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Best available practice in life cycle assessment of climate change, stratospheric ozone depletion, photo-oxidant formation, acidification, and eutrophication

Backgrounds on general issues

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This report is prepared by the SETAC Europe **S**cientific **T**ask **G**roup on **G**lobal **A**nd **R**egiona**L** **I**mpact **C**ategories (SETAC-Europe/STG-GARLIC) that is installed by the 2nd SETAC Europe working group on life cycle impact assessment. The report is published for the account of the Directiorate-General of the National Institute of Public Health and the Environment (RIVM), within the framework of project 550015.

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Abstract

This report has been prepared by the SETAC Europe **S**cientific **T**ask **G**roup on **G**lobal **A**nd **R**egiona**L** **I**mpact **C**ategories (SETAC-Europe/STG-GARLIC) that is installed by the 2nd SETAC Europe working group on life cycle impact assessment (WIA-2). This document is background to a chapter written by the same authors under the title “Climate change, stratospheric ozone depletion, photo-oxidant formation, acidification and eutrophication” in Udo de Haes et al. (2002). The chapter summarises the work of the STG-GARLIC and aims to give a state-of-the-art review of the best available practice(s) regarding category indicators and lists of concomitant characterisation factors for climate change, stratospheric ozone depletion, photo-oxidant formation, acidification, and aquatic and terrestrial eutrophication. Backgrounds on each of the specific impact categories are given in another background report from Klöpffer and Potting (2001).

This background report provides details on a selection of general issues relevant in relation to LCA and characterisation of impact in LCA. The document starts with a short introduction of the LCA methodology and impact assessment in LCA for non LCA-experts. LCA experts, on the other hand, will usually not be familiar in-depth with scientific and political backgrounds of the specific impact categories. A review of this is given. Also the discussion is provided about the issue of the position of the category indicator in the causality chain, and into the related issue of spatial differentiation. These two issues appeared to be one of the core items for SETAC-Europe/STG-GARLIC.

Preface

Methods like integrated modelling of the chain from cause to environmental effect are of growing importance for the support of European Environmental policy. RIVM explores the potential of broadening the basis of such integrated environmental assessment methods with knowledge and conventions applied in Life Cycle Assessment and Substance Flow Analysis in close collaboration with the Center for Environmental Science of the University of Leiden, and the dept. of Science, Technology and Society of Utrecht University, and SETAC1. We are therefore happy to publish this document as a RIVM report.

This document is prepared by SETAC's Europe Scientific Task Group on Global And RegionaL Impact Categories (SETAC-Europe/STG-GARLIC) that is installed by the second SETAC Europe working group on life cycle impact assessment (WIA-2). This working group has adopted as a priority aim to establish best available practice(s) regarding impact categories, category indicators, and equivalency factors to be used in impact in Life Cycle Assessment. Scientific Task Groups are formed around groups of impact categories to start this process. SETAC-Europe/STG-GARLIC deals with acidification, aquatic and terrestrial eutrophication, tropospheric ozone formation, stratospheric ozone depletion and climate change.

The ultimate aim is to develop general indicators that integrate environmental side-effects of economic activities, which can be used in decision-making by governments, companies and consumers.

Drs. Rob Maas
(Head of the Environmental Assessment Bureau of RIVM)

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[1] SETAC is the acronym for Society of Environmental Toxicology and Chemistry.

Contents

Samenvatting

Levenscyclus analyse is een instrument om de milieuprestatie van producten en service systemen te evalueren. LCA methodologie is gebruikt en heeft zich continue ontwikkeld vanaf eind jaren zestig in de vorige eeuw. Gedurende de negentiger jaren zijn LCA gerelateerde activiteiten zowel nationaal als internationaal sterk in omvang toegenomen. Harmonisatie en standaardisatie van LCA methodologie vindt plaats in de context van de ISO 14000 serie en ook door de activiteiten van de verschillende SETAC werkgroepen.

The Europese afdeling van SETAC heeft April 1998 de tweede SETAC Europa Werkgroep voor milieu-effectbeoordeling in LCA (WIA-2) ingesteld. De WIA-2 heeft zich tot doel gesteld om de best beschikbare methoden voor default gebruik in LCA te identificeren voor wat betreft milieu-effect categorieën, effect indicatoren en bijbehorende lijsten met karakterisatie factoren. Wetenschappelijke taakgroepen zijn geformeerd rond groepen van milieu-effect categorieën. De auteurs van dit document representeren de taakgroep voor mondiale en regionale milieuproblemen, de zogeheten SETAC-Europe/STG-GARLIC. De milieu-effect categorieën omvatten klimaat verandering, stratosferische ozon afbraak, formatie van foto-oxidanten, verzuring en aquatische en terrestrische vermesting.

Dit document vormt de achtergrond bij een hoofdstuk met de titel “Climate change, stratospheric ozone depletion, acidification, photo-oxidant formation and eutrophication” in Udo de Haes et al. (2002). Dit hoofdstuk omvat het werk van de SETAC-Europe/STG-GARLIC en beoogd een overzicht stand van zaken met betrekking tot best beschikbare methoden voor milieu-effect categorieën, effect indicatoren en bijbehorende karakterisatie factoren voor default gebruik in LCA. Achtergronden met betrekking tot elk van de specifieke milieu-effect categorieën worden gegeven in een ander achtergrond rapport. Dit achtergrond rapport gaat dieper in op een selectie van algemene onderwerpen relevant in relatie tot LCA en milieu-effectbeoordeling in LCA.

Hoofdstuk 2 geeft een kort overzicht van milieu-effectbeoordeling in LCA . SETAC’s “code of practice” en de recente internationale standaarden in de ISO 14000 serie zijn breed geaccepteerd als algemeen raamwerk voor LCA:

- ISO EN 14040 (1997) on principles and framework,
- ISO EN 14041 (1998) on goal and scope definition and inventory analysis,
- ISO EN 14042 (2000) on life cycle impact assessment, and
- ISO EN 14043 (2000) on life cycle interpretation.

Deze publicaties geven LCA gebruikers geen gedetailleerd methodologisch overzicht of concrete handvatten voor milieu-effect beoordeling in LCA (ISO 14042).

Hoofdstuk 3 gaat in op de mogelijkheden om de milieu-relevantie van de milieu-effect beoordeling in LCA te verbeteren. De milieu-relevantie van ruimtelijke differentiatie neemt toe door een aanzienlijke afname van de onzekerheid in de resultaten van de effect karakterisatie als de indicator verder in de oorzaak-gevolg keten is gedefinieerd. Reductie van de onzekerheid door plaats-generieke karakterisatie verder in de oorzaak-gevolg keten (zonder ruimtelijke differentiatie dus) is echter relatief klein. Lokatie-afhankelijke karakterisatie in LCA vraagt als extra informatie een grove indicatie van het land waar een proces en zijn emissies plaatsvinden. Deze informatie is meestal al beschikbaar van de doelbepaling en/of de inventarisatie. We bevelen daarom een ruimtelijk gedifferentieerde karakterisatie in LCA aan. De gebruiker kan desalniettemin reden hebben om af te zien van ruimtelijke differentiatie in LCA. In dat geval wordt aanbevolen om de onzekerheden als gevolg hiervan te kwantificeren ten behoeve van de beleidsmaker en als attendering dat dit kan leiden tot mogelijke foute optimalisaties.

Mondiale en regionale luchtverontreinigingsproblemen zijn de afgelopen twee decennia onderwerp van wetenschap en politiek. De geobserveerde effecten veroorzaakt door lange afstandstransport van verontreinigingen vormen de basis voor internationale samenwerking in het analyseren van de problemen en het formuleren van oplossingen. Hoofdstuk 4 beoogt een overzicht te geven van de staat waarin het milieu verkeert voor wat betreft de milieu-effect categorieën hier in relatie tot internationale onderzoek- en beleidsactiviteiten. Het overzicht is incomplete voor de regionale milieu-effect categorieën, hetgeen – behalve dat de betrokken auteurs overwegend uit Europa afkomstig zijn – ook illustreert dat deze problemen van wisselend belang zijn voor de verschillende continenten. Dit, tezamen met culturele verschillen in probleemoplossing en milieu-management zouden een verklaring kunnen zijn voor het verschil in sophisticatie in regionale modellen om deze milieu-effect categorieën te karakteriseren.

Summary

Life cycle assessment (LCA) is a tool to evaluate the environmental performance of product and service systems. LCA methodology has been practised and continually developed since the late 1960's. During the 1990's, LCA-related activity has greatly intensified. Harmonisation and standardisation of LCA methodology takes place within the context of the ISO 14000 series as well as through the activities of the several SETAC working groups.

The European branch of SETAC started in April 1998 the second SETAC Europe working group on life cycle impact assessment (WIA-2). The WIA-2 has adopted as a priority aim to establish best available practice(s) regarding impact categories, together with category indicators, and lists of concomitant characterisation factors to be default used in LCIA. Scientific Task Groups are formed around groups of impact categories to start this process. The authors of this document represent the SETAC-Europe/STG-GARLIC and deal with climate change, stratospheric ozone depletion, groundlevel ozone formation or photo-oxidant formation, acidification, and eutrophication.

