.453
276	1M-CC6E	1-Methyl Cyclohexene		7.811
277	4M-CC6E	4-Methyl Cyclohexene		4.478
278	12M-CC6E	1,2-Dimethyl Cyclohexene		6.769
279	13-BUTDE	1,3-Butadiene	106-99-0	13.583
280	ISOPRENE	Isoprene	78-79-5	10.688
281	C6-OL2D	C6 Cyclic or di-olefins	693-89-0	8.642
282	C7-OL2D	C7 Cyclic or di-olefins	591-49-1	7.486
283	C8-OL2D	C8 Cyclic or di-olefins		6.019
284	C9-OL2D	C9 Cyclic or di-olefins		5.389
285	C10-OL2D	C10 Cyclic or di-olefins		4.558
286	C11-OL2D	C11 Cyclic or di-olefins		4.289
287	C12-OL2D	C12 Cyclic or di-olefins		3.798
288	C13-OL2D	C13 Cyclic or di-olefins		3.422
289	C14-OL2D	C14 Cyclic or di-olefins		3.108
290	C15-OL2D	C15 Cyclic or di-olefins		2.852
291	CYC-PNDE	Cyclopentadiene	542-92-7	7.614
292	3-CARENE	3-Carene		3.215
293	A-PINENE	a-Pinene		4.290
294	B-PINENE	b-Pinene	127-91-3	3.284
295	D-LIMONE	d-Limonene	5989-27-5	3.990
296	SABINENE	Sabinene	3387-41-5	3.668
297	TERPENE	Terpene		3.786
298	STYRENE	Styrene	100-42-5	1.949
299	AME-STYR	a-Methyl Styrene	25013-15-4	1.717
300	C9-STYR	C9 Styrenes		1.717

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
301	C10-STYR	C10 Styrenes		1.535
302	BENZENE	Benzene	71-43-2	0.815
303	TOLUENE	Toluene	108-88-3	3.973
304	C2-BENZ	Ethyl Benzene	100-41-4	2.788
305	I-C3-BEN	Isopropyl Benzene (cumene)	104-51-8	2.322
306	N-C3-BEN	n-Propyl Benzene		2.196
307	C9-BEN1	C9 Monosubstituted Benzenes	95-47-6	2.196
308	S-C4-BEN	s-Butyl Benzene		1.967
309	C10-BEN1	C10 Monosubstituted Benzenes		1.967
310	N-C4-BEN	n-Butyl Benzene		1.967
311	C11-BEN1	C11 Monosubstituted Benzenes	108-67-8	1.781
312	C12-BEN1	C12 Monosubstituted Benzenes		1.627
313	C13-BEN1	C13 Monosubstituted Benzenes		1.497
314	M-XYLENE	m-Xylene	103-65-1	10.607
315	O-XYLENE	o-Xylene	103-65-1	7.489
316	P-XYLENE	p-Xylene	106-42-3	4.247
317	C8-BEN2	C8 Disubstituted Benzenes	1330-20-7	7.477
318	C9-BEN2	C9 Disubstituted Benzenes	25550-14-5	6.606
319	C10-BEN2	C10 Disubstituted Benzenes		5.918
320	C11-BEN2	C11 Disubstituted Benzenes		5.354
321	C12-BEN2	C12 Disubstituted Benzenes		4.897
322	C13-BEN2	C13 Disubstituted Benzenes		4.504
323	C8-BEN	Isomers of Ethylbenzene		5.164
324	123-TMB	1,2,3-Trimethyl Benzene		11.255
325	124-TMB	1,2,4-Trimethyl Benzene	95-63-6	7.179
326	135-TMB	1,3,5-Trimethyl Benzene	108-38-3	11.225
327	C9-BEN3	C9 Trisubstituted Benzenes	25551-13-7	9.898
328	C9-BEN	Isomers of Propylbenzene		6.119
329	C10-BEN4	C10 Tetrasubstituted Benzenes		8.863
330	C10-BEN3	C10 Trisubstituted Benzenes	576-73-8	8.863
331	C10-BEN	Isomers of Butylbenzene		5.478
332	C11-BEN5	C11 Pentasubstituted Benzenes		8.026
333	C11-BEN4	C11 Tetrasubstituted Benzenes		8.026
334	C11-BEN3	C11 Trisubstituted Benzenes		8.026
335	C11-BEN	Isomers of Pentylbenzene		4.957
336	C12-BEN5	C11 Pentasubstituted Benzenes		7.335
337	C12-BEN6	C12 Hexaasubstituted Benzenes		7.335
338	C12-BEN4	C12 Tetrasubstituted Benzenes		7.335
339	C12-BEN3	C12 Trisubstituted Benzenes		7.335
340	C12-BEN	Isomers of Hexylbenzene		4.531
341	C13-BEN3	C13 Trisubstituted Benzenes		6.749
342	INDAN	Indan	496-11-7	3.165
343	NAPHTHAL	Naphthalene	91-20-3	3.262
344	TETRALIN	Tetralin		2.830
345	ME-NAPH	Methyl Naphthalenes		4.613
346	1ME-NAPH	1-Methyl Naphthalene		4.613
347	2ME-NAPH	2-Methyl Naphthalene		4.613
348	C11-TET	C11 Tetralin or Indane		2.558
349	23-DMN	2,3-Dimethyl Naphthalene		5.543
350	C12-NAP2	C12 Disubstituted Naphthalenes		5.543

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
351	DM-NAPH	Dimethyl Naphthalenes		5.543
352	C12-NAP1	C12 Monosubstituted Naphthalene		4.200
353	C13-NAP2	C13 Disubstituted Naphthalenes		5.080
354	C13-NAP3	C13 Trisubstituted Naphthalenes		5.080
355	C13-NAP1	C13 Monosubstituted Naphthalene		3.856
356	ACETYLEN	Acetylene	74-86-2	1.247
357	ME-ACTYL	Methyl Acetylene	74-99-7	6.447
358	2-BUTYNE	2-Butyne	503-17-3	16.328
359	ET-ACTYL	Ethyl Acetylene	107-00-6	6.195
360	MEOH	Methanol	67-56-1	0.712
361	ETOH	Ethanol	64-17-5	1.691
362	I-C3-OH	Isopropyl Alcohol	67-63-0	0.714
363	N-C3-OH	n-Propyl Alcohol	71-23-8	2.739
364	I-C4-OH	Isobutyl Alcohol	78-83-1	2.242
365	N-C4-OH	n-Butyl Alcohol	71-36-3	3.337
366	S-C4-OH	s-Butyl Alcohol	78-92-2	1.594
367	T-C4-OH	t-Butyl Alcohol	75-65-0	0.450
368	CC5-OH	Cyclopentanol	96-41-3	1.956
369	2-C5OH	2-Pentanol		1.739
370	3-C5OH	3-Pentanol		1.735
371	C5OH	Pentyl Alcohol	71-41-0	3.353
372	CC6-OH	Cyclohexanol	108-93-0	2.248
373	1-C6OH	1-Hexanol		2.740
374	2-C6OH	2-Hexanol		2.468
375	1-C7OH	1-Heptanol		2.207
376	1-C8-OH	1-Octanol	111-87-5	2.011
377	2-ETC6OH	2-Ethyl-1-Hexanol	104-76-7	2.201
378	2-C8-OH	2-Octanol	4128-31-8	2.162
379	3-C8-OH	3-Octanol	20296-29-1	2.563
380	4-C8-OH	4-Octanol		3.067
381	I-C10-OH	8-Methyl-1-Nonanol (Isodecyl Alcohol)	25339-17-7	1.232
382	ET-GLYCL	Ethylene Glycol	107-21-1	3.362
383	PR-GLYCL	Propylene Glycol	57-55-6	2.752
384	GLYCERL	Glycerol	56-81-5	3.272
385	12-C4OH2	1,2-Butandiol		2.212
386	C6-GLYCL	1,2-Dihydroxy Hexane	107-41-5	2.749
387	2M24C5OH	2-Methyl-2,4-Pentanediol		1.042
388	ME-O-ME	Dimethyl Ether	115-10-6	0.929
389	TME-OX	Trimethylene Oxide		5.221
390	METHYLAL	Dimethoxy methane	109-87-5	1.040
391	THF	Tetrahydrofuran		4.946
392	ET-O-ET	Diethyl Ether	60-29-7	4.016
393	AM-THF	Alpha-Methyltetrahydrofuran		4.621
394	THP	Tetrahydropyran		3.809
395	ET-O-IPR	Ethyl Isopropyl Ether	625-54-7	3.861
396	MNBE	Methyl n-Butyl Ether		3.651
397	MTBE	Methyl t-Butyl Ether	628-28-4	0.777
398	22MEOC3	2,2-Dimethoxy Propane	77-76-9	0.518
399	PR-O-PR	Di n-Propyl Ether		3.244
400	ENBE	Ethyl n-Butyl Ether		3.862

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
401	ETBE	Ethyl t-Butyl Ether	637-92-3	2.114
402	MTAE	Methyl t-Amyl Ether		2.141
403	2BU-THF	2-Butyl Tetrahydrofuran		2.529
404	IBU2-O	Di-Isobutyl Ether		1.295
405	BU-O-BU	Di-n-butyl Ether	142-96-1	3.172
406	C5-O-C5	Di-n-Pentyl Ether		2.635
407	MEO-ETOH	2-Methoxyethanol	109-86-4	2.981
408	MEOC3OH	1-Methoxy-2-Propanol	107-98-2	2.615
409	ETO-ETOH	2-Ethoxyethanol	110-80-5	3.781
410	2MEOC3OH	2-Methoxy-1-Propanol		3.006
411	DET-GLCL	Diethylene Glycol	111-46-6	3.548
412	ETOC3OH	1-Ethoxy-2-Propanol	1569-02-4	3.249
413	2PROETOH	2-Propoxyethanol		3.513
414	3ETOC3OH	3-Ethoxy-1-Propanol		4.241
415	3MEOC4OH	3-Methoxy-1-Butanol		0.967
416	MOEOETOH	2-(2-Methoxyethoxy) Ethanol	111-77-3	2.897
417	PROXC3OH	1-Propoxy-2-Propanol		2.861
418	BUO-ETOH	2-Butoxyethanol	111-76-2	2.903
419	3MOMC4OH	3 methoxy -3 methyl-Butanol		1.742
420	CARBITOL	2-(2-Ethoxyethoxy) EtOH	111-90-0	3.189
421	DPR-GLCL	Dipropylene Glycol		2.483
422	PG-1TB-E	1-tert-Butoxy-2-Propanol		1.714
423	PG-2TB-E	2-tert-Butoxy-1-Propanol		1.812
424	BUOC3OH	n-Butoxy-2-Propanol		2.696
425	DGPE	2-(2-Propoxyethoxy) ethanol	6881-94-3	2.998
426	DPRGOME	Dipropylene Glycol Methyl Ether (structure inferred)		2.202
427	DPGOME2	Dipropylene Glycol Methyl Ether [(2-methoxymethylethoxy) propanol]	34590-94-8	3.020
428	TGME	2-[2-(2-Methoxyethoxy) ethoxy] ethanol	112-35-6	2.613
429	EGHE	2-Hexyloxyethanol	112-25-4	2.452
430	TMPDG	2,2,4-Trimethyl-1,3-Pentanediol	144-19-4	1.738
431	C8-CELSV	2-(2-Butoxyethoxy)-EtOH	112-34-5	2.702
432	TGEE	2-[2-(2-Ethoxyethoxy) ethoxy] ethanol	112-50-5	2.661
433	TGPE	2-[2-(2-Propoxyethoxy) ethoxy] ethanol		2.461
434	TETRAGME	2,5,8,11-Tetraoxatridecan-13-ol	23783-42-8	2.146
435	EGEHE	2-(2-Ethylhexyloxy) ethanol	1559-35-9	1.710
436	DGHE	2-(2-Hexyloxyethoxy) ethanol	112-59-4	2.033
437	TGBE	2-[2-(2-Butoxyethoxy) ethoxy] ethanol	143-22-6	2.238
438	TPRGOME	Tripropylene Glycol Monomethyl Ether		1.896
439	TETRAGBE	3,6,9,12-Tetraoxahexadecan-1-ol	1559-34-8	1.891
440	ME-FORM	Methyl Formate	107-31-3	0.064
441	ET-FORM	Ethyl Formate		0.520
442	ME-ACET	Methyl Acetate	79-20-9	0.073
443	ET-ACET	Ethyl Acetate	141-78-6	0.638
444	ME-PRAT	Methyl Propionate	554-12-1	0.706
445	C3-FORM	n-Propyl Formate		0.925
446	ET-PRAT	Ethyl Propionate	105-37-3	0.794
447	IPR-ACET	Isopropyl Acetate	108-21-4	1.121
448	ME-BUAT	Methyl Butyrate		1.162
449	ME-IBUAT	Methyl Isobutyrate		0.694
450	C4-FORM	n-Butyl Formate	592-84-7	0.949