This document is background to a chapter with the title "Climate change, stratospheric ozone depletion, acidification, photo-oxidant formation and eutrophication" in Udo de Haes et al. (2002). The chapter summarises the work of the SETAC-Europe/STG-GARLIC. It aims to give a state-of-the-art review of the best available practice(s) regarding category indicators and lists of concomitant characterisation factors for climate change, stratospheric ozone depletion, photo-oxidant formation, acidification, and aquatic and terrestrial eutrophication. Backgrounds on each of the specific impact categories are given in another background report from Klöpffer and Potting (2001). This background report provides details on a selection of general issues relevant in relation to LCA and characterisation of impact in LCA.

Chapter 2 gives a short review of LCA and impact characterisation as part of that. SETACs "Code of practice" (Consoli et al. 1993), and the recent international standards and draft standards in the ISO 14000 series are widely accepted as the general framework for life cycle assessment (LCA):

- ISO EN 14040 (1997) on principles and framework,
- ISO EN 14041 (1998) on goal and scope definition and inventory analysis,
- ISO EN 14042 (2000) on life cycle impact assessment, and
- ISO EN 14043 (2000) on life cycle interpretation.

These publications do not provide practitioners with detailed methodological guidance or concrete tools for the actual performance of life cycle impact assessment (ISO EN 14042).

Chapter 3 discusses possibilities to increase environmental relevance of impact assessment as presently typical in LCA. The environmental relevance gained by spatial differentiation increases by a considerable reduction of uncertainty in characterisation results as the indicator is defined further along the causality chain. Reduction of uncertainty from site-generic characterisation further along the causality chain as such is on the other hand relative small. Site-dependent characterisation in LCA needs only a rough indication of the region where a process ant its emission takes place. This information is often readily available from goal & scoping and inventory analysis. We therefore recommend a spatial resolved impact characterisation in LCA. The LCA practitioner may nevertheless have a number of reasons to refrain from spatial differentiation in LCA. Quantification of the uncertainty by refraining from spatial resolved characterisation is recommended in that case to facilitate the decision-maker and to raise awareness of possible false optimisations as result of that.

Global and regional air-pollution problems have been topic of scientific and political concern in the world over the last two decades. The observed effects caused by long-range transport of pollutants from transboundary sources formed the basis for international environmental co-operation in analysing the problem and formulating solutions. Chapter 4 attempts to give a global overview of the state of the environment for the impact categories covered here in relation to the relevant international research and policy activities. The incompleteness of this review with regard to the non-global impacts basically illustrates – except that the involved authors were predominantly from Europe – that these categories also have varying degrees of importance on the several continents. This, together with cultural differences in problem solving and environmental management may explain deviating levels of sophistication of regional models to characterise the several impacts.

1 Introduction

(based on G. Norris 1998, J. Potting 2000, U. de Haes et al. 1999)

Life cycle assessment (LCA) is a tool to evaluate the environmental performance of product and service systems. LCA focuses on the entire life cycle of a product: from the extraction of resources and processing of raw materials, through the manufacture, distribution and use of the product, to the final processing of the disposed product. Through all these stages, extraction and consumption of resources (including energy) and releases to air, water and soil are identified and quantified. Subsequently, the potential contribution of these resource extractions and environmental releases to several important types of environmental impact (impact categories in LCA terms) are assessed and evaluated.

LCA methodology has been practised and continually developed since the late 1960's. During the 1990's LCA-related activity has greatly intensified, in terms of efforts to:

- Advance LCA methodology,
- Standardise LCA practice,
- Develop databases and software capabilities,
- Apply LCA in product design, improvement, and marketing, and
- Apply LCA in developing environmental policies.

Harmonisation and standardisation of LCA methodology takes place within the context of the ISO[2] 14000 series as well as through the activities of the SETAC[3] working groups. Whereas ISO focuses on standardisation and harmonisation of current practice, her recommendations remain on a general level and do not cover the choice of specific methodologies. SETAC aims at taking further forward a coherent scientific development of LCA methodology.

The European branch of SETAC started in April 1998 the second SETAC Europe working group on life cycle impact assessment (WIA-2). The WIA-2 has adopted as a priority aim to establish best available practice(s) regarding impact categories, together with category indicators, and lists of concomitant equivalency factors to be default used in life cycle impact assessment (LCIA). Scientific Task Groups are formed around groups of impact categories to start this process. One of these Groups deals with climate change, stratoospheric ozone depleteion, photo-oxidant formation, acidification, aquatic and terrestrial eutrophication,. (SETAC-Europe/STG-GARLIC) [4]

This document is background to a chapter written by Potting et al. in Udo de Haes et al. (2002). The chapter summarises the work of the SETAC-Europe/STG-GARLIC and aims to

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[2] ISO is the acronym for International Standard Organisation.

[3] SETAC is the acronym for Society of Environmental Toxicology and Chemistry.

give a state-of-the-art review of the best available practice(s) regarding category indicators and lists of concomitant characterisation factors for climate change, stratospheric ozone depletion, photo-oxidant formation, acidification, and aquatic and terrestrial eutrophication. Backgrounds on each of the specific impact categories are given in another background report from Klöpffer and Potting (2001). This background report provides details on a selection of general issues relevant in relation to LCA and characterisation of impact in LCA.

The recommendations in the summary report have been submitted for review to LCA experts and – limited – to experts from scientific disciplines supplying to the relevant impact categories. The experts from scientific disciplines supplying to the relevant impact categories will in general not be familiar in-depth with LCA. Therefore, this document starts with a short introduction of the LCA methodology and impact assessment in LCA (Chapter 2). LCA experts, on the other hand, will in general not be familiar in-depth with scientific and political backgrounds of the specific impact categories that is provided in Chapter 4. Chapter 3 goes into the issue of the position of the category indicator in the causality chain, and into the related issue of spatial differentiation. These two issues appeared to be one of the core items for SETAC-Europe/STG-GARLIC.

The present document will take its starting point in the Background document for the Second Working Group on Life Cycle Impact Assessment of SETAC-Europe (WIA-2) (Udo de Haes et al. 1999) and the earlier report from Nichols et al. (1996) written in the context of the first SETAC-Europe working group.

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[4] STG-GARLIC is the acronym for **S**cientific **T**ask **G**roup on **G**lobal **A**nd Regiona**L** **I**mpact **C**ategories

2 Life cycle assessment

2.1 Introduction

(based on G. Norris 1998 and J. Potting 2000)

SETACs “Code of practice” (Consoli et al. 1993), and the recent international standards and draft standards in the ISO 14000 series are widely accepted as the general framework for life cycle assessment (LCA):

- ISO EN 14040 (1997) on principles and framework,
- ISO EN 14041 (1998) on goal and scope definition and inventory analysis,
- ISO EN 14042 (2000) on life cycle impact assessment, and
- ISO EN 14043 (2000) on life cycle interpretation.

These publications do not provide practitioners with detailed methodological guidance or concrete tools for the actual performance of life cycle impact assessment (ISO EN 14042). Weidema (1997) gives a concise and up-to-date introduction to LCA. More comprehensive and detailed guidelines are supplied by the Danish method book (Wenzel et al. 1997), the Nordic guidelines (Lindfors et al. 1995), the Dutch LCA guide (Heijungs et al. 1992), and the North American publication with guidelines on inventory and principles (Vigon et al. 1993). Udo de Haes and Wrisberg (1997) report on a recent European review of the state of the art and research priorities for the breadth of LCA. Finally, two reviews of life Cycle Impact Assessment (LCIA) – which is the subject of this document – were published by parallel working groups of SETAC. Whereas Udo de Haes (1996) provides the European perspective on LCIA, the North American view is given in Barnthouse et al. (1997). A brief overview of the general framework is given in Section 2.2. Section 2.3 gives a description of the impact assessment phase in present LCA.

2.2 The general framework

(based on Norris 1998 and Potting 2000)

The concept of LCA originates from energy analysis in the late sixties and early seventies. Energy analysis has later on been broadened to take into account also the extraction and consumption of resources, and releases to air, water and soil. It has only recently become common practice to interpret these environmental inputs and outputs with regard to their potential to contribute to environmental impact. (Consoli et al. 1993, Fava et al. 1993, Udo de Haes et al. 1994, Weidema 1997)

SETACs “Code of practice” (Consoli et al. 1993) distinguishes four methodological phases within LCA: goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and life cycle improvement assessment. In the standard ISO EN 14040 (1997), life cycle improvement assessment is not longer regarded as a phase on its own, but rather seen as having its influence throughout the whole LCA methodology. Another phase has been added in stead: life cycle interpretation.

Figure 2.1 presents the ISO framework for LCA (ISO EN 14040 1997). An important notion of ISO EN 14040 is the iterative character of LCA. All phases may have to be passed through more than once due to new demands posed by a later phase. Though decisions and actions may follow the interpretation phase, these decisions and actions in themselves are outside the framework of LCA. However, possible “direct applications” are indicated in ISO EN 14040.

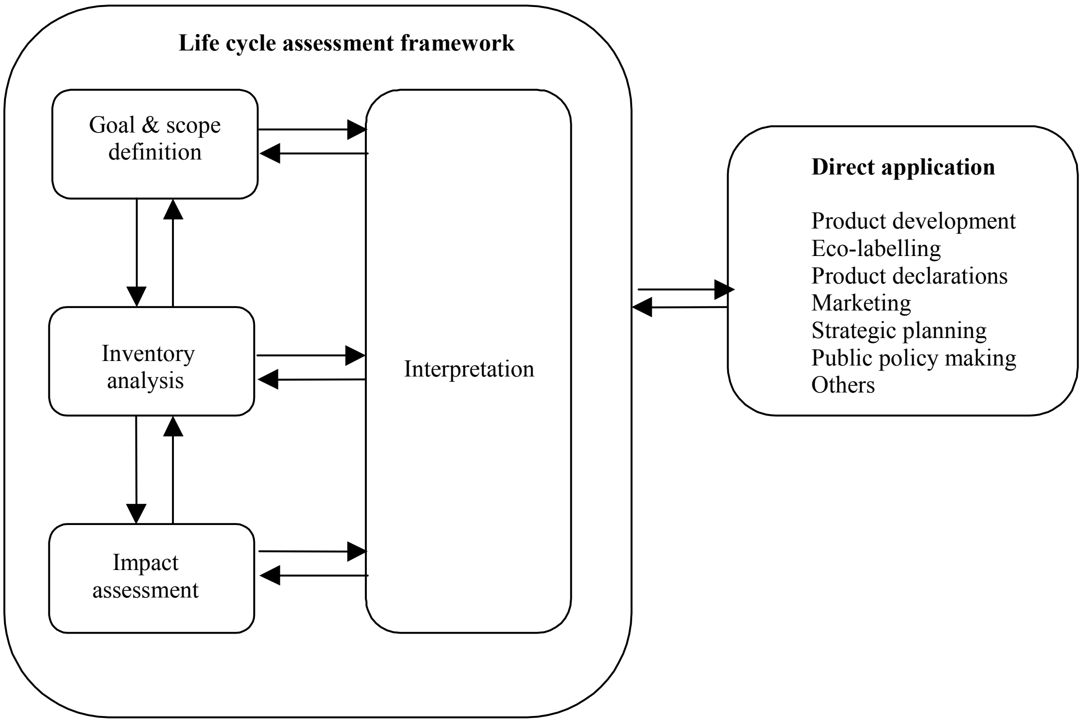

Figure 2.1: The phases of an LCA according to ISO EN 14040 (1997).

The *goal definition* clarifies the initial reasons, the intended application and the audience of the LCA. The main applications supported by LCA usually ask for a comparative assertion. That means, either comparison of different products that are functionally equivalent, or comparison of the processes within the life cycle of one product.

The *scope definition* specifies the object of the LCA and directs the specific methodology to be followed in the next phases. It also defines the basis on which the relevant products are compared. A particular product can provide different services and a given service can be provided by different products. The object studied in a LCA is therefore a product service rather than a product itself. The functional unit, a measure for the service performance of a product, ensures that the comparison is made on a common basis. The methodological choices about boundaries and procedures for the other phases are according to ISO EN 14040 specified in scope definition.

Inventory analysis identifies and quantifies the resource extractions and consumptions, and the releases to the environment relating to the processes that make up the life cycle of the examined product(s). These environmental inputs and outputs are expressed as quantities per functional unit (and do not contain a specification of the temporal and spatial characteristics of these).

The life cycle of a product consists roughly of four stages. Figure 2.2 gives a schematic overview of the life cycle of an arbitrary product. However, each stage may consist of a number of processes which each uses one or more inputs from previous processes and gives outputs to one or more next processes. Consumption (and the preceding production) of energy, and transportation processes do take place in practically all stages.

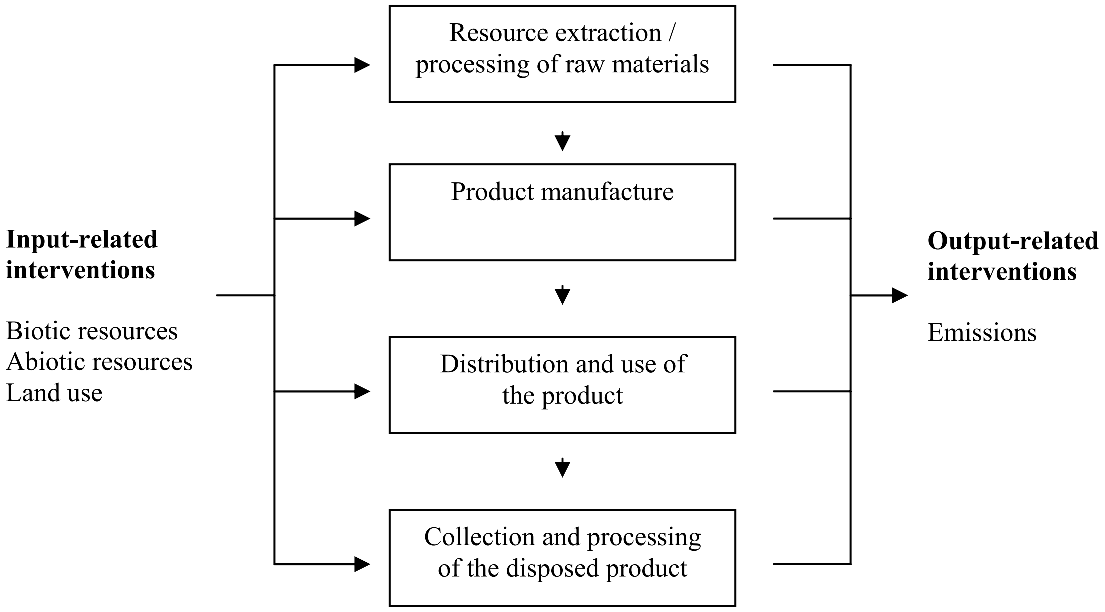

Figure 2.2: Schematic overview of the life cycle of an arbitrary product and its inputs to, and outputs from the environment.

Each input to a process can be followed upstream to its origin (“cradle”) and each output to a process downstream to its final end (“grave”). The total of connected processes is called the product system, process tree or life cycle. It is easy to imagine that a product system can become rather complex as a product consists of more than one material or component. Even one material or component may represent a complex subsystem, however: A material like polyamide can be synthesised by many, very different technologies, and several producers can apply each technology.

The system boundaries determine which processes will be included in the LCA. The definition of the product system and its boundaries takes, according to ISO EN 14040 and 14041, place in scope definition. Scope definition decides also for which environmental inputs and outputs data should be collected, and about the procedure to allocate these to processes with multiple outputs to next processes. The inventory phase therefore only consists of the actual data collection and data processing. Inventory analysis results in an inventory table where environmental inputs and outputs are listed and are summed-up per type of intervention. A sector-specific aggregation of the inputs and outputs can be performed in addition, indicating quantitatively in which stages of the life cycle the interventions occur or, the other way round, this information should not be lost during aggregation to the final inventory table.

Environmental inputs and outputs have the potential to bring about several kinds of impact on the environment (impact categories in LCA terms). *Life cycle impact assessment* estimates the potential contributions from these inputs and outputs to a number of impact categories and may choose to continue by weighting across impact categories in order to reduce the collection of impact categories into one measure.

As a first step, impact categories and category indicators should be selected together with the models to quantify the contribution to the selected impact categories and category indicators. Once these selections have been made, the environmental inputs and outputs are assigned to the impact categories selected in scope definition (classification). The contribution to an impact category from each input or output is then next modelled (characterisation). In very specific cases and only when meaningful, the modelling results for one impact category or subcategory may be aggregated with those of another one (grouping and weighting). The next section provides some more information about the impact assessment phase.

Life cycle interpretation is the phase in which the results of the inventory phase and the impact assessment phase are combined in line with the defined goal and scope. Conclusions and recommendations to the decision-maker may be drawn, unless reviewing and revising of previous phases is needed. Both concluding/recommending and reviewing/revising should preferably be based on uncertainty and sensitivity analysis.

2.3 The impact assessment phase

According to ISO EN 14042 (2000), life cycle impact assessment consists of four mandatory elements and two optional elements:

- Selection of impact categories, category indicators and characterisation models (mandatory),
- Assignment of results from the inventory phase to the selected impact categories (classification; mandatory),
- Calculation of category indicator results with help of selected models (characterisation; mandatory),
- Calculating the magnitude of category indicator results relative to reference values(s) (normalisation; optional),
- Grouping or weighting (optional, weighting not allowed in comparative assertion disclosed to the public), and
- Data quality analysis (mandatory in comparative assertion).

Udo de Haes (1996) provides a default list of impact categories in LCA (see Table 2.1). The list is not meant as a minimum and neither as a maximum list. The impact categories from this list that are relevant in this report are climate change, stratospheric ozone depletion, tropospheric ozone formation, acidification and eutrophication.

According to ISO EN 14042 (2000), a category indicator is identified as the quantifiable representation of an impact category, being the object of characterisation modelling. The category indicator can be defined at any level of the causality chain or chain of environmental mechanisms within an impact category that cause an environmental intervention to have an impact on the final endpoints. An example regarding climate change may illustrate this. The environmental mechanisms connecting an intervention (e.g. a CO_2 emission) to the final impact on its endpoints (e.g. damage to flora, fauna and human beings) may be:

> Emission of CO_2 → increased radiative forcing → climate change (i.e., average global temperature rise, other climate changes) → rise of sea level → flooding of land → damage to ecosystems and human beings (e.g., expressed as years of human life lost).