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
451	PR-ACET	Propyl Acetate	109-60-4	0.867
452	ET-BUAT	Ethyl Butyrate		1.252
453	IBU-ACET	Isobutyl Acetate	110-19-0	0.670
454	ME-PVAT	Methyl Pivalate	598-98-1	0.394
455	BU-ACET	n-Butyl Acetate	138-22-7	0.888
456	PR-PRAT	n-Propyl Propionate		0.925
457	SBU-ACET	s-Butyl Acetate		1.434
458	TBU-ACET	t-Butyl Acetate	540-88-5	0.204
459	BU-PRAT	Butyl Propionate		0.883
460	AM-ACET	Amyl Acetate		0.948
461	PR-BUAT	n-Propyl Butyrate		1.163
462	E3EOC3OH	Ethyl 3-Ethoxy Propionate		3.610
463	23MC4ACT	2,3-Dimethylbutyl Acetate		0.838
464	2MC5-ACT	2-Methylpentyl Acetate		1.109
465	3MC5-ACT	3-Methylpentyl Acetate		1.311
466	4MC5-ACT	4-Methylpentyl Acetate		0.915
467	IBU-IBTR	Isobutyl Isobutyrate	297-85-8	0.614
468	BU-BUAT	n-Butyl Butyrate		1.120
469	NC6-ACET	n-Hexyl Acetate		0.873
470	24MC5ACT	2,4-Dimethylpentyl Acetate		0.980
471	2MC6-ACT	2-Methylhexyl Acetate		0.892
472	3EC5-ACT	3-Ethylpentyl Acetate		1.237
473	3MC6-ACT	3-Methylhexyl Acetate		1.009
474	4MC6-ACT	4-Methylhexyl Acetate		0.906
475	5MC6-ACT	5-Methylhexyl Acetate		0.795
476	IC5IBUAT	Isoamyl Isobutyrate		0.887
477	NC7-ACET	n-Heptyl Acetate		0.727
478	24MC6ACT	2,4-Dimethylhexyl Acetate		0.927
479	2ETHXACT	2-Ethyl-Hexyl Acetate		0.784
480	34MC6ACT	3,4-Dimethylhexyl Acetate		1.161
481	35MC6ACT	3,5-Dimethylhexyl Acetate		1.087
482	3EC6-ACT	3-Ethylhexyl Acetate		1.028
483	3MC7-ACT	3-Methylheptyl Aceate		0.755
484	45MC6ACT	4,5-Dimethylhexyl Acetate		0.856
485	4MC7-ACT	4-Methylheptyl Acetate		0.725
486	5MC7-ACT	5-Methylheptyl Aceate		0.730
487	NC8-ACET	n-Octyl Acetate		0.642
488	235M6ACT	2,3,5-Teimethylhexyl Acetate		0.865
489	23MC7ACT	2,3-Dimethylheptyl Acetate		0.846
490	24MC7ACT	2,4-Dimethylheptyl Acetate		0.875
491	25MC7ACT	2,5-Dimethylheptyl Acetate		0.864
492	2MC8-ACT	2-Methyloctyl Acetate		0.627
493	35MC7ACT	3,5-Dimethylheptyl Acetate		1.013
494	36MC7ACT	3,6-Dimethylheptyl Acetate		0.874
495	3EC7-ACT	3-Ethylheptyl Acetate		0.704
496	45MC7ACT	4,5-Dimethylheptyl Acetate		0.959
497	46MC7ACT	4,6-Dimethylheptyl Acetate		0.829
498	4MC8-ACT	4-Methyloctyl Acetate		0.681
499	5MC8-ACT	5-Methyloctyl Acetate		0.672
500	NC9-ACET	n-Nonyl Acetate		0.583

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
501	36MC8ACT	3,6-Dimethyloctyl Acetate		0.877
502	3IPC7ACT	3-Isopropylheptyl Acetate		0.715
503	46MC8ACT	4,6-Dimethyloctyl Acetate		0.848
504	357M8ACT	3,5,7-Trimethyloctyl Acetate		0.826
505	3E6M8ACT	3-Ethyl-6-Methyloctyl Acetate		0.801
506	47MC9ACT	4,7-Dimethylnonyl Acetate		0.636
507	2357M8AC	2,3,5,7-Tetramethyloctyl Acetate		0.737
508	357M9ACT	3,5,7-Trimethylnonyl Acetate		0.761
509	368M9ACT	3,6,8-Trimethylnonyl Acetate		0.718
510	2468M8AC	2,4,6,8-Tetramethylnonyl Acetate		0.628
511	3E67M9AC	3-Ethyl-6,7-Dimethylnonyl Acetate		0.765
512	479M10AC	4,7,9-Trimethyldecyl Acetate		0.549
513	23568M9A	2,3,5,6,8-Pentaamethylnonyl Acetate		0.741
514	3579M10A	3,5,7,9-Tetramethyldecyl Acetate		0.585
515	5E368M9A	5-Ethyl-3,6,8-Trimethylnonyl Acetate		0.769
516	DMC	Dimethyl Carbonate		0.059
517	PC	Propylene Carbonate	108-32-7	0.251
518	ME-LACT	Methyl Lactate		2.746
519	MCSVACET	2-Methoxyethyl Acetate	110-49-6	1.187
520	ET-LACT	Ethyl Lactate		2.716
521	MIPR-CB	Methyl Isopropyl Carbonate		0.690
522	PGME-ACT	1-Methoxy-2-Propyl Acetate	108-65-6	1.701
523	CSV-ACET	2-Ethoxyethyl Acetate	111-15-9	1.893
524	2PGMEACT	2-Methyoxy-1-propyl Acetate		1.125
525	DBE-4	Dimethyl Succinate	106-65-0	0.233
526	ETGLDACT	Ethylene Glycol Diacetate	111-55-7	0.729
527	DIPR-CB	Diisopropyl Carbonate		1.042
528	DBE-5	Dimethyl Glutarate	1119-40-0	0.508
529	2BUETACT	2-Butoxyethyl Acetate		1.666
530	DBE-6	Dimethyl Adipate	627-93-0	1.959
531	DGEEA	2-(2-Ethoxyethoxy) ethyl acetate	112-15-2	1.507
532	DGBEA	2-(2-Butoxyethoxy) ethyl acetate	124-17-4	1.377
533	SC7ESC12	Substituted C7 ester (C12)		0.916
534	TEXANOL2	1-Hydroxy-2,2,4-Trimethylpentyl-3-Isobutyrate	18491-15-1	0.923
535	TEXANOL1	3-Hydroxy-2,2,4-Trimethylpentyl-1-Isobutyrate	77-68-9	0.880
536	TEXANOL	Texanol isomers	25265-77-4	0.894
537	SC9ESC12	Substituted C9 Ester (C12)		0.886
538	DBE-10	Dimethyl Sebacate	106-79-6	0.478
539	ETOX	Ethylene Oxide	75-21-8	0.044
540	PROX	Propylene Oxide	75-56-9	0.317
541	12BUOX	1,2-Epoxybutane		1.012
542	FORMACID	Formic Acid	64-18-6	0.076
543	ACETACID	Acetic Acid	64-19-7	0.707
544	GLYACD	Glycolic Acid	79-14-1	2.674
545	PAA	Peroxyacetic Acid	79-21-2	
546	ACYRACID	Acrylic Acid	79-10-7	11.665
547	PROPACID	Propionic Acid	79-09-4	1.159
548	MACRACD	Methacrylic Acid	79-41-4	18.782
549	2ETHXACD	2-Ethyl Hexanoic Acid	149-57-5	4.408
550	ME-ACRYL	Methyl Acrylate	96-33-3	12.234

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
551	VIN-ACET	Vinyl Acetate	108-05-4	3.280
552	MBUTENOL	2-Methyl-2-Butene-3-ol	34454-78-9	5.124
553	ET-ACRYL	Ethyl Acrylate	140-88-5	8.815
554	ME-MACRT	Methyl Methacrylate	80-62-6	15.838
555	BU-MACRT	Butyl Methacrylate		9.079
556	IBUMACRT	Isobutyl Methacrylate		8.988
557	2ETHXACR	2-Ethyl-Hexyl Acrylate	103-11-7	2.422
558	FURAN	Furan		16.539
559	FORMALD	Formaldehyde	50-00-0	8.969
560	ACETALD	Acetaldehyde	75-07-0	6.836
561	PROPALD	Propionaldehyde	123-38-6	7.884
562	2MEC3AL	2-Methylpropanal	78-84-2	5.870
563	1C4RCHO	Butanal		6.731
564	C4-RCHO	C4 aldehydes	123-72-8	6.731
565	22DMC3AL	2,2-Dimethylpropanal (pivaldehyde)		5.399
566	3MC4RCHO	3-Methylbutanal (Isovaleraldehyde)	590-86-3	5.521
567	1C5RCHO	Pentanal (Valeraldehyde)	110-62-3	5.764
568	C5-RCHO	C5 Aldehydes		5.763
569	GLTRALD	Glutaraldehyde		4.794
570	1C6RCHO	Hexanal		4.978
571	C6-RCHO	C6 Aldehydes		4.978
572	1C7RCHO	Heptanal		4.231
573	C7-RCHO	C7 Aldehydes		4.231
574	1C8RCHO	Octanal		3.650
575	C8-RCHO	C8 Aldehydes		3.650
576	GLYOXAL	Glyoxal	107-22-2	14.224
577	MEGLYOX	Methyl Glyoxal	78-98-8	16.207
578	ACROLEIN	Acrolein	107-02-8	7.607
579	CROTALD	Crotonaldehyde	123-73-9	10.065
580	METHACRO	Methacrolein		6.231
581	HOMACR	Hydroxy Methacrolein		6.608
582	BENZALD	Benzaldehyde	100-52-7	-0.609
583	TOLUALD	Tolualdehyde		-0.538
584	ACETONE	Acetone	67-64-1	0.427
585	CC4-KET	Cyclobutanone		0.683
586	MEK	Methyl Ethyl Ketone	78-93-3	1.477
587	CC5-KET	Cyclopentanone		1.427
588	KET5C	C5 Cyclic Ketones		1.427
589	MPK	2-Pentanone		3.065
590	DEK	3-Pentanone		1.445
591	KET5	C5 Ketones		3.065
592	CC6-KET	Cyclohexanone		1.608
593	KET6C	C6 Cyclic Ketones		1.608
594	MIBK	4-Methyl-2-Pentanone	108-10-1	4.306
595	MNBK	Methyl n-Butyl Ketone		3.548
596	MTBK	Methyl t-Butyl Ketone		0.785
597	KET6	C6 Ketones		3.548
598	KET7C	C7 Cyclic Ketones		1.407
599	C7-KET-2	2-Heptanone	110-43-0	2.799
600	2M-3-HXO	2-Methyl-3-Hexanone		1.787

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	**Name**	**Description**	**CAS No.**	**MIR** (g O_3/g VOC)
601	DIPK	Di-Isopropyl Ketone		1.629
602	KET7	C7 Ketones		2.799
603	5M2HXO	5-Methyl-2-Hexanone	110-12-3	2.101
604	3M2HXO	3-Methyl-2-Hexanone	2550-21-2	2.812
605	KET8C	C8 Cyclic Ketones		1.250
606	C8-KET-2	2-Octanone		1.659
607	KET8	C8 Ketones		1.659
608	KET9C	C9 Cyclic Ketones		1.125
609	C9-KET-2	2-Nonanone		1.305
610	DIBK	Di-isobutyl ketone (2,6-dimethyl-4-heptanone)		2.935
611	KET9	C9 Ketones		1.305
612	KET10C	C10 Cyclic Ketones		1.023
613	C10-K-2	2-Decanone		1.058
614	KET10	C10 Ketones		1.058
615	BIACETYL	Biacetyl		20.729
616	MVK	Methylvinyl ketone		8.727
617	HOACET	Hydroxy Acetone		3.083
618	MEOACET	Methoxy Acetone		2.135
619	DIACTALC	Diacetone Alcohol	123-42-2	0.683
620	PHENOL	Phenol	108-95-2	1.824
621	CRESOL	Alkyl Phenols	1319-77-3	2.343
622	M-CRESOL	m-Cresol		2.343
623	P-CRESOL	p-Cresol		2.343
624	O-CRESOL	o-Cresol	8001-58-9	2.343
625	PGPHE	1-phenoxy-2-propanol	770-35-4	1.735
626	NO2-BENZ	Nitrobenzene	98-95-3	0.067
627	P-TI	Para Toluene Isocyanate		0.933
628	TDI	Toluene Diisocyanate	26471-62-5	-0.132
629	MDI	Methylene Diphenylene Diisocyanate		0.794
630	DM-AMINE	Dimethyl Amine		9.377
631	ET-AMINE	Ethyl Amine		7.796
632	TM-AMINE	Trimethyl Amine	75-50-3	7.065
633	ME-NITRT	Methyl Nitrite		
634	ETOH-NH2	Ethanolamine	141-43-5	5.969
635	DMAE	Dimethylaminoethanol		4.758
636	ETOH2-NH	Diethanol Amine		4.051
637	ETOH3-N	Triethanolamine		2.757
638	ACRYLNIT	Acrylonitrile	107-13-1	
639	NMP	N-Methyl-2-Pyrrolidone		2.557
640	CH3-CL	Methyl Chloride	74-87-3	0.035
641	CL2-ME	Dichloromethane	75-09-2	0.066
642	ME-BR	Methyl Bromide		0.017
643	CHCL3	Chloroform	67-66-3	0.034
644	CCL4	Carbon Tetrachloride	56-23-5	
645	ME-BR2	Methylene Bromide	74-95-3	
646	CL-ETHE	Vinyl Chloride	75-01-4	2.921
647	C2-CL	Ethyl Chloride	75-00-3	0.246
648	11CL2-C2	1,1-Dichloroethane		0.101
649	12CL2-C2	1,2-Dichloroethane		0.098
650	C2-BR	Ethyl Bromide		0.108

Table C.5(a) Maximum Incremental Reactivities for VOCs for Ozone Formation Impact (Continued)

Seq.	Name	Description	CAS No.	MIR (g O_3/g VOC)
651	111-TCE	1,1,1-Trichloroethane		0.004
652	112CL3C2	1,1,2-Trichloroethane		0.058
653	11BR2-C2	1,2-Dibromoethane		0.046
654	12CL2-C3	1,2-Dichloropropane		
655	C3-BR	n-Propyl Bromide		0.348
656	C4-CL	1-Chlorobutane		
657	C4-BR	n-Butyl Bromide		0.602
658	3CLME-C8	3-(Chloromethyl)-Heptane	123-04-6	
659	11CL2ETH	1,1-Dichloroethene		
660	T-12-DCE	Trans-1,2-Dichloroethene		0.810
661	CL3-ETHE	Trichloroethylene	79-01-6	0.601
662	CL4-ETHE	Perchloroethylene	127-18-4	0.040
663	CL2IBUTE	2-(Cl-methyl)-3-Cl-Propene		1.129
664	CL-BEN	Monochlorobenzene	108-90-7	0.362
665	CL2-BEN	p-Dichlorobenzene		0.202
666	CF3-BEN	Benzotrifluoride		0.264
667	PCBTF	p-Trifluoromethyl-Cl-Benzene		0.113
668	CCL3NO2	Chloropicerin		
669	DMS	Dimethyl Sulfide		
670	DMSO	Dimethyl Sulfoxide	67-68-5	6.895
671	SI2OME6	Hexamethyldisiloxane		
672	SI2OMEOH	Hydroxymethyldisiloxane		
673	(SIOME)4	D4 Cyclosiloxane		
674	(SIOME)5	D5 Cyclosiloxane		
675	INERT	Unreactive VOCs		
676	Mixtures			
677	ARBROG	Base ROG Mixture		3.708
678	RFA-TLEV	TLEV Exhaust -- RFA		4.094
679	PH2-TLEV	TLEV Exhaust -- Phase 2		4.051
680	LPG-TLEV	TLEV Exhaust -- LPG		2.105
681	CNG-TLEV	TLEV Exhaust -- CNG		0.749
682	E85-TLEV	TLEV Exhaust -- E-85		2.703
683	M85-TLEV	TLEV Exhaust -- M-85		1.572
684	RFA-LEV	Final LEV -- RFA		3.639
685	PH2-LEV	Final LEV -- Phase 2		3.554
686	MS-D	Mineral Spirits "D" (Type II-C)		0.788
687	MS-A	Mineral Spirits "A" (Type I-B, 91% Alkanes)		1.273
688	MS-B	Mineral Spirits "B" (Type II-C)		0.778
689	D95	Exxon Exxol(r) D95 Fluid	90438-79-2	0.667
690	MS-C	Mineral Spirits "C" (Type II-C)	88230-35-7	0.782
691	ISOPARM	Exxon Isopar(r) M Fluid	108419-32-5	0.654
692	OC13ACET	Oxo-Tridecyl Acetate		0.674
693	OC12ACET	Oxo-Dodecyl Acetate		0.721
694	OC10ACET	Oxo-Decyl Acetate	770-35-4	0.828
695	OC9-ACET	Oxo-Nonyl Acetate	108419-35-8	0.849
696	OC8-ACET	Oxo-Octyl Acetate		0.957
697	OC7-ACET	Oxo-Heptyl Acetate		0.972
698	OC6-ACET	Oxo-Hexyl Acetate	108419-33-6	1.025

Table C.5(b) POCP for VOCs for Ozone Formation Impact

(Source: Derwent et al.1998)

Seq	Organic compound	POCP	Seq	Organic compound	POCP	Seq	Organic compound	POCP
	Alkanes		42	Isoprene	109.2	82	s-Butanol	40.0
1	Methane	0.6		*Alkynes*		83	t-Butanol	12.3
2	Ethane	12.3	43	Acetylene	8.5	84	3-Pentanol	42.2
3	Propane	17.6		*Aromatics*		85	2-Methylbutan-1-ol	40.7
4	n-Butane	35.2	44	Benzene	21.8	86	3-Methylbutan-1-ol	41.2
5	i-Butane	30.7	45	Toluene	63.7	87	3-Methylbutan-2-ol	36.6
6	n-Pentane	39.5	46	o-Xylene	105.3	88	2-Methylbutan-2-ol	14.2
7	i-Pentane	40.5	47	m-Xylene	110.8	89	Diacetone alcohol	26.2
8	Neopentane	17.3	48	p-Xylene	101.0		*Glycols*	
9	n-Hexane	48.2	49	Ethylbenzene	73.0	90	Ethylene glycol	38.2
10	2-Methylpentane	42.0	50	Propylbenzene	63.6	91	Propylene glycol	45.7
11	3-Methylpentane	47.9	51	i-Propylbenzene	50.0		*Ethers*	
12	2,2-Dimethylbutane	24.1	52	1,2,3-Trimethylbenzene	126.7	92	Dimethylether	17.4
13	2-3-Dimethylbutane	54.1	53	1,2,4-Trimethylbenzene	127.8	93	Methyl-t-butylether	15.2
14	n-Heptane	49.4	54	1,3,5-Trimethylbenzene	138.1	94	Diethylether	46.7
15	2-Methylhexane	41.1	55	o-Ethyltoluene	89 8	95	Diisopropyleter	47.6
16	3-Methylhexane	36.4	56	m-Ethyltoluene	101.9	96	Ethyl-t-butylether	21.4
17	n-Octane	45.3	57	p-Ethyltoluene	90.6		*Alchol and glycol ethers*	
18	n-Nonane	41.4	58	3,5-Dimethylethylbenzene	132.0	97	2-Methoxyethanol	30.0
19	n-Decane	38.4	59	3,5-Diethyltoluene	129.5	98	2-Ethoxyethanol	38.7
20	n-Undecane	38.4		*Aldehydes*		99	1-Butoxypropanol	43.6
21	n-Dodecane	35.7	60	Formaldehyde	51.9	100	2-Butoxyethanol	43.8
	Cycloalkanes		61	Acetaldehyde	64.1	101	1-Methoxy-2-Propanol	36.8
22	Cyclohexane	29.0	62	Propionaldehyde	79.8		*Carboxylic acids*	
23	Cyclohexanone	29.9	63	Butyraldehyde	79.5	102	Formic acid	3.2
24	Cyclohexanol	44.6	64	i-Butyraldehyde	51.4	103	Acetic acid	9.7
	Alkenes		65	Pentanaldehyde	76.5	104	Propanoic acid	15.0
25	Ethylene	100.0	66	Benzaldehyde	-9.2		*Esters*	
26	Propylene	112.3		*Ketones*		105	Methyl formate	3.3
27	But-1-ene	107.9	67	Acetone	9.4	106	Methyl acetate	4.6
28	cis-But-2-ene	114.6	68	Methylethylketone	37.3	107	Ethyl acetate	21.3
29	trans-But-2-ene	113.2	69	Methyl-i-butylketone	49.0	108	i-Propyl acetate	21.3
30	Methylpropene	62.7	70	Methylpropylketone	54.8	109	n-propyl acetate	29.0
31	cis-Pent-2-ene	112.1	71	Diethylketone	41.4	110	n-Butyl acetate	24.1
32	trans-Pent-2-ene	111.7	72	Methyl-i-propylketone	36.4	111	s-butyl acetate	26.7
33	Pent-1-ene	97.7	73	Hexan-2-one	57.2	112	t-butyl acetate	6.5
34	2-Methylbut-1-ene	77.1	74	Hexan-3-one	59.9		*Halocarbons*	
35	3-Methylbut-1-ene	67.1	75	Methyl-t-butylketone	32.3	113	Methyl chloride	0.5
36	2-Methylbut-2-ene	84.2		*Alcohols*		114	Methylene chloride	6.8
37	Hex-1-ene	87.4	76	Methanol	13.1	115	Chloroform	2.3
38	cis-Hex-2-ene	106.9	77	Ethanol	38.6	116	cis-Dichloroethylene	44.7
39	trans-Hex-2-ene	107.3	78	n-Propanol	54.3	117	trans-Dichloroethylene	39.2
40	Styrene	14.2	79	n-Butanol	61.2	118	Tetrachloroethylene	2.9
	Dialkenes		80	i-Propanol	14.0	119	Trichloroethylene	32.5
41	1,3-Butadiene	85.1	81	i-Butanol	37.5	120	Methyl chloroform	0.9