The category indicator for climate change is typically the increased radiative forcing and this radiative forcing is typically quantified with help of the Global Warming Potentials (GWP) as reported by the Intergovernmental Panel on Climate Change (Houghton 1996).

Table 2.2 lists the category indicators and models presently typically *selected* for the other impact categories being subject of this report. Presently typical characterisation in LCA uses equivalency or characterisation factors[5].

Table 2.1: Default list of impact categories to be characterised in LCA (after Udo de Haes 1996)

Input related categories (Resource depletion or competition)	**Output related categories** ("Pollution")	**Pro memory** (Flows not followed up to system boundary)
Abiotic resources Biotic resources Land	Climate change Stratospheric ozone depletion Human toxicity Eco-toxicity Tropospheric ozone formation Acidification Eutrophication Odour Noise Radiation Casualties	Input related: energy, materials, plantation wood etc.) Output related: solid waste etc.

Classification is the second mandatory element of the impact assessment phase and consists of assigning data from inventory analysis to the relevant impact categories. One intervention may well be included in more than one impact category. An emission of NO_x, for example, contributes to acidification as well as to eutrophication.

Characterisation quantifies the contribution of an intervention to the relevant impact categories. The category indicator - like the increased radiative forcing in climate change - is the quantifiable representation of an impact category. Characterisation factors - like the global warming potentials for climate change – are used to convert the environmental inputs and outputs assigned to a given impact category into their contribution to that impact category at the level of the category indicator. The characterisation results in the contributions to a number of impact categories, also indicates as the environmental profile of a product. Table 2.2 lists the presently typical characterisation factors related to the impact categories and category indicators being subject of this report.

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[5] Equivalency or characterisation factors represent models that establishing linear relations between emissions and their impact on the environment at the level in the causality chain where the category indicator is defined. Such factors are simple in use, which does not necessarily mean they inaccurately characterise impact as these factors can be derived from underlying sophisticated models (i.e., they are meta-models of these sophisticated models).

Table 2.2: Category indicators and characterisation factors for the impact categories climate change, stratospheric ozone depletion, tropospheric ozone formation, acidification and eutrophication (aquatic) as presently typical used in LCIA.

Impact category:	**Climate change.**
Category indicator:	Increased radiative forcing.
Characterisation factors:	Global warming potentials (GWP) according to IPCC (Houghton 1996).
Comment:	Global warming potentials (GWP) quantify the potential of a substance to increase radiative forcing relative to that of CO_2. IPCC provides factors for three time horizons (20, 100 and 500 years). All three may be included, but a presently typical LCIA works with the horizon of 100 years.
Impact category:	**Stratospheric ozone depletion.**
Category indicator:	Breakdown of ozone.
Characterisation factors:	Ozone depletion potentials (ODP) according to WMO (WMO 1994&1995, Solomon and Albritton et al. 1992).
Comment:	The ozone depletion potentials (ODP) quantify the breakdown of ozone relative to CFC-11. The factors provided by WMO (1994 and 1995) describe the impact of an emission integrated over an infinitive time span. A typical LCIA uses these factors. However, WMO (1995) and also Nichols (1996) advise to use factors integrated over a shorter time-span. Solomon and Albriton (1992) provide these factors.
Impact category:	**Photo-oxidant formation**
Category indicator:	Groundlevel ozone formation.
Characterisation factors:	Photochemical ozone creation potentials (POCP).
Comment:	Photochemical ozone creation potentials (POCP) typically used n LCIA do only provide factors for volatile organic compounds and do not take into account background concentrations. However, the formation of ozone from volatile organic compounds is highly dependent of the background concentrations of NO_x. Some LCA manuals therefore recommend working with potentials for high background concentrations of NO_x (Derwent et al. 1998), and potentials for low background concentrations of NO_x (Andersson-Sköld et al. 1992). In the USS, the Maximum Incremental Reactivity (MIR) scale from Carter (199?) and based on laboratory testing of model atmospheres representing the average urban air in US cities. MIR estimates the worst-case production of ozone, and therefore is valid for a wide range of locations.
Impact category:	**Acidification**
Category indicator:	Release of hydrogen potentials.
Characterisation factors:	Potential to release hydrogen ions (acidification potential or AP).
Comment:	The potential of a substance to release hydrogen ions (acidification potential or AP) is usually expressed relative to SO_2 based on chemical stoichiometry.
Impact category:	**Eutrophication** (aquatic).
Category indicator:	Growth of (aquatic) biomass.
Characterisation factors:	Share in average composition of algae (Stumm and Morgan 1981, Redfield, (1942). Eutrophication or nutrification potential (NP's)
Comment:	The share of a substance in the average composition of algae can be expressed relative to phosphor, nitrogen or oxygen. All measures are used. This category indicator typically quantifies aquatic eutrophication, but is only little relevant to quantify terrestrial eutrophication. Only Lindfors et al. (1995) provides a procedure to characterise terrestrial eutrophication by simply adding together all airborne nitrogen emissions.

The optional step *"normalisation"* relates the results obtained in the characterisation step to the total indicator result of a country, region or the world. The ratios thus obtained for the different categories provide an indication for the relative importance of the categories in the context of a specific LCA study, if the same reference area is taken for all categories. This constitutes, of course, a kind of weighting but it does not involve the societal values necessary in weighting in the more narrow sense used in ISO 14042.

The optional step *"weighting"* tries to establish an hierarchy of the categories, either within the framework of a specific LCA study, or in general. Inevitably, weighting requires societal values and, thus, cannot be done with the methodology of the (exact) sciences. Weighting, although excluded by ISO 14042 for use in comparative assertions to be disseminated in the public, is often regarded necessary in comparative LCAs, since the results of the impact assessment are in most cases not unambiguous, showing bad ratings in some categories and good ones in others.

2.4 Focus of the work

According to ISO EN 14042 (2000), life cycle impact assessment consists of four mandatory elements (selection, classification, characterisation and data quality analysis) and two optional elements (normalisation and grouping & weighting). The elements of normalisation, and grouping & weighting can thus follow upon characterisation but are not mandatory. It is left optional whether or not to perform these elements, though weighting is not allowed in comparative assertion disclosed to the public.

Normalisation and grouping and weighting have been subject of intensive debate in the ISO-process because of the subjectivity involved in performing these elements. An important group of LCA practitioners is reluctant towards weighting in particular, but even as many practitioners consider weighting an unavoidable element since no explicit weighting leads to implicit weighting. Weighting and/or aggregating the different category indicators is usually necessary to compare product systems or evaluate improvement options, since one alternative hardly ever scores better on all category indicators than the other alternative.

Some practitioners attempt to minimise the subjectivity from weighting (i.e. to partly avoid weighting) by choosing their category indicators further along the causality chain, close to or at the endpoint point and expressing the impact on the endpoint in the same unit (facilitating aggregation of indicators without weighting). This reduces the number of category indicators to be weighted and aggregated since the number of endpoints is limited and several impact categories therefore share their indicator. However, an important question is whether the state-of-the art does yet allow to model up to the endpoint. This is subject of the next chapter.

The focus of SETAC-Europe/STG-GARLIC is on best available practice(s) in characterisation. Best available practice in characterisation is not necessarily best available practice in weighting. These are two different optimisations that at the present-state-of-the-art may prompt to acceptance of “less” than best available practice in one of the two elements. That is, prioritising easiness of weighting by choosing category indicators with the same unit (i.e. avoiding weighting) may require “less” than best available practice in characterisation (and the other way around).

The choice to optimise either towards characterisation or to weighting by modelling further along the causality chain depends on the application and context of the given LCA, but there is thus a tension between best available practice(s) in characterisation and best available practice(s) in weighting in LCA. The next chapter touches upon this tension, but does not further address the issue of weighting since another Scientific Task Group deals with this element and also with grouping and normalisation.

3 Sophistication of category indicators

3.1 Introduction

The impact assessment phase in LCA is relatively young and emerged from the wish to simplify the large amount of data from inventory analysis by aggregation into a manageable number of impact indicators. This wish has in the course of time been extended with the wish to increase the environmental relevance of the impact assessment phase in LCA.

Discussions about the environmental relevance of impact assessment in LCA has long been dominated by two closely connected issues: evaluation of threshold exceedance (Section 3.2) and spatial (and temporal) differentiation (Section 3.3) in characterisation of impact. These issues and thus this chapter are particularly relevant for the non-global impact categories.

The discussion about whether or not to perform evaluation of threshold exceedance has last few years shifted towards the question whether characterisation should go beyond threshold evaluation (risk analysis) and focus on quantification of subsequent damage. In other words, at which level within the causality chain (see also Section 2.3) should category indicators preferably be defined in LCA: somewhere in between emissions and endpoint (a so-called mid-point indicator), or at the endpoint (so-called endpoint indicator).