Table C.5(c) Site Factors for Characterization of VOC and NO_X for Photochemical Ozone Formation Impact
(Source: EPA 2003a)

State	SF_{Smog}	State	SF_{Smog}	State	SF_{Smog}
AL	0.894	MD	0.767	SC	0.799
AK	1.000(1)	MA	0.409	SD	1.305
AZ	1.525	MI	0.701	TN	0.976
AR	1.128	MN	0.856	TX	1.201
CA	1.768	MS	1.150	UT	1.619
CO	1.521	MO	0.886	VT	0.705
CT	0.483	MT	1.410	VA	0.829
DE	0.253	NE	1.449	WA	1.528
DC	0.925	NV	1.520	WV	0.561
FL	0.564	NH	0.592	WI	0.845
GA	0.812	NJ	0.595	WY	1.443
HI	1.000(1)	NM	1.496	Regional	
ID	1.921	NY	0.649	NE	0.585
IL	0.894	NC	0.721	MW	0.887
IN	0.897	ND	0.754	S	0.926
IA	1.005	OH	0.739	W	1.650
KS	1.116	OK	1.240	East of Mississippi	0.753
KY	0.852	OR	1.661	West of Mississippi	1.326
LA	1.005	PA	0.659		
ME	0.501	RI	0.274	U.S. Average	1.000

(1) Use U.S. average

APPENDIX D

REPORTED TMDLs AND LIST OF ONLINE REFERENCES

This appendix summarizes the reported total maximum daily loads (TMDLs) for impaired waters in U.S. states. And also presented in this appendix is the list of the websites that post those TMDLs.

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
AK							
	N/A						
AL							
	Lake Weiss	P	13657 sq.km.	144,000	175,000		kg/yr
AR							
	Hicks Creek	NO_3^-	15 sq.mi.	417			lb/yr
	Holman Creek	NO_3^-	27 sq.mi.	167			lb/yr
	Whig Creek	NO_3^-	8 sq.mi.	542			lb/yr
AZ							
	N/A						
CA							
	Newport Bay	N	154 sq.mi.	422,401	328,040		lb/yr
	Newport Bay	P	154 sq.mi.	22,076	64,836		lb/yr
CO							
	N/A						
CT							
	Long Island Sound	N	16000 sq.mi.	17,799,000	54,440,000		kg/yr
DC							
	N/A						
DE							
	Inland Bays	N	240.11 sq.mi.	1,349	1,393		lb/yr
	Inland Bays	P	240.11 sq.mi.		78		lb/yr
	Murderkill River	N	106 sq.mi.	406	560		lb/yr
	Murderkill River	P	106 sq.mi.	27	96		lb/yr
	Nanticoke	N	397 sq.mi.	668	1,055		kg/d
	Nanticoke	P	397 sq.mi.	26	97		kg/d
FL							
	N/A						
GA							
	N/A						
HI							
	N/A						
IA							
	Silver Lake	P	187 ac.		60		lb/yr
	Rock Creek Lake	P	26719 ac.		10,400		kg/yr
	Lake Miami	P	3595 ac.		1,400		kg/yr

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions (Continued)

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
ID							
	Cottonwood Creek	N	124439 ac.		11,309		lb/yr
	Cottonwood Creek	P	124439 ac.		3,770		lb/yr
	Jim Ford Creek	N	65838 ac.		9,932		lb/yr
	Jim Ford Creek	P	65838 ac.		3,681		lb/yr
	Snake River and Hells Canyon	P	73000 sq.mi.	2,495	916		kg/d
	Winchester Lake	P	7411 ac.		757		lb/yr
IL							
	Governor Bond Lake	P	22520 ac.		1,856	57	kg/yr
IN							
	Kokomo Lake	P	23402 ac.		6		kg/d
	Kokomo Lake	NH_3	23402 ac.	1	0		kg/d
KS							
	Russell Lake	P	3.8 sq.mi.		181		lb/yr
	Lake Meade	P	91.3 sq.mi.	1	35	4	lb/yr
	Hamilton Lake	P	15.8 sq.mi.		253	25	lb/yr
	Ford County Lake	N	13.4 sq.mi.		4,067	452	lb/yr
	Ford County Lake	P	13.4 sq.mi.		230	26	lb/yr
	Stone Lake	P	3.6 sq.mi.		25	3	lb/yr
	Blue River	P	64.8 sq.mi.	1	3		lb/yr
	Pony Creek Lake	N	6.56 sq.mi.	13,098		1,455	lb/yr
	Pony Creek Lake	P	6.56 sq.mi.	3,018		335	lb/yr
	Lake Jewell	P	15.1 sq.mi.		882	98	lb/yr
	Mission Lake	P	8.1 sq.mi.		736	80	lb/yr
	Little Lake	P	8.8 sq.mi.		479	53	lb/yr
	Clinton Lake	P	367 sq.mi.		70,000	10,000	kg/yr
	Tuttle Creek Lake	P	9628 sq.mi.		860,000	100,000	kg/yr
	Pomona Lake	P	319 sq.mi.	1,385	264,983	29,597	lb/yr
	Hillsdale Lake	P	142.2 sq.mi.	10,148	64,244	8,266	lb/yr
	Quivira Big Salt Marsh	P	101.9 sq.mi.		1,179	131	lb/yr
	Quivira Little Salt Marsh	P	862 sq.mi.	539	23,717	2,695	lb/yr
	Mingenback Lake	N	18.3 sq.mi.		3,721	413	lb/yr
	Mingenback Lake	P	18.3 sq.mi.		5,159	573	lb/yr
	Pracht Weltand	N	2823 ac.		1,240	138	lb/yr
	Pracht Weltand	P	2823 ac.		476	53	lb/yr
	Cheny Lake	P	880.6 sq.mi.	2,352	103,501	11,762	lb/yr
	Hargis Lake	N	31.3 sq.mi.		7,213	801	lb/yr
	Hargis Lake	P	31.3 sq.mi.		750	33	lb/yr
KY							
	Baughman Fork	P	9 sq.mi.	1			lb/yr
	Brooks Run	P	10 sq.mi.	29			lb/yr

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions (Continued)

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
LA							
	N/A						
MA							
	Lake Boon	P	684 ha.		254	28	kg/yr
	Browning Pond	P	106 ac.		200		kg/yr
	Long Pond	P	18 ac.		68		kg/yr
	Minechong Pond	P	21 ac.		53		kg/yr
	Mona Lake	P	11 ac.		19		kg/yr
	Spectacle Pond	P	16 ac.		14		kg/yr
	Sugden Reservoir	P	83 ac.		230		kg/yr
	Wickabong Pong	P	320 ac.		729		kg/yr
	Leesville Pond	P	6358 ha.		1,007	53	kg/yr
	Indian Lake	P	795 ha.		298	16	kg/yr
	Lake Quinsigamond	P	118.2 sq.km.	18	1,117	60	kg/yr
	Salisburg Pond	P	1820 ha.		1,028	54	kg/yr
	Bare Hill Pond	P	1082 ha.		511	27	kg/yr
MD							
	Wicomico River	N	441 sq.km.	409,130	832,460	24,940	lb/yr
	Wicomico River	P	441 sq.km.	68,190	33,850	1,440	lb/yr
	Chicamacomico Run	N	51.6 sq.mi.		197,500	6,108	lb/yr
	Chicamacomico Run	P	51.6 sq.mi.		13,587	420	lb/yr
	Johnson Pond	P	10114 ha.	1,135	3,449	509	lb/yr
	Manokin River	N	81.8 sq.mi.	42,730	42,730	9,060	lb/yr
	Marshyhope Creek	P	560 sq.km.	4,980	2,988	1,236	lb/yr
	Wicomico Creek	N	31.2 sq.mi.		101,538	3,046	lb/yr
	Wicomico Creek	P	31.2 sq.mi.		5,833	175	lb/yr
	Bohemia River	N	55.5 sq.mi.	4,380	9,528	504	lb/yr
	Bohemia River	P	55.5 sq.mi.	1,224	588	36	lb/yr
	Corsica River	N	40 sq.mi.	7,598	268,211	11,861	lb/yr
	Corsica River	P	40 sq.mi.	1,424	19,380	1,440	lb/yr
	Transquaking River	N	110.8 sq.mi.	14,954	410,729	13,170	lb/yr
	Transquaking River	P	110.8 sq.mi.	1,496	29,298	952	lb/yr
	Broadford Lake	P	6.8 sq.mi.		1,095	122	lb/yr
	Lake Habeeb	P	8.8 sq.mi.		836	93	lb/yr
	Tony Tank Lake	P	13.8 sq.mi.		662	74	lb/yr
	Port Tobacco River	N	44 sq.mi.	24,920	190,470	27,920	lb/yr
	Port Tobacco River	P	44 sq.mi.	4,060	12,500	2,010	lb/yr
	Fairlake Creek	N	13.2 sq.mi.	260	79,490	3,670	lb/yr
	Fairlake Creek	P	13 2 sq.mi.	140	5,780	390	lb/yr
ME							
	Cobbossee Lake	P	32.3 sq.mi.		5,904		kg/yr
	Madawaska Lake	P	5691 ha.		1,836	263	kg/yr

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions (Continued)

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
	Sebasticook Lake	P	21995 ha.		4,514	301	kg/yr
	East Pond	P	4.3 sq.mi.		389	12	kg/yr
	China Lake	P	32 sq.mi.		2,850	378	kg/yr
MI							
	N/A						
MN							
	N/A						
MO							
	James River	N	987 sq.mi.	3,949	4,000		lb/yr
	James River	P	987 sq.mi.	197	200		lb/yr
MS							
	N/A						
MT							
	N/A						
NC							
	Neuse River Basin	N	6200 sq.mi.	1,640,000	5,120,000		lb/yr
ND							
	N/A						
NE							
	N/A						
NH							
	N/A						
NJ							
	Sylvan Lake	P	459 ac.		66		kg/yr
	Strawbridge Lake	P	12.6 sq.mi.	222	565		kg/yr
NM							
	Rio Chamita	P	38 sq.mi.	1	1		lb/yr
	Redondo Creek	P	12 sq.mi.		0	0	lb/yr
NV							
	Truckee River	N	2300 sq.mi.	500	500		lb/yr
	Truckee River	P	2300 sq.mi.	134	80		lb/yr
NY							
	Amawalk Reservoir	P	42.2 sq.km	390	806	133	kg/yr
	Bog Brook Reservoir	P	9.5 sq.km.	28	309	38	kg/yr
	Boyd Corners Reservoir	P	60.9 sq.km.		869	97	kg/yr
	Cross River Reservoir	P	77.2 sq.km.	108	1,067	168	kg/yr
	Croton Falls Reservoir	P	43 8 sq.km.	615	3,425	713	kg/yr
	Diverty Reservoir	P	18.9 sq.km.	232	2,160	406	kg/yr
	East Branch Reservoir	P	198.4 sq.km.	449	2,020	353	kg/yr
	Kensico Reservoir	P	26 sq.km.		25,448	2,828	kg/yr
	Middle Branch Reservoir	P	55.2 sq.km.	173	643	133	kg/yr
	Muscoot Reservoir	P	193.7 sq.km.	1,405	7,052	940	kg/yr

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions (Continued)

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
	New Croton	P	153.6 sq.km.	209	11,468	1,297	kg/yr
	Titicus Reservoir	P	60.3 sq.km.		984	174	kg/yr
	West Branch Reservoir	P	52.8 sq.km.	28	15,285	1,701	kg/yr
	Ashokan Reservoir	P	661 sq.km.	268	77,661	8,659	kg/yr
	Cannonville Reservoir	P	1178 sq.km.	1,059	45,885	6,706	kg/yr
	Neversink Reservoir	P	238 sq.km.		20,298	2,255	kg/yr
	Pepacton Reservoir	P	961 sq.km.	388	70,862	7,917	kg/yr
	Randout Reservoir	P	245 sq.km.	125	49,570	5,522	kg/yr
	Schoharje Reservoir	P	818 sq.km.	789	25,996	2,976	kg/yr
OH							
	Middle Cuyahoga River	N	135 sq.mi.	1,062	290		kg/d
	Rocky River	N	292 sq.mi.	311,360	234,195		kg/yr
	Rocky River	P	292 sq.mi.	11,570	17,273		kg/yr
	Little Miami River	P	657 sq.mi.	74	274		kg/d
	Sugar Creek	P	357 sq.mi.	18	140		kg/d
OK							
	N/A						
OR							
	Snake River and Hells Canyon	P	73000 sq.mi.	2,495	916		kg/d
	Upper Klamath Lakes	P	9758 sq.km.	1,300	108,700		kg/yr
PA							
	Chickies Creek	P	65 sq.mi.	8,809	27,151	3,996	lb/yr
	Conewago Creek	P	53.2 sq.mi.	2,038	9,340	1,264	lb/yr
	Conneaut Lake	P	16352 ac.		4,481	149	lb/yr
	Conodoguinet Creek	P	507 sq.mi.	1,765	33,639	3,933	lb/yr
	Conowingo Creek	P	34 sq.mi.		19,204	2,134	lb/yr
	Deep Run	P	6.2 sq.mi.		2,561	285	lb/yr
	Donegal Creek	P	17.2 sq.mi.		2,958	329	lb/yr
	Earlakill Run	P	4.4 sq.mi.		1,802	200	lb/yr
	Lake Carey	P	2879 ac.	605	169	86	lb/yr
	Lake Luxemberg	P	6306 ac.		1,686	28	lb/yr
	Muddy Run	P	9 sq.mi.		5,237	582	lb/yr
	North Branch Mahan Tango	P	3195 ac.		1,273	141	lb/yr
	North Fork Cowanesque River	P	16.25 sq.mi.		4,402	489	lb/yr
	Pequea Creek	N	148 sq.mi.	22,790	1,235,404	139,799	lb/yr
	Pequea Creek	P	148 sq.mi.	4,961	63,923	7,654	lb/yr
	Quittapahilla Creek	P	77 sq.mi.	1,129	12,950	1,565	lb/yr
	South Branch Wyalusing Creek	N	5.4 sq.mi.		12,976	1,442	lb/yr
	South Branch Wyalusing Creek	P	5.4 sq.mi.		1,672	186	lb/yr
	Stephen Foster Lake	P	6577 ac.		1,372	23	lb/yr
	Virgin Run Lake	P	2131 ac.		271	30	lb/yr

Table D.1 Reported TMDLs in the Evaluation of the Critical Loads for Nitrogen and Phosphorus Emissions (Continued)

State	Water	Pollutant	Area	TMDL			Unit
				WLA	LA	MOS	
RI							
	Stafford Pond	P	947 ac.		390	21	kg/yr
SC							
	N/A						
SD							
	Elm Lake	P	165240 ac.		10,800		kg/yr
	Lake Hendricks	P	31693 ac.		1,672		kg/yr
TN							
	N/A						
TX							
	N/A						
UT							
	Upper East Canyon Reservoir	P	144 sq.mi.	1,462	4,093	92	lb/yr
	Spring Creek	P	29.3 sq.mi.	310	1,445	195	kg/yr
	Bear River	P	7118 sq.mi.	20	692		kg/d
VA							
	Muddy Creek	N	77000 ac.	49,389	1,177,882		lb/yr
VT							
	N/A						
WA							
	N/A						
WI							
	N/A						
WV							
	Bear Lake	P	185.8 ha.		52	3	kg/yr
	Castleman Run Lake	P	5256 ac.		601	23	kg/yr
	Ridenour Lake	P	613 ha.		162	9	kg/yr
	Turkey Run Lake	P	2147 ac.		223	8	kg/yr
WY							
	N/A						

Table D.2 List of Online Reference for TMDL Reports

State	Water	Online Reference
AK		http://www.state.ak.us/dec/dawq/tmdl/index.htm
	N/A	
AL		http://www.adem.state.al.us/EnviroProtect/Water/Surface/tmdl/tmdl.htm
	Lake Weiss	http://www.epa.gov/region4/water/tmdl/alabama/weiss/WeissTMDL.PDF
AR		http://www.adeq.state.ar.us/water/branch_planning.htm#TMDL
	Hicks Creek	http://www.adeq.state.ar.us/ftproot/Pub/pa/TMDL_Summaries/2001-01-19_TMDL_for_Hicks_Creek_(PDF_File).pdf
	Holman Creek	http://www.adeq.state.ar.us/ftproot/Pub/pa/TMDL_Summaries/2001-01-19_TMDL_for_Town_Branch_and_Holman_Creek_(PDF_File).pdf
	Whig Creek	http://www.adeq.state.ar.us/ftproot/Pub/pa/TMDL_Summaries/2001-01-19_TMDL_for_Whig_Creek_(PDF_File).pdf
AZ		http://www.adeq.state.az.us/environ/water/assess/tmdl.html
	N/A	
CA		http://www.swrcb.ca.gov/tmdl/303d_lists.html
	Newport Bay	http://www.epa.gov/region09/water/tmdl/newport/npnutmdl.pdf
	Newport Bay	http://www.epa.gov/region09/water/tmdl/newport/npnutmdl.pdf
CO		http://water.state.co.us/default.asp
	N/A	
CT		http://dep.state.ct.us/wtr/
	Long Island Sound	http://dep.state.ct.us/wtr/lis/tmdl.pdf
DC		
	N/A	
DE		http://www.dnrec.state.de.us/water2000/Sections/Watershed/TMDL/tmdlinfo.htm
	Inland Bays	http://www.dnrec.state.de.us/newpages/pdf/ibtmdlanalysis.pdf
	Inland Bays	http://www.dnrec.state.de.us/newpages/pdf/ibtmdlanalysis.pdf
	Murderkill River	http://www.dnrec.state.de.us/water2000/Sections/Watershed/TMDL/MKTechDoc.PDF
	Murderkill River	http://www.dnrec.state.de.us/water2000/Sections/Watershed/TMDL/MKTechDoc.PDF
	Nanticoke	http://www.dnrec.state.de.us/DNREC2000/Library/Misc/Unorg/nbrtmdla.pdf
	Nanticoke	http://www.dnrec.state.de.us/DNREC2000/Library/Misc/Unorg/nbrtmdla.pdf
FL		http://www.dep.state.fl.us/water/tmdl/index.htm
	N/A	
GA		http://www.ganet.org/dnr/environ/
	N/A	
HI		http://www.state.hi.us/dlnr/cwrm/
	N/A	
IA		http://www.deq.state.la.us/technology/tmdl/index.htm
	Silver Lake	http://www.state.ia.us/government/dnr/organiza/epd/wtresrce/files/silverfinal.pdf
	Rock Creek Lake	http://www.state.ia.us/government/dnr/organiza/epd/wtresrce/files/rock2.pdf
	Lake Miami	http://www.state.ia.us/government/dnr/organiza/epd/wtresrce/files/miami2.pdf
ID		http://www2.state.id.us/deq/water/water1.htm#TMDLs
	Cottonwood Creek	http://www2.state.id.us/deq/water/tmdls/cottonwood/cottonwood_tmdl.htm
	Cottonwood Creek	http://www2.state.id.us/deq/water/tmdls/cottonwood/cottonwood_tmdl.htm
	Jim Ford Creek	http://www2.state.id.us/deq/water/tmdls/jimford/jimford_tmdl.htm
	Jim Ford Creek	http://www2.state.id.us/deq/water/tmdls/jimford/jimford_tmdl.htm
	Snake River and Hells Canyon	http://www2.state.id.us/deq/water/tmdls/snakeriver_hellscanyon/Snake_River_Hells_Canyon_11.pdf
	Winchester Lake	http://www2.state.id.us/deq/water/tmdls/winchester/winchester_tmdl.htm
IL		http://www.epa.state.il.us/water/tmdl/index.html
	Governor Bond Lake	http://www.epa.state.il.us/water/watershed/publications/governor-bond-lake.pdf
IN		http://www.in.gov/idem/water/assessbr/tmdl_assess.html
	Kokomo Lake	http://www.in.gov/idem/water/assessbr/Kokomo.pdf
	Kokomo Lake	http://www.in.gov/idem/water/assessbr/Kokomo.pdf
KS		http://www.kdhe.state.ks.us/tmdl/
	Russell Lake	http://www.kdhe.state.ks.us/tmdl/ci/russell.pdf
	Lake Meade	http://www.kdhe.state.ks.us/tmdl/ci/MeadeE.pdf
	Hamilton Lake	http://www.kdhe.state.ks.us/tmdl/ua/HamiltonWAE.pdf

Table D.2 List of Online Reference for TMDL Reports (Continued)