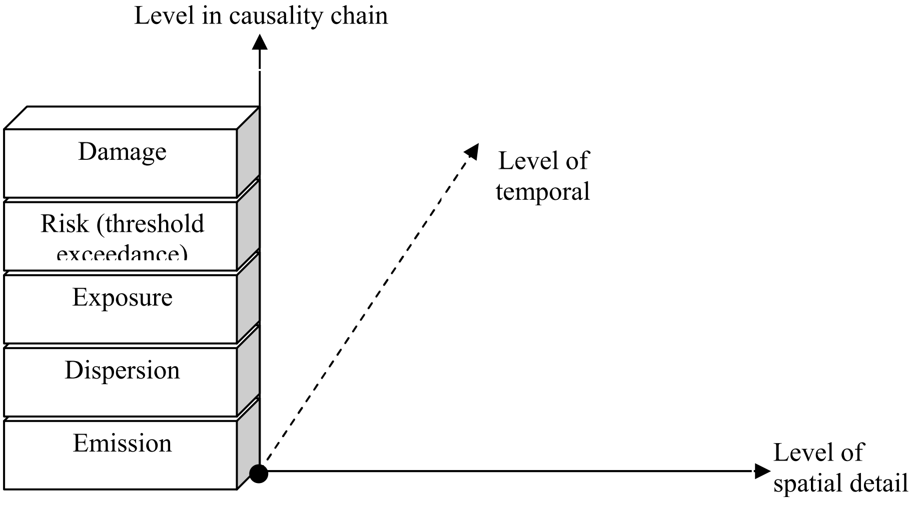

Figure 3.1: Levels of possible spatial and temporal sophistication in LCA and levels at which the category indicator can be defined.

The level at which category indicators are chosen is one form of sophistication. Another form of sophistication is the extent to which spatial and temporal resolution is taken account in the characterisation modelling up to the chosen level in the causality chain. The chosen level of spatial and temporal differentiation can thus be seen as a sophistication perpendicular on the chosen level in the causality chain at with the category indicator is defined (see Figure 3.1). This also presages the close relation between chosen level of spatial and temporal resolution on the one hand and the level in the causality chain at which the category indicator is defined on the other hand as will be further clarified in this chapter.

3.2 Position of category indicators

3.2.1 The threshold issue

The first generation characterisation methods took their basis in equivalency assessment with help of characterisation factors that were derived from intrinsic substance characteristics like the potential to release hydrogen ions (for acidification assessment), toxic effect-levels (for toxicity assessment)[6] or the ability to radiative forcing (for climate change). Modelling of exposure and assessment of threshold exceedance (i.e. PEC/PNEC ≥ 1)[7] were not performed since the available data did not allow such evaluation.

Threshold information, usually in the form of a no-effect-level, was in the first generation characterisation models thus used only in toxicity assessment to express the emission of a given substance as a dilution volume of the receiving environment. The basis of equivalency was taken in the toxicity potential of each substance by setting to one for the impact from an emission quantity equal to the no-effect-level[8], and the impact from any deviating quantity as the ratio of the emission quantity divided by the no-effect-level.

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[6] No-effect-concentrations are based on experiments on test-species under laboratory conditions and therefore say something about the intrinsic substance characteristic to cause toxic effect (rather than something about the sensitivity of a species in real-life for this toxic substance).

[7] “PEC” stands for Predicted Environmental Concentration, and “PNEC” for Predicted No Effect Concentration.

[8] The underlying assumption is that the toxicity impact from a quantity at the no-effect-level of one substance has the same importance as the toxicity impact from a quantity at the no-effect-level of another substance. To put it more clearly: If the quantities of both substances are at their no-effect-level, the impacts from a neuro-toxic substance and a skin irritating substance are regarded as equally important. The adding together of very different effect types is one of the more serious problems in LCA, but is not further addressed in this document.

Present toxicity factors now also cover exposure modelling that quantifies the change in exposure from an emission (rather than the total exposure resulting from the emission). The aggregation of the calculated exposure increases from different substances is still often based on no-effect-levels. Toxicity impact is now defined as the change in predicted environmental concentration (i.e., change in exposure) divided by the – predicted – no-effect-concentration (ΔPEC/PNEC).

Evaluation of threshold exceedance is still not covered by the present generation of toxicity factors, however, and neither feasible in the context of characterisation of toxicity. Such evaluation needs information about operative background exposures on top of which an exposure increase takes place. There are simply too many potential toxic substances that make such information yet hard to achieve within the present state-of-the-art science in toxicity assessment.

Though evaluation of threshold exceedance was initially not performed due to lack of data, it has meanwhile turned for some practitioners into a principle in itself that is justified by the reasoning that “less pollution is better”. An important other group of practitioners has always been advocating an “only above threshold” approach on the other hand. An impact assessment following an “only above threshold” approach accounts only for those emissions leading to exposure levels above threshold.

The discussion whether to perform evaluation of threshold exceedance in LCA is topical for already quite some years now but has mainly taken place in relation to toxicity assessment. However, recent publications of Potting et al. (1998), Pleijel et al. (not published), Lindfors et al. (1998), Huijbregts et al. (2000), Hauschild and Potting (2003), Potting and Hauschild (2003) and Krewitt et al. (2001) show evaluation of threshold exceedance to be quite well possible for the impact categories photo-oxidant formation, acidification and terrestrial eutrophication with help of characterisation factors simple to use in LCA. These characterisation factors are derived from sophisticated underlying models and can thus be seen as meta-models summarising the sophisticated modelling in a linear relation between emissions and their impact at the level in the causality chain where the indicator is defined. These publications actually also show that more approaches are possible than “less pollution is better” and “only above threshold” only. Few of them are described in the following section.

3.2.2 Threshold based category indicators

In a *“less pollution is better”* approach, all emissions are considered to be relevant on the basis of their intrinsic harmful properties. The assumption initially underlying the “less pollution is better” approach is that all emissions give rise to exposure increases at sites that have a similar sensitivity to the given substance. Recent work of Huijbregts et al. (2000)

takes a somewhat more sophisticated approach. Huijbregts et al. used the RAINS-model[9] to calculate characterisation factors for acidification and terrestrial eutrophication that do allow ecosystems to have different a priori tolerances for exposure increase (each ecosystem has its own critical load)[10], but does not take into account differences in background exposure. His definition of the category indicator is similar as to the one that – by necessity – is typically used in toxicity assessment (see also Section 3.2.1). Impact on the individual ecosystem is defined by Huijbregts et al. as the predicted exposure increase divided by the critical load of the given ecosystem[11,12]. This results in the impact from an exposure increase on an ecosystem exposed just below its critical load (and thus about to be in danger) regarded equally important as an ecosystem facing the same exposure increase but already exposed far above its critical load (and thus difficult to rescue). Similarly, the impact from exposure increases on these ecosystems are regarded equally important as an ecosystem facing the same exposure increase but still exposed far below its critical load (and thus hardly in danger). Whether ecosystems are thus exposed far below or at or far above their critical loads, they are all characterised as equally vulnerable by Huijbregts et al.. An ecosystem with a high critical load is on the other hand taken as more vulnerable than an ecosystem facing the same exposure increase but having a low critical load.

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[9] The RAINS model provided the model calculations being at the basis of the several protocols under the CLRTAP. It is an 'easy-to-use' computer tool that combines spatial resolved information on regional emission levels with information on long-range atmospheric transport to estimate patterns of deposition and concentration for comparison with critical loads and thresholds for acidification, terrestrial eutrophication and tropospheric ozone formation in Europe. See Chapter 5 for more information.

[10] So are the Scottish moors very intolerant towards acidifying loading in contrast with the ecosystems on the rather insensitive lime soils in South Europe.

[11] As a matter of fact, the a priori tolerance of ecosystems for acidifying loading is described by critical load *functions* (rather than by one unique value per substance). The critical load function gives all combinations of sulphur and nitrogen deposition above which an ecosystem is at risk to be damaged (see Posch et al. 1995). For the sake of clarity, the term “critical load” is used in this paper in stead of the correct term “critical load function”.

[12] The non-existence of a unique critical load for given acidifying substances (see previous footnote) complicates the approach of Huijbregts et al. (since the denominator is not a fixed value, but is depending on the loading of other acidifying substances). Huijbregts et al. solves this by using the maximum critical sulphur and nitrogen load. Another possibility would have been to work with the critical sulphur and nitrogen loads as uniquely defined by the exceedance function (i.e., the point that provides the “shortest” distance to exceedance of the critical load function; see Posch et al. 1999). Such approach would be in accordance with the work done under the UNECE convention on long range transboundary air pollution (see also Chapter 3), and also facilitates to address exceedance of the critical loads in a consistent and comparable way (see also Section 5.5). The characterisation factors of Lindfors et al. (1998) and Potting et al. (1998) are both based on critical load *functions*.