State	Water	Online Reference
	Ford County Lake	http://www.kdhe.state.ks.us/tmdl/ua/FordE.pdf
	Ford County Lake	http://www.kdhe.state.ks.us/tmdl/ua/FordE.pdf
	Stone Lake	http://www.kdhe.state.ks.us/tmdl/ua/StoneE.pdf
	Blue River	http://www.kdhe.state.ks.us/tmdl/mo/BlueRBOD_Nutr.pdf
	Pony Creek Lake	http://www.kdhe.state.ks.us/tmdl/mo/PonyE.pdf
	Pony Creek Lake	http://www.kdhe.state.ks.us/tmdl/mo/PonyE.pdf
	Lake Jewell	http://www.kdhe.state.ks.us/tmdl/klr/jewellE.pdf
	Mission Lake	http://www.kdhe.state.ks.us/tmdl/klr/MissionE.pdf
	Little Lake	http://www.kdhe.state.ks.us/tmdl/klr/little.pdf
	Clinton Lake	http://www.kdhe.state.ks.us/tmdl/klr/ClintonE.pdf
	Tuttle Creek Lake	http://www.kdhe.state.ks.us/tmdl/klr/TuttleE.pdf
	Pomona Lake	http://www.kdhe.state.ks.us/tmdl/mc/PomonaE.pdf
	Hillsdale Lake	http://www.kdhe.state.ks.us/tmdl/mc/HillsdaleE.pdf
	Quivira Big Salt Marsh	http://www.kdhe.state.ks.us/tmdl/la/QuiviraBigE.pdf
	Quivira Little Salt Marsh	http://www.kdhe.state.ks.us/tmdl/la/QuiviraLittleE.pdf
	Mingenback Lake	http://www.kdhe.state.ks.us/tmdl/la/Mingenback.pdf
	Mingenback Lake	http://www.kdhe.state.ks.us/tmdl/la/Mingenback.pdf
	Pracht Weltand	http://www.kdhe.state.ks.us/tmdl/la/CadillacE.pdf
	Pracht Weltand	http://www.kdhe.state.ks.us/tmdl/la/CadillacE.pdf
	Cheny Lake	http://www.kdhe.state.ks.us/tmdl/la/CheneyE.pdf
	Hargis Lake	http://www.kdhe.state.ks.us/tmdl/la/HargisE.pdf
	Hargis Lake	http://www.kdhe.state.ks.us/tmdl/la/HargisE.pdf
KY		http://water.nr.state.ky.us/dow/tmdl.htm
	Baughman Fork	http://water.nr.state.ky.us/dow/baughman.pdf
	Brooks Run	http://water.nr.state.ky.us/dow/brooks1.pdf
LA		http://www.deq.state.la.us/technology/tmdl/index.htm
	N/A	
MA		http://www.state.ma.us/dep/brp/wm/wmpubs.htm
	Lake Boon	http://www.state.ma.us/dep/brp/wm/files/lakeboon.pdf
	Browning Pond	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Long Pond	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Minechong Pond	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Mona Lake	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Spectacle Pond	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Sugden Reservoir	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Wickabong Pong	http://www.state.ma.us/dep/brp/wm/files/chicopee.doc
	Leesville Pond	http://www.state.ma.us/dep/brp/wm/files/leesvill.pdf
	Indian Lake	http://www.state.ma.us/dep/brp/wm/files/indianma.pdf
	Lake Quinsigamond	http://www.state.ma.us/dep/brp/wm/files/quinsig.doc
	Salisburg Pond	http://www.state.ma.us/dep/brp/wm/files/salisbur.doc
	Bare Hill Pond	http://www.state.ma.us/dep/brp/wm/files/fbh81007.pdf
MD		http://www.mde.state.md.us/tmdl/index.html
	Wicomico River	http://www.mde.state.md.us/tmdl/lower_wicomico/lwr_wicomico_main_fin.PDF
	Wicomico River	http://www.mde.state.md.us/tmdl/lower_wicomico/lwr_wicomico_main_fin.PDF
	Chicamacomico Run	http://www.mde.state.md.us/tmdl/chicamacomico/chica_tmdl_main_fin.pdf
	Chicamacomico Run	http://www.mde.state.md.us/tmdl/chicamacomico/chica_tmdl_main_fin.pdf
	Johnson Pond	http://www.mde.state.md.us/tmdl/johnson/jp_tmdl_main_and_appx.pdf
	Manokin River	http://www.mde.state.md.us/tmdl/manokin/man_main_fin.pdf
	Marshyhope Creek	http://www.mde.state.md.us/tmdl/marshyhope/mh_main.pdf
	Wicomico Creek	http://www.mde.state.md.us/tmdl/wico_creek/wcr_tmdl_main_fin.pdf
	Wicomico Creek	http://www.mde.state.md.us/tmdl/wico_creek/wcr_tmdl_main_fin.pdf

Table D.2 List of Online Reference for TMDL Reports (Continued)

State	Water	Online Reference
	Bohemia River	http://www.mde.state.md.us/tmdl/bohemia/boh_tmdl_main_fin.pdf
	Bohemia River	http://www.mde.state.md.us/tmdl/bohemia/boh_tmdl_main_fin.pdf
	Corsica River	http://www.mde.state.md.us/tmdl/corsica/corsica_tmdl_fin.PDF
	Corsica River	http://www.mde.state.md.us/tmdl/corsica/corsica_tmdl_fin.PDF
	Transquaking River	http://www.mde.state.md.us/tmdl/transquaking/transquaking_tmdl.PDF
	Transquaking River	http://www.mde.state.md.us/tmdl/transquaking/transquaking_tmdl.PDF
	Broadford Lake	http://www.mde.state.md.us/tmdl/broadford/broadford_tmdl.pdf
	Lake Habeeb	http://www.mde.state.md.us/tmdl/habeeb/habeeb_tmdl.pdf
	Tony Tank Lake	http://www.mde.state.md.us/tmdl/tonytank/tonytank_tmdl.pdf
	Port Tobacco River	http://www.mde.state.md.us/tmdl/porttobacco/pt_tmdl_fin.PDF
	Port Tobacco River	http://www.mde.state.md.us/tmdl/porttobacco/pt_tmdl_fin.PDF
	Fairlake Creek	http://www.mde.state.md.us/tmdl/fairlee/fc_tmdl_fin.PDF
	Fairlake Creek	http://www.mde.state.md.us/tmdl/fairlee/fc_tmdl_fin.PDF
ME		http://www.state.me.us/dep/blwq/monitoring.htm#303
	Cobbossee Lake	http://www.state.me.us/dep/blwq/docmonitoring/tmdlcobsupp.pdf
	Madawaska Lake	http://www.state.me.us/dep/blwq/docmonitoring/tmdlmada.pdf
	Sebasticook Lake	http://www.state.me.us/dep/blwq/docmonitoring/tmdlsebrep.pdf
	East Pond	http://www.state.me.us/dep/blwq/docmonitoring/tmdleastpondrep.pdf
	China Lake	http://www.state.me.us/dep/blwq/docmonitoring/tmdlchinalakerep.pdf
MI		http://www.deq.state.mi.us/swq/gleas/gleas.htm
	N/A	
MN		http://www.pca.state.mn.us/water/tmdl.html
	N/A	
MO		http://www.dnr.state.mo.us/deq/wpcp/wpc-tmdl.htm
	James River	http://www.dnr.state.mo.us/deq/wpcp/tmdl/james_river_final_tmdl.pdf
	James River	http://www.dnr.state.mo.us/deq/wpcp/tmdl/james_river_final_tmdl.pdf
MS		http://www.deq.state.ms.us/newweb/swhome.nsf/pages/SWDivision/$file/tmd3.html
	N/A	
MT		http://deq.state.mt.us/ppa/mdm/TMDL/tmdl_index.asp
	N/A	
NC		http://h2o.enr.state.nc.us/
	Neuse River Basin	http://h2o.enr.state.nc.us/mtu/files/TMDL/Neuse_TMDL_1999.pdf
ND		http://www.health.state.nd.us/ndhd/environ/wq/index.htm
	N/A	
NE		http://www.deq.state.ne.us/
	N/A	
NH		http://www.des.state.nh.us/water_intro.htm
	N/A	
NJ		http://www.state.nj.us/dep/watershedmgt/tmdl.htm
	Sylvan Lake	http://www.state.nj.us/dep/watershedmgt/DOCS/pdfs/tmdl/lower_sylvan_lake_tmdl.pdf
	Strawbridge Lake	http://www.state.nj.us/dep/watershedmgt/DOCS/pdfs/tmdl/strawbridge_lake_tmdl.pdf
NM		http://www.nmenv.state.nm.us/swqb/tmdlds.html
	Rio Chamita	http://www.nmenv.state.nm.us/swqb/Ammonia_Fecal_Coliform_Phosphorus_TMDL_For_Rio_Chamita_08-19-2000.pdf
	Redondo Creek	http://www.nmenv.state.nm.us/swqb/Total_Phosphorus_TMDL_for_Redondo_Creek.pdf
NV		http://ndep.state.nv.us/bwqp/tmdl.htm
	Truckee River	http://ndep.state.nv.us/bwqp/truckee2.pdf
	Truckee River	http://ndep.state.nv.us/bwqp/truckee2.pdf
NY		http://www.dec.state.ny.us/website/dow/tmdl.html
	Amawalk Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/amawalk.pdf
	Bog Brook Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/bogbrook.pdf
	Boyd Corners Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/boydscorner.pdf

Table D.2 List of Online Reference for TMDL Reports (Continued)

State	Water	Online Reference
	Cross River Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/crossriver.pdf
	Croton Falls Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/crotonfalls.pdf
	Diverty Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/diverting.pdf
	East Branch Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/eastbranch.pdf
	Kansico Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/kensico.pdf
	Middle Branch Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/midbranch.pdf
	Muscoot Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/muscoot.pdf
	New Croton	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/newcroton.pdf
	Titicus Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/titicus.pdf
	West Branch Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/westbranch.pdf
	Ashokan Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/ashokan.pdf
	Cannonville Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/cannonsville.pdf
	Neversink Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/neversink.pdf
	Pepacton Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/pepacton.pdf
	Randout Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/rondout.pdf
	Schoharje Reservoir	http://www.ci.nyc.ny.us/html/dep/pdf/tmdl/schoharie.pdf
OH		http://www.epa.state.oh.us/dsw/tmdl/index.html
	Middle Cuyahoga River	http://www.epa.state.oh.us/dsw/tmdl/MidCuyFinalTMDL.pdf
	Rocky River	http://www.epa.state.oh.us/dsw/documents/RockyRiverTMDL11_28draft.pdf
	Rocky River	http://www.epa.state.oh.us/dsw/documents/RockyRiverTMDL11_28draft.pdf
	Little Miami River	http://www.epa.state.oh.us/dsw/tmdl/ULMRdraft2.pdf
	Sugar Creek	http://www.epa.state.oh.us/dsw/tmdl/SugarCrFinalDraftTMDL.pdf
OK		http://www.deq.state.ok.us/WQDnew/tmdl/index.html
	N/A	
OR		http://www.deq.state.or.us/wq/tmdls/tmdls.htm
	Snake River and Hells Canyon	http://www2.state.id.us/deq/water/tmdls/snakeriver_hellscanyon/Snake_River_Hells_Canyon_11.pdf
	Upper Klamath Lakes	http://www.deq.state.or.us/wq/TMDLs/UprKlamath/Walker_Report.pdf
PA		http://www.dep.state.pa.us/watermanagement_apps/tmdl/
	Chickies Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Chickies_TMDL.pdf
	Conewago Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Conewago_TMDL.pdf
	Conneaut Lake	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Conneaut_TMDL.pdf
	Conodoguinet Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Conodoguinet_TMDL.pdf
	Conowingo Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Conowingo_TMDL.pdf
	Deep Run	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Deep_Run_TMDL.pdf
	Donegal Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Donegal_TMDL.pdf
	Earlakill Run	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Earlakill_TMDL.pdf
	Lake Carey	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Carey_TMDL.pdf
	Lake Luxemberg	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Lux_TMDL.pdf
	Muddy Run	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Muddy_TMDL.pdf
	North Branch Mahan Tango	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/NB_Mahantango_TMDL.pdf
	North Fork Cowanesque River	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/NF_Cowanesque_TMDL.pdf
	Pequea Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Pequea_TMDL.pdf
	Pequea Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Pequea_TMDL.pdf
	Quittapahilla Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Quitty_TMDL.pdf
	South Branch Wyalusing Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/South_Wyalusing_TMDL.pdf
	South Branch Wyalusing Creek	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/South_Wyalusing_TMDL.pdf
	Stephen Foster Lake	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/SFoster_TMDL.pdf
	Virgin Run Lake	http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/Virgin_Rn_TMDL.pdf

Table D.2 List of Online Reference for TMDL Reports (Continued)

State	Water	Online Reference
RI		http://www.state.ri.us/dem/programs/benviron/water/quality/rest/index.htm
	Stafford Pond	http://www.state.ri.us/dem/programs/benviron/water/quality/rest/pdfs/stafford.pdf
SC		http://www.scdhec.net/water/
	N/A	
SD		http://www.state.sd.us/denr/TMDL/denrtmdl.htm
	Elm Lake	http://www.state.sd.us/denr/DFTA/WatershedProtection/TMDL/TMDLELM.htm
	Lake Hendricks	http://www.state.sd.us/denr/DFTA/WatershedProtection/TMDL/TMDLHendricks.htm
TN		http://www.state.tn.us/environment/wpc/tmdl.htm
	N/A	
TX		http://www.tnrcc.state.tx.us/water/quality/tmdl/index.html
	N/A	
UT		http://www.deq.state.ut.us/EQWQ/TMDL/TMDL_WEB.HTM
	Upper East Canyon Reservoir	http://www.deq.state.ut.us/EQWQ/TMDL/ecr_tmdl_f.pdf
	Spring Creek	http://www.deq.state.ut.us/EQWQ/TMDL/SpringCK_report.pdf
	Bear River	http://www.deq.state.ut.us/EQWQ/TMDL/LowerBearTMDL_2-42-02.pdf
VA		http://www.deq.state.va.us/tmdl/
	Muddy Creek	http://www.deq.state.va.us/tmdl/tmdls/shenrvr/muddyni.pdf
VT		http://www.anr.state.vt.us/dec/waterq/planningTMDL.htm
	N/A	
WA		http://www.ecy.wa.gov/programs/wq/tmdl/index.html
	N/A	
WI		http://www.dnr.state.wi.us/org/water/wm/wqs/303d/index.html
	N/A	
WV		http://www.dep.state.wv.us/wr/index.cfm?page=OWR_Website/C_and_D/tmdl.htm
	Bear Lake	http://www.epa.gov/reg3wapd/tmdl/pdf/bear629.pdf
	Castleman Run Lake	http://www.epa.gov/reg3wapd/tmdl/pdf/castlemn.pdf
	Ridenour Lake	http://www.epa.gov/reg3wapd/tmdl/pdf/ridenour.pdf
	Turkey Run Lake	http://www.epa.gov/reg3wapd/tmdl/pdf/turkeyrn.pdf
WY		http://deq.state.wy.us/wqd/wtrshedpg.htm
	N/A	

APPENDIX E

PNECs OF CHEMICALS WITH HIGH PRODUCTION VOLUME

This appendix summarizes the available predicted no-effect concentrations (PNECs) of high production volume (HPV) chemicals within OECD. These PNECs are the ones that are available online on the OECD website (http://cs3-hq.oecd.org/scripts/hpv/).

Table E.1 Summary of PNEC Studied by OECD Countries
(Source: OECD 2000)

OECD Country	Chemical	Predicted no-effect concentration (PNEC)			
		Aquatic mg/l	Microorganism mg/l	Sediment mg/kg	Soil mg/kg
Australia	Vinyl-pyrrolidinone (CAS 88120)	0.045	20	0.0518	0.0187
	Benzene, 1,2-dichloro- (CAS 95501)	0.0063			
	Dioxane (CAS 123911)	57.5			
Austria	Aniline, 2-methoxy- (CAS 90040)	0.0055	8	0.008	0.0037
Belgium	Benzene, 1,1'-methylenebis(isocyanato- (CAS 26447405)	1			1
Denmark	Toluene (CAS 108883)	0.074	8.4		4.3
	Benzene, 1,2,4-trichloro- (CAS 120821)	0.004	10		0.05
	Phenol, 4-chloro-2-methyl- (CAS 1570645)	0.05	0.55		0.36
	Methyl t-butyl ether (CAS 1634044)	2.6		2.05	0.73
	Hydrogen peroxide (CAS 7722841)	0.01			
	Ethane, 1-chloro-1,1-difluoro- (CAS 75683)	0.045			
	Ethane, 1,1,2,2-tetrachloro- (CAS 79345)	0.14			
	Aniline, 2-nitro- (CAS 88744)	0.008			
	Guanidine, 1,3-diphenyl- (CAS 102067)	0.006			
	Benzene, 1,4-dichloro- (CAS 106467)	0.02			
	Glyoxal (CAS 107222)	0.215			
	Cyclohexane (CAS 110827)	0.009			
	Undecanoic acid, 11-amino- (CAS 2432997)	0.045			
Germany	Caffeine (CAS 58082)	0.0058			
	Theophylline (CAS 58559)	0.087			
	E.D.T.A (CAS 60004)	0.1			
	Tetrasodium E.D.T.A. (CAS 64028)	0.0015			
	Acrylic acid (CAS 79107)	0.003	0.9		0.1
	Methyl acetate (CAS 79209)	0.32			
	Methacrylic acid (CAS 79414)	0.164			
	Methacrylate, methyl- (CAS 80626)	0.74			
	Benzene, 1-chloro-2-nitro- (CAS 88733)	0.026			0.0032
	Phthalonitrile (CAS 91156)	0.14			
	Toluene, 2-chloro- (CAS 95498)	0.014			0.089
	Aniline, 3,4-dichloro- (CAS 95761)	0.003			
	Benzene, 1-chloro-4-nitro- (CAS 100005)	0.0028			
	Ethanol, 2-(diethylamino)- (CAS 100378)	0.044			
	Aniline, 4,4'-methylenebis- (CAS 101779)	0.003			
	Azepin-2-one, hexahydro- (CAS 105602)	0.13			
	Butane, 1,2-epoxy- (CAS 106887)	0.02			
	Ethane, 1,2-dichloro- (CAS 107062)	1.1			
	Octadecanaminium, N,N-dimethyl-N-octadecyl-, chloride (CAS 107642)	0.0062		55	20
	Propane, 1-Amino-3-dimethylamino- (CAS 109557)	0.056			
	Dodecanamine, N,N-dimethyl- (CAS 112185)	2.35E-05			
	Triphenyl phosphate (CAS 115866)	0.00074			
	Pentanedione (CAS 123546)	0.005			
	Di-tert-butyl-p-cresol (CAS 128370)	0.0014			
	Ethyl acetoacetate (CAS 141979)	0.275			
	Butanal, 3-methyl- (CAS 590863)	0.0033			

Table E.1 Summary of PNEC Studied by OECD Countries (Continued)

OECD Country	Chemical	Predicted no-effect concentration (PNEC)			
		Aquatic mg/l	Microorganism mg/l	Sediment mg/kg	Soil mg/kg
	Butene, 3,4-dichloro- (CAS 760236)	0.0083			
	Hexamethylene diisocyanate (CAS 822060)	0.0774			
	Imidazolidinone, 4,5-dihydroxy-1,3-bis (CAS 1854268)	0.6			
	dimethyl-4,4'-methylenebis(cyclohexylamine) (CAS 6864375)	0.0021			
	Benzenesulfonic acid, 2,2'-(1,2-ethenediyl)bis[5-[[4-[bis(2-hydroxyethyl)am (CAS 16470249)	0.2		4.3	10
	Formaldehyde, prods. with sulfonated 1,1'-oxybistoluene, sod (CAS 90387578)	0.029			
Italy	Terephthalic acid (CAS 100210)	8			
	Dimethyl terephthalate (CAS 120616)	0.096			
	Acetamide, N,N-dimethyl- (CAS 127195)	0.5			
Japan	Dicyclopentadiene (CAS 77736)	0.032			
	Propanenitrile, 2-hydroxy- (CAS 78977)	0.0017			
	Diaminostilbene-2,2'-disulfonic acid (CAS 81118)	0.32			
	Anthraquinone, 1-amino- (CAS 82451)	0.001			
	Benzene, 1,4-dichloro-2-nitro- (CAS 89612)	0.01			
	Phthalonitrile (CAS 91156)	0.14			
	Triazine-2,4-diamine, 6-phenyl- (CAS 91769)	0.0191			
	Toluene, 2,4-dichloro- (CAS 95738)	0.02			
	Benzene, (1-methylethenyl)- (CAS 98839)	0.018			
	Benzyl chloride (CAS 100447)	0.001			
	Benzene, 1,4-diethyl- (CAS 105055)	0.0093			
	Dibutyl adipate (CAS 105997)	0.02			
	Glycidyl methacrylate (CAS 106912)	0.01			
	Dibutyl phosphate (CAS 107664)	0.66			
	Toluidine (CAS 108441)	0.0001			
	Propanol, 1-methoxy-, acetate (CAS 108656)	0.635			
	Butane, 1-chloro- (CAS 109693)	0.14			
	Butanediol (CAS 110634)	0.85			
	Pentaerythritol (CAS 115775)	0.6			
	Dichlorotoluene (CAS 118694)	0.0032			
	Cresol, 6,6'-di-tert-butyl-2,2'-methylenedi- (CAS 119471)	0.0068		2	
	Diacetone alcohol (CAS 123422)	1			
	Naphthol (CAS 135193)	0.00085			
	3H-Indol-3-one, 2-(1,3-dihydro-3-oxo-2H-indol-2-ylidene)-1,2-dihydro-(CAS 482893)	0.0078			
	Trimethyl phosphate (CAS 512561)	3.2			
	Benzene, 2,4-dichloro-1-nitro- (CAS 611063)	0.00056			
	Butenedioic acid (2E)-, diethyl ester (CAS 623916)	0.0056			
	Butene, 3,4-dichloro- (CAS 760236)	0.0083			
	Hydroxyethyl methacrylate (CAS 868779)	0.141			
	Phenylene-bis(methylamine) (CAS 1477550)	0.047			
	Phenol, 2-(1,1-dimethylethyl)-4,6-dimethyl- (CAS 1879090)	0.14			
	Piperidinol, 2,2,6,6-tetramethyl- (CAS 2403885)	0.037			
	Pentanediol, 3-methyl- (CAS 4457710)	1			
	C.I.Pigment Red 57:1 (CAS 5281049)	0.03			
	Citral (CAS 5392405)	0.01			

Table E.1 Summary of PNEC Studied by OECD Countries (Continued)