An *"only above threshold"* approach considers only those emissions relevant that result in exposure increases at sites where critical loads are already exceeded. Impact on the individual ecosystem is valued to be one for any exposure increase on top of an exposure already being above the critical load of the given ecosystem. An ecosystem exposed far above its critical load (and thus difficult to rescue) is considered to be equally vulnerable as an ecosystem exposed just above its critical loads (and thus relatively easy to rescue). The impact from an exposure increase is on the other hand valued as zero for all ecosystems being exposed below their critical loads. This applies to ecosystems exposed far below their critical loads (and thus hardly in danger) as well as to ecosystems exposed just below their critical loads and thus in danger to become above their critical load by a small exposure increase.[13] "Only above threshold" approaches nevertheless take into account the a priori tolerances and differences in background exposures of ecosystems in order to evaluate whether the critical loads of the given ecosystems are already exceeded. Pleijel et al. proposed, and Lindfors et al. (1998) subsequently established "only above threshold" factors[14] for acidification and terrestrial eutrophication with help of models parental to the RAINS-model.

Potting et al. (1998) introduced an approach alternative to the two previous ones and that could be called *"only around threshold"*. This approach gives priority to ecosystems exposed around their critical load rather than to ecosystems exposed far below (and thus hardly in danger) or ecosystems exposed far above (and thus difficult to rescue). The work of Potting et al. should be evaluated against the fact that the emission of one source (i.e. one process) contributes to many ecosystems (far more than hundred thousand). Each of these ecosystems may differ in their operative background exposures, but they can also differ in a priori tolerance for exposure. Potting et al. based their characterisation factors on the slope of the curvilinear (sigmoid) dose-effect curve that is defined by the critical loads of all ecosystems to which one source contributes. The slope of this dose-effect curve is established with help of the RAINS-model in the working point that is thus determined by the operative background exposure. The resulting characterisation factors of Potting et al. quantify the area of ecosystems that becomes exposed above their critical loads as a result of the exposure increase from our functional unit. The "only around threshold" characterisation factors express the sensitivity of a "population" of exposed ecosystems for the changes in their background loading evoked by a given source[15]. The factors thus account for both differences in ecosystem a priori tolerance for loading and differences in background load. In this way,

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[13] As is clear from the before discussion, "only above threshold" factors do not characterise emissions on the basis of the change in impact that they invoke, but on the basis of the existing impact from the operative background loads. This assumes that the considered emissions do not evoke any change in that impact. This makes "only above threshold" characterisation to a so-called average approach.

[14] The factors of Pleijel et al. and Lindfors et al. are as a matter of fact no characterisation factors but modifiers to be used in combination with the hydrogen release potentials for acidification and the biomass production potentials for eutrophication (see Table 3.2 in Chapter 3). The modifier expresses what share of the emission from a given region deposits on ecosystems with background loads above their critical loads.

[15] To be useful in LCA, characterisation factors should not be too sensitive for changes in the background exposure of receptors (see also Section 5). Potting et al. (1998^{a}) showed that the "only around threshold" acidification factors are rather robust for changes in the background situation.

ecosystems are prioritised that are exposed close to their critical load (and thus about to be in danger or just above the critical load and therefore relatively easy to rescue). Hauschild and (2003) and Potting and Hauschild (2003) established "only around threshold" characterisation factors for acidification, terrestrial eutrophication and groundlevel ozone formation.

A critical load can only tell whether there is a risk on ecosystem damage, but not whether this risk actually results in damage and how large this damage is. One nevertheless expects the damage to become (asymptotically) larger with increasing exceedance of a critical load. More sophisticated would therefore be to follow a *damage approach*. Goedkoop and Spriensma (2000) go in the direction of such approach by quantifying, in the case of acidification and terrestrial eutrophication, the fraction of vascular plant species potentially disappeared as a result of a change in deposition. An advantage of this approach, and also the initial reason to develop it, is that acidification and terrestrial eutrophication can be aggregated since they are expressed in the same unit. Similarly to Potting et al., the damage factors from Goedkoop and Spriensma are calculated as the slope of the dose-damage curve established in the working point that is determined by the operative background exposure. The damage factors are calculated with help of the so-called Nature Planner. This model is comparable with the RAINS-model, but the model domain of the Nature Planner is limited to the Dutch territory, which does not allow spatial resolved impact factors[16].

3.2.3 Midpoint – endpoint modelling

The previous section discussed a number of category indicators using threshold information (or damage information in the last one). The above list of indicators is not exhaustive. More definitions are possible, like for instance so-called fate-factors from Norris in Bare et al. (2003) that quantify which share of an emission deposits on land (regardless whether on ecosystem area or otherwise). The discussed selection of indicators is summarised in Table 3.1.

An interesting feature about the category indicators in Table 3.1 is that they are defined increasingly further in the causality chain. Closer to the endpoint thus. This may in practice lead to a change in relative importance between substances assigned to an impact category. For example, the importance of nitrogen relative to sulphur decreases for category indicators defined further along the causality chain compared to an indicator defined at the begin-point (like the hydrogen release potential). This is due to the fact that deposition of nitrogen will often be assimilated by ecosystems and then does not contribute to acidification, or deposits on ecosystems not sensitive to the emission. Sort-like "corrections" in the characterisation factors are gained for other impact categories if their category indicator is defined further along the causality chain.

Defining category indicators further along the causality chain thus adds to the environmental relevance of the quantification of the selected impact categories (provided that the underlying modelling is sound). However, the additional environmental relevance remains small as long as the modelling further along the causality chain is not combined with spatial (and temporal) differentiation. This is subject of the next sections.

Table 3.1: A selection of category indicators that are available for characterisation modelling in LCA.

Type of category indicator	**Definition of category indicator** (per unit emission)	**Author**	**Comment**
Fate factors times old equivalency factor	$I(i)=\sum^{j}\Delta PEL_j*EF$ PEL is Predicted Environmental Load	Norris in Bare et al. (2003) Hauschild and Potting (2003), Huijbregts and Seppälä (2001)	For all impact categories, spatial resolved for 50 states in USA For aquatic eutrophication, spatial resolved for all European countries
Less pollution is better	$I(i)=\sum^{j}(\Delta PEL_j/CL_j)$	Huijbregts et al. (2000) Krewit et al.(2001)	For acidification and eutrophication, spatial resolved for 44 countries in Europe (Huijbregts et al. calculated a similar indicator for ecosystems already above their critical loads; thus ignoring ecosystems below their critical loads) For photochemical ozone formation, spatial resolved for Europe
Only above threshold	$I(i)=\sum^{j}(PEL_j>CL_j)$	Lindfors et al. (1998)	For acidification, spatial resolved for two Swedish regions, East England, Central Germany and Poland
Only around threshold	$I(i)=\sum^{j}(\Delta UES_j)$ UES is ecosystem area that is unprotected since its critical load is exceeded	Hauschild &Potting (2003) Krewit et al. (2001)	For all impact categories, spatial resolved for 44 countries in Europe For acidification and eutrophication, spatial resolved over Europe and South America
Damage	$I(i)=\sum^{j}(\Delta damage_j)$	Goedkoop and Spriensma (1999)	Modelling domain limited to modelling of Dutch territory

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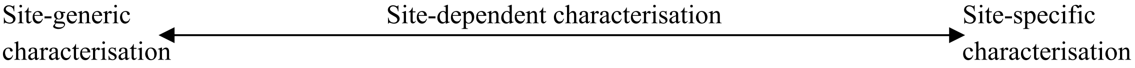
[16] Acidifying and eutrophying substances typically travel over distances of several hundreds to thousand kilometres for the relevant substances, which is far outside the Dutch territory.

3.3 Resolution aspects

3.3.1 Spatial differentiation

The long distance transport of emissions means that one has to look over several hundreds to thousands kilometres to catch most of the impact from a source. The large impact area of an emission makes the precise location of a source of less importance because the dispersion patterns and impact area of neighbouring sources overlap (i.e. show largely the same gradients). Dispersion patterns and impact area will only start to deviate considerably when sources are located at larger distances from each other. This makes it possible to establish so-called site-dependent factors that characterise impact somewhere between site-generic and site-specific and that with reasonable good accuracy estimate the impact from a source located in a given region on its receiving environment:



It is important that spatial resolved characterisation in LCA does not put unfeasible demands for additional data on inventory analysis (see also Section 3.3.3). Therefore, the resolution of site-dependent characterisation factors should be kept as low as possible. Hence that spatial resolution here refers to the size of the distinguished source regions for which site-dependent factors are established (and not to the spatial resolution of the modelling underlying those factors; this resolution can be and is typical fairly high).

Site-dependent factors for acidification, terrestrial eutrophication and tropospheric ozone formation have been established for North America and Europe (an overview is given in Table 3.1). These factors express the impact from an emission in a given region over its full impact area.

Site-dependent characterisation factors for each state in North America are established by Norris in Bare et al. (2003). These factors represent so-called fate factors that quantify the share of emissions leading to concentration or exposure increases on land surface (he does not quantify which receptors are exposed and whether they are sensitive to this).

Several sets of spatial resolved characterisation factors are available for the 44 countries forming together Europe. Most of these sets are based on different definitions of the category indicator (see Section 3.2.2). The factors of Hauschild and Potting (2003) and Guinée et al (in preparation) are derived with help of the RAINS model and the ones of Lindfors et al. (1998) with help models parental to RAINS. The factors from Krewit at al. (2001) are calculated with the EcoSense, a model comparable to RAINS. The acidification and eutrophication factors or Hauschild and Potting (2003) and Krewit et al. (2001) are based on the same definition of the category indicator.

Potting and Hauschild (2003) made a comparison between the fate-factors or the “less pollution is better” factors of Huijbregts et al. (2000), and the “only around threshold” factors of Hauschild and Potting (2003). All sets of spatial resolved acidification and eutrophication factors were calculated with help of RAINS and can be regarded as based on a category indicator defined increasingly closer to the midpoint. A number of interesting conclusions can be draw from this comparison:

- The range of values is relative small for the spatial resolved fate-factors (less than a factor 10 between lowest and highest value) but becomes increasingly larger as the category indicator is defined further along the causality chain (factor hundred for “less pollution is better” factors and a factor thousand for “only around threshold” factors between lowest and highest value). Spatial differentiation thus becomes more important if the category indicator is chosen closes to the endpoint.
- A site-generic characterisation factor for the “less pollution is better” and “only around threshold” factors, as mean value from the set of spatial resolved factors, is strongly influenced by the regions selected (e.g., EU15, EU15+Norway+Switzerland, East Europe or all 44 European countries). The standard deviation for each set is about 100% of the mean value. Site-generic characterisation by refraining from spatial differentiation thus results in a considerable uncertainty in the category indicator.
- There is a considerable variation in mean value for selections of large neighbouring regions (like East Europe and EU15+Norway+Switzerland). Site-dependent factors for small neighbouring regions (like Belgium and the Netherlands) are on the other hand quite similar. This means that characterisation factors can be resolved over rather large regions, but these regions should not be too large. About 500*500km resolution seems adequate. Such resolution will often comply with the size of countries or larger administrative regions within large countries.

3.3.2 Temporal differentiation

The small or marginal contribution from a single source to exposure of its receptors means that the time behaviour of the emission from that source (i.e. whether it is a flux or pulse) becomes less important. The temporal variation of the contribution from a single source will usually namely to a large extent be cancelled out against the high background exposure from all sources together. Exposure of receptors thus show a relative invariability in time for the contribution of single sources (this does not imply the opposite reasoning, that the temporal variations in the total exposure of receptors is unimportant).

If the exposure of receptors shows a relative invariability in time for the contribution of a single but full source, the same of course inherently holds true for the emission per functional unit. The temporal variation of emissions is thus of minor importance in characterisation modelling in LCA. This is an important observation since the lack of information about the temporal variations in emissions has long been an issue of intensive debate in LCA.

The calendar time to which the different processes in a product system relate is a more important issue when it comes to impact assessment in LCA. The calendar time of a process determines the (estimated) total economic activity with all its emissions being responsible for the total environmental load causing an impact to which background that process adds. The background situation can be rather different between calendar times (and thus between different processes in an LCA as discussed below). This could result in rather different characterisation factors related to these different calendar times (since “only above threshold” and “only around threshold” factors do account for background exposures; see Section 3.2.2). The photochemical ozone creation potential from an emission of a volatile organic compound may be different in 1990 and 2010 due to considerable differences in the background concentration levels of nitrogen oxides posed by the total economy in those years.

A product can easily cover a time frame of several decades depending on time-of-use of the product and the time needed for each subsequent process in the product system. For example, a linoleum floor covering will on average first be discarded 15 years after is has been bought (Potting and Blok 1995). The factors used to characterise the impact from a given process should relate to the calendar time in which that process takes place (a time-dependent characterisation factor thus).

As a matter of fact, also processes themselves can cover a time frame of several years to decades. A specific type of linoleum will be produced over a certain time interval before the type is taken from the market. This determines the time or calendar interval over which that type is marketed, used and disposed. Basically, the used characterisation factor should thus not relate to the calendar time, but to the calendar interval in which a given process takes place.

Similar to spatial differentiation, the level of temporal differentiation can be placed on a continuum stretched up by the two extremes time-specific and time-generic characterisation. Time-dependent characterisation has thus a level of detail somewhere between those extremes:

Time-generic characterisation ◄ Time-dependent characterisation ► Time-specific characterisation

Time-dependency has been explored by Potting and Hauschild (2003) and also by Krewit et al. (2001) by establishing “only around threshold” factors for 1990 and 2010. Trends in the site-dependent factors remain roughly the same across countries, though 2010 factors are considerable lower than 1990 factors. This is an obvious result since emissions projections between 1990 and 2010 are considerably different. Without going into details, however, trend analysis shows emission projections to be relatively stable over a couple of years (Jol and Kielland 1997). This suggests a time dependent assessment based on time-intervals of several years to be adequate for LCA. Time-dependency seems to be one of the coming issues in LCA.

3.3.3 Data availability

Application of site-dependent characterisation factors requires rough information about the geographical region where a process and its emission takes place. Similarly, a time dependent characterisation requires information about the years over which an emission is expected to take place. A major objection against site-dependent (and time-dependent) characterisation in LCA is that the demand for this data should cause unfeasible complications for inventory analysis. This has also been subject of intensive discussion between the authors of this report. Some main lines are drawn here.

Before outlining the discussion, it may be interesting to observe that unfeasible complications for inventory analysis of additional data demand are never put so prominently forwards as in relation to site-dependent characterisation in LCA. This is somewhat surprising since there are a number of recent extensions to LCA methodology that never received the same resistance, though they certainly require considerable additional data and they do complicate inventory analysis. Examples are expansion of product systems to avoid allocation as also recommended by ISO EN 14041, and the adoption of several new impact categories asking for new inventory items to be collected.

Let’s return to the issue and further focus on site-dependent characterisation (the discussion will be roughly the same for time-dependent characterisation, but no concrete methodology is yet available here). An indications of the region where a process and its emission takes place is usually readily available from goal & scoping and inventory analysis since this information is necessary to quantify transport within a product system. It also complies with meanwhile common sense between LCA practitioners that inventory analysis has to be more time specific and site-specific in order to improve the quality of inventory data (these specifications are now also demanded by the SPOLD and SPINE data formats and several LCA databases).

It is important to stress that site-dependent characterisation does not ask for *exact* information about the location of a process. It is sufficient for site-dependent characterisation, as discussed in Section 3.3.1, to identify the larger region where a process is located (countries or larger administrative regions within large countries; regions of roughly 500*500km).

While this information thus will often be available, there remain situations for which availability of this information may be troublesome:

- Processes that are supplied with materials from the spot-market or electricity from the public grid. The materials or electricity can here originate from several unknown regions. Trade between regions may change regularly, especially in case of materials from the spot market. However, it seems well feasible to generate information about average supplier composition and their regional origins over a longer period for both materials and electricity. The main supplying regions can than be used to site-dependently characterise the emissions related to production of the given materials or electricity, by multiplying the share in total emissions from these regions with the relevant site-dependent factor.
- Processes for which time and location are unknown, because they are planned for the future or because they are simply unknown. In these cases it seems the best solution to use the site-generic factor to characterise the emissions of those processes and to report the related uncertainty of refraining from a site-dependent characterisation.

Availability and application of the relevant site-dependent factors may be a more pregnant problem in applying site-dependent characterisation. The processes in a product system can basically take place all over the world. Washing his jeans requires electricity from the public grid for a Dutch consumer. This electricity is for a large part generated in the Netherlands and from cokes, but the cokes themselves are imported from Australia and South America. Cotton for the jeans may be grown in the United States of America. Site-dependent characterisation of the emissions that are related to the jeans product system thus requires factors for each of these regions. This is a problem since presently site-dependent factors are only available for regions on a limited number of continents (Europe, United states of America and South America) and these characterisation factors are based on different definitions of the category indicator (see also Section 3.2). Practical here seems to use a set of spatial resolved factors related to the continent where the majority of processes take place. The region that took the initiative to perform the LCA will often determine this.

A last, but not the least obstacle in applying spatial resolved characterisation factors is caused by these factors not yet being implemented in LCA software that with a few pushes on the button performs the job. This is hopefully just a matter of time since information technology basically already allows to use even higher spatial resolved GIS-techniques and some practitioners seem to already combine these techniques with their LCA's.

Even when implemented in LCA software, some practitioners like those performing routine screening LCA's may choose to continue refraining from spatial resolved characterisation. This is in the end up to the practitioner and his commissioner. For any reason to avoid site-dependent characterisation, however, it seems reasonable to ask for a quantification of the uncertainty posed by the site-generic characterisation. This at least will trigger the decision-maker that he may run into a potential false optimisation.

3.4 Best available practice in characterisation

The state-of-the-art science in general for each of the impact categories (see also Chapter 4) determines both the possible level of spatial (and temporal) resolution, as well as the level in the causality chain at which the category indicator can be defined. Environmental relevance gained by spatial differentiation increases by considerable reduction of uncertainty in characterisation results as the indicator is defined further along the causality chain. The example in Appendix 1 illustrates this. Reduction of uncertainty from site-generic characterisation further along the causality chain is on the other hand relative small. The level in the causality chain of the category indicator is limited by the available spatial and temporal modelling if best available practice in characterisation is pursued (not to confuse with best available practice in weighting; see Section 2.4). Temporal differentiation is anticipated to also add some environmental relevance. However, this is for the time being a matter of further research.