OECD Country	Chemical	Predicted no-effect concentration (PNEC) Aquatic mg/l	Microorganism mg/l	Sediment mg/kg	Soil mg/kg
	Propanoic acid, 2-methyl-, 2,2-dimethyl-1-(1-methylethyl)-1,3-propanediyl e (CAS 6846500)	0.032			
	Propanol, [(1-methyl-1,2-ethanediyl)bis(oxy)]bis- (CAS 24800440)	10			
	Diphenyl tolyl phosphate (CAS 26444495)	0.0012			
Korea	Acetamide, N-phenyl- (CAS 103844)	0.135			
	Disodium disulphite (CAS 7681574)	0.1			
Norway	Pentane (CAS 109660)	0.027		0.424	0.494
Spain	Cumene (CAS 98828)	0.022		0.388	0.34
Sweden	Toluene, 2-nitro- (CAS 88722)	0.05			
Switzerland	Octadien-3-ol, 3,7-dimethyl- (CAS 78706)	0.2			
	Picoline, 5-ethyl- (CAS 104905)	0.0689			
	Metilox (CAS 6386385)	0.0025			
The Netherlands	Dimethyl sulfate (CAS 77781)	0.014			
	Acrolein (CAS 107028)	0.0034			
	Diethylenetriamine (CAS 111400)	12			
	Ethanol, 2-(2-methoxyethoxy)- (CAS 111773)	1	71		
	Ethanol, 2-(butoxyethoxy)- (CAS 112345)	57.5			
	Hydrofluoric acid (CAS 7664393)	0.9			11
United Kingdom	Glycerol (CAS 56815)	777		479	92.1
	Oxirane, methyl- (CAS 75569)	0.052		43.2	16.5
	Propenamide (CAS 79061)	0.0204			
	Phenol A (CAS 80057)	0.0016			23
	Vinyl-pyrrolidinone (CAS 88120)	0.045	19.55	0.0518	0.0187
	Phenylenediamine, N-(1-methylethyl)-N'-phenyl- (CAS 101724)	0.00034			
	Pentanediol, 2-methyl- (CAS 107415)	4.3		0.295	0.0786
	Ethene, tetrachloro- (CAS 127184)	0.051	100	0.632	
	Chromium trioxide (CAS 1333820)	0.0034	32	0.15	0.035
	Sodium chromate (Na2CrO4) (CAS 7775113)	0.0034	32	0.15	0.035
	Potassium dichromate (K2Cr2O7) (CAS 7778509)	0.0034	32	0.15	0.035
	Ammonium dichromate (CAS 7789095)	0.0034	32	0.15	0.035
	Sodium dichromate (CAS 10588019)	0.0034	32	0.15	0.035
	Phenol, nonyl- (CAS 25154523)	0.00033			0.3
	Benzene, 1,1'-oxybis-, pentabromo deriv. (CAS 32534819)	0.00053			0.32
	Phenol, 4-nonyl-, branched (CAS 84852153)			0.039	0.3
	Alkanes, C10 - C13, chloro- (CAS 85535848)	0.0005			
	Alkanes, C 14-17, chloro- (CAS 85535859)	0.0002			2.1
United States	Propanediol (CAS 57556)	183			
	Acetone (CAS 67641)	21			
	Cumene (CAS 98828)	0.022		0.388	0.34
	Terephthalic acid (CAS 100210)	8			
	Acetoacetanilide (CAS 102012)	0.32			
	(2-ethylhexyl) adipate (CAS 103231)	0.0035			
	Glycidyl methacrylate (CAS 106912)	0.01			
	Propanol, 1-methoxy- (CAS 107982)	208			
	Propanol, 1,1'-oxydi- (CAS 110985)	32			

Table E.1 Summary of PNEC Studied by OECD Countries (Continued)

OECD Country	Chemical	Predicted no-effect concentration (PNEC)			
		Aquatic mg/l	Microorganism mg/l	Sediment mg/kg	Soil mg/kg
	Dimethyl terephthalate (CAS 120616)	0.096			
	Hydroquinone (CAS 123319)	0.00044			
	Tributyl phosphate (CAS 126738)	0.037			
	Methyl t-butyl ether (CAS 1634044)	2.6		2.05	
	Ethane, 1,1-dichloro-1-fluoro- (CAS 1717006)	0.31			
	Propanol, 1-chloro-, phosphate (3:1) (CAS 13674845)	0.64			
	Propanol, oxybis- (CAS 25265718)	32			
	Propanol, 1(or 2)-(2-methoxymethylethoxy)- (CAS 34590948)	19			

APPENDIX F

UNCERTAINTY ANALYSIS FOR CARRYING CAPACITY ESTIMATES

This appendix summarizes the uncertainty analysis as the quality index matrix (high to low uncertainty) for sources and causes of uncertainty within components of the carrying capacity estimates and types of uncertainty. The matrices in this appendix are for global warming impact (Table F.1), stratospheric ozone depletion impact (Table F.2), acidification impact (Table F.3), eutrophication impact (Table F.4), photochemical ozone formation impact (Table F.5), eco-toxicity impact (Table F.6), and resource depletion impact (Table F.7).

Table F.1 Uncertainties Associated with the Carrying Capacity Evaluation for Global Warming Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
IPCC emission & mitigation scenarios	Collecting and measuring data	Data inaccuracy Unrepresentative data Uncertainty due to choices	Inherent within sources	High
	Modeling	Model uncertainty Uncertainty due to choices Temporal variability Epistemological uncertainty	Inherent within sources	High
Characterization factors (GWP)	Collecting and measuring data	Data inaccuracy Unrepresentative data Uncertainty due to choices	Inherent within sources	High
	Evaluating	Model uncertainty Uncertainty due to choices Temporal variability Variability between sources and objects	Inherent within sources	High
Carrying capacity estimate	CC Estimating	Data inaccuracy	Low	High
		Model uncertainty	Medium	High
		Uncertainty due to choices	Medium	High
STM implementation	Data acquiring	Uncertainty due to choices	Medium	High
		Data inaccuracy	Medium	High

Table F.2 Uncertainties Associated with the Carrying Capacity Evaluation for Stratospheric Ozone Depletion Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
Projected ODS emission according to Protocol	Collecting and measuring data	Data inaccuracy Unrepresentative data Uncertainty due to choices	Inherent within sources	High
	Modeling	Model uncertainty Uncertainty due to choices Temporal variability Epistemological uncertainty	Inherent within sources	High
Characterization factors (ODP)	Collecting and measuring data	Data inaccuracy Unrepresentative data Uncertainty due to choices	Inherent within sources	High
	Evaluating	Model uncertainty Uncertainty due to choices Temporal variability Variability between sources and objects	Inherent within sources	High
Carrying capacity estimate	CC Estimating	Uncertainty due to choices	Low	High
STM implementation	Data acquiring	Uncertainty due to choices	Low	High
		Data inaccuracy	Low	High

Table F.3 Uncertainties Associated with the Carrying Capacity Evaluation for Acidification Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
Projected emissions according to Title V CAAA 1990	Collecting and measuring data	Data inaccuracy Uncertainty due to choices	Inherent within sources	High
	Evaluating	Model uncertainty Uncertainty due to choices Spatial variability Variability between sources and objects Epistemological uncertainty	Inherent within sources	Medium
Characterization factors (AP)	Evaluating	Uncertainty due to choices Variability between sources and objects Epistemological uncertainty	Inherent within sources	Low
Carrying capacity estimate	CC Estimating	Uncertainty due to choices	Medium	Low
		Spatial variability	High	
		Variability between sources and objects	Medium	
		Epistemological uncertainty	High	
STM implementation	Data acquiring	Uncertainty due to choices	High	Medium
		Data inaccuracy	High	Medium

Table F.4 Uncertainties Associated with the Carrying Capacity Evaluation for Eutrophication Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
TMDLs	Collecting and measuring data	Data inaccuracy Uncertainty due to choices	Inherent within sources	High
	Modeling	Model uncertainty Uncertainty due to choices Spatial variability Variability between sources and objects Epistemological uncertainty	Inherent within sources	Medium
Characterization factors (EP)	Evaluating	Uncertainty due to choices Variability between sources and objects Epistemological uncertainty	Inherent within sources	Low
Carrying capacity estimate	Estimating critical load	Unrepresentative data	High	Medium
		Model uncertainty	Low	
		Spatial variability	Low	
		Temporal variability	Low	
		Variability between sources and objects	Low	
	CC estimating	Uncertainty due to choices	Medium	Medium
		Spatial variability	Low	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
STM implementation	Data acquiring	Uncertainty due to choices	High	Medium
		Data inaccuracy	High	Medium

Table F.5 Uncertainties Associated with the Carrying Capacity Evaluation for Photochemical Ozone Formation Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
OZIPR data	Collecting and measuring data	Data inaccuracy	Inherent within sources	Low
		Data gaps		
		Unrepresentative data		
		Uncertainty due to choices		
Characterization factors (MIR and POCP)	Evaluating	Data inaccuracy	Inherent within sources	High
		Data gaps		
		Model uncertainty		
		Uncertainty due to choices		
		Spatial variability		
		Temporal variability		
		Epistemological uncertainty		
Carrying capacity estimate	Modeling initial VOC, NOx concentrations using OZIPR	Data inaccuracy	High	Low
		Data gaps	High	
		Unrepresentative data	High	
		Model uncertainty	High	
		Uncertainty due to choices	High	
		Spatial variability	High	
		Temporal variability	High	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
	Calculating VOC, NOx emission rates using EKMA	Uncertainty due to choices	High	Low
		Epistemological uncertainty	High	
	CC estimating	Unrepresentative data	Medium	Low
		Uncertainty due to choices	Low	
		Spatial variability	High	
		Temporal variability	Low	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
STM implementation	Data acquiring	Uncertainty due to choices	High	Medium
		Data inaccuracy	High	Medium

Table F.6 Uncertainties Associated with the Carrying Capacity Evaluation for Eco-Toxicity Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
Toxicity data (PNEC)	Collecting and measuring data	Data inaccuracy	Inherent within sources	High
		Data gaps		
		Unrepresentative data		
	Extrapolating	Model uncertainty	Inherent within sources	Medium
		Uncertainty due to choices		
		Variability between sources and objects		
Carrying capacity estimate	CC extrapolating	Unrepresentative data	High	Low
		Uncertainty due to choices	Medium	
		Temporal variability	High	
		Variability between sources and objects	High	
		Epistemological uncertainty	High	
STM implementation	Data acquiring	Uncertainty due to choices	High	Medium
		Data inaccuracy	High	Medium

Table F.7 Uncertainties Associated with the Carrying Capacity Evaluation for Resource Depletion Impact

Component	Procedure containing uncertainty	Type of uncertainty	Uncertainty	Reliability of data sources
Resource reserves	Collecting and estimating data	Data inaccuracy	Inherent within sources	High
		Data gaps		
		Unrepresentative data		
		Spatial variability		
Water recharge data	Collecting and estimating data	Data inaccuracy	Inherent within sources	Medium
		Data gaps		
		Spatial variability		
		Temporal variability		
Carrying capacity estimate	CC estimating	Uncertainty due to choices	High	Medium
		Spatial variability		
		Temporal variability		
		Variability between sources and objects		
		Epistemological uncertainty		
STM implementation	Data acquiring	Uncertainty due to choices	High	Medium
		Data inaccuracy	High	Medium

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