Site-dependent characterisation in LCA needs only a rough indication of the region where a process ant its emission takes place. This information is often readily available from goal and scoping and inventory analysis, and does not seem to evoke large complications in the sense of additional data gathering. There may nevertheless a number of valid reasons to refrain from spatial differentiation in LCA. For instance because site-dependent factors lack for too many regions covered by the product system, or because they are not yet implemented in LCA software and therefore ask additional efforts from the practitioner. Refraining from spatial differentiation creates a considerable uncertainty in characterisation results and may therefore result in false optimisations. Whatever reasons a practitioner may have to refrain from spatial resolved characterisation, quantification of the related uncertainty is recommended to facilitate the decision-maker.

4 Scientific and political backgrounds

4.1 Introduction

It was in 1962 that Rachel Carson's book "Silent spring" woke up the world at once about the hazards involved in the dramatically increased production and use of chemicals, especially the chlorinated pesticides (from which some of them are now going to be banned in the UNEP POPs Convention). Since then, western societies have put great efforts in establishing an environmental policy. Now, national authorities have to their disposal an extensive body of policy instruments to bring under control the most urgent situations caused by those chemicals and to prevent similar situations in the future. Licenses, levies and subsidies, and anti-pollution taxes are just a small portion of this whole set of instruments available.

Western environmental policies have been a great success in some respects. Emissions of most individual installations have been regulated such that the risk posed by chemicals in the direct vicinity of those installations is within acceptable levels. In addition, the total releases of many chemicals have been reduced considerable (in some cases with even more than 90%; Hulskotte 1997), whereas the emissions of most combustion gasses have been stabilised or are gradually reducing (EEA 1998). Despite these impressive achievements, the environment has unfortunately not disappeared from the political agenda. The character of environmental problems and related policies is rather changing.

Initially, the focus of environmental concern was on mastering environmental problems in the vicinity of large point sources particularly. Later on, the attention moved to the transboundary effects of emission from all sources (large and small ones). Observed effects caused by long-range transport of pollutants from transboundary sources formed the basis for international environmental co-operation in analysing the problem and formulating solutions. Global and regional air-pollution problems have been topic of scientific and political concern in the world over the last two decades. The analytical tools developed in support of these policies are relevant for the global and regional air-pollution in this report as they can be used – and already have been used (see previous chapter) – to establish characterisation factors for life cycle assessment. This chapter therefore attempts to give an overview of the global and regional state of the environment and policies for the impact categories covered here.

4.2 Present environmental situation

Though risks of chemicals in the direct vicinity of installations are usually under control now, the long-range atmospheric and fluvial transport of, amongst others, emissions from combustion processes result in exposure increases at distances far from their sources. This together with the high intricate net of sources all over Europe creates a situation where total exposure in any point usually results from a multiplicity of sources. Present global and regional environmental problems like climate change, stratospheric ozone depletion, acidification, tropospheric ozone formation, and terrestrial and aquatic eutrophication are characterised by:

- Multiple substances, though usually a limited number, contributing to a given impact category following similar environmental mechanisms (principal pollutants: phosphor and nitrogen for eutrophication, nitrogen and sulphur for acidification, nitrogen oxide and volatile organic compounds for tropospheric ozone formation, halo-carbons for stratospheric ozone depletion and greenhouse gasses for climate change),
- Multiple receptors, i.e. more than several hundreds to thousands, being exposed by the emission of one source whereas each receptor the other way around is exposed by
- Multiple sources, i.e. more than several hundreds to thousands, being responsible for the release of the relevant substances (combustion processes in all economic sectors are the dominant sources for most impact categories mentioned, whereas agriculture, sewage water treatment plants are the main contributors to aquatic eutrophication).

An example regarding acidification may illustrate this.

Industrial sector	SO_2	NO_x	NH_3
Refineries	40.0%	3.1%	< 0.1%
Energy sector	11.3%	11.1%	
Industry	22.1%	11.8%	2.6%
Transport	21.3%	63.2%	
Consumers	0.6%	4.2%	3.9%
Agriculture	0.2%	4.7%	93.2%
Waste disposal	1.1%	0.6%	< 0.1%
Nature	%	0.2%	
Others	2.8%	1.5%	0.3%
Total	100% 147kt	100% 535kt	100% 156kt

Fig 4.1: The breakdown at the aggregate level of the economic sectors contributing to the atmospheric emission of acidifying compounds in the Netherlands (Hulskotte et al., 1997).

Figure 4.1 gives a breakdown at the aggregate level of contributing economic sectors, and Figure 4.2 shows the emission density of acidifying compounds in the Netherlands in 1998. Of course, each sector can be further broken down up to the individual sources. It may be clear that those individual sources have only a small, in most cases marginal contribution to the total Dutch emission of acidifying substances. The same phenomena exist for other countries (Amann et al., 1996), as well as for the principal pollutants related to the other global and regional impact categories (EEA 1998).

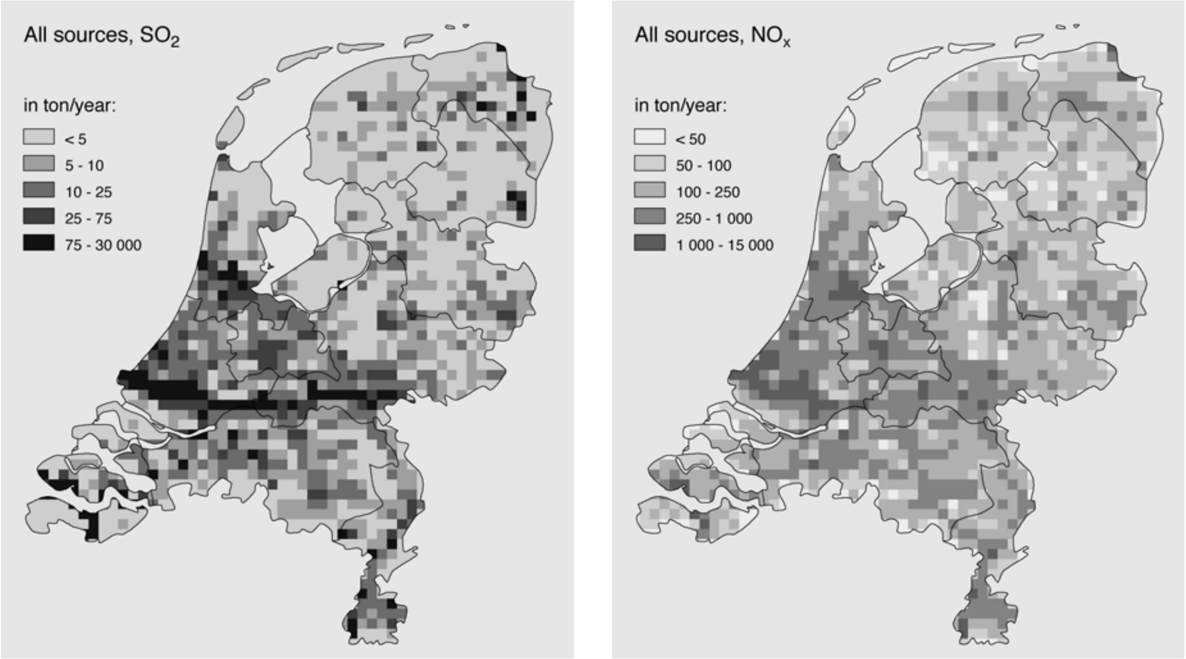


Figure 4.2: *The emission densities of acidifying compounds to air in the Netherlands in 1998 per 5*5km grid elements expressed in tons per year (VROM/HIMH 2001).* VROM/MIMH presents a summary of the emissions of 700 substances to air, water and soil from all sources in the Netherlands for the year 1998 (final data) and for 1999 (preliminary data) which have been collected in the national Pollutant Emission Register (actually the Dutch Pollutant Release and Transfer Register) to monitor the progress of environmental policy. Agreement about definitions, methods and emission factors, based on reports by expert groups, is achieved in the Co-ordination Committee for the Monitoring of Target Sectors (CCDM). The inventory has been established in a co-operation between the Inspectorate for Environmental Protection of the Ministry of Housing, Spatial Planning and the Environment (VROM/HIMH), Statistics Netherlands (CBS), the National Institute of Public Health and Environment (RIVM), the Ministry of Agriculture, Nature Conservation and Fishery (LNV), the Ministry of Transport, Public Works and Water Management (V&W) through representation by the National Institute of Water Management and Waste Water Treatment (RWS/RIZA) and the Netherlands Organization for Applied Scientific Research (TNO).

The principal pollutants for the global and regional impact categories have as a common characteristic that they are relatively long-lived (i.e. persistent). This is the reason why those pollutants have usually travelled already large distances before they arrive at the locations where they exert their impact. Lifetimes of most greenhouse gasses (climate change) and halo-carbons are even so long that they distribute globally. These substances may exert their impact everywhere on earth independent of the location where they are released (that is why they are called global impact categories). Principle pollutants related to the regional impact categories have shorter, though still relative long lifetimes of a few days. These substances may travel several hundreds to thousands of kilometres and thus have their impact regional to their location or release (Barret and Berge, 1996). Figure 4.3 gives an overview of the pattern of sulphur deposition for an emission of sulphur dioxide in the Netherlands. Whereas exposure increases are obviously largest within the first hundreds of kilometres, depositions can still be traced over some 2000km.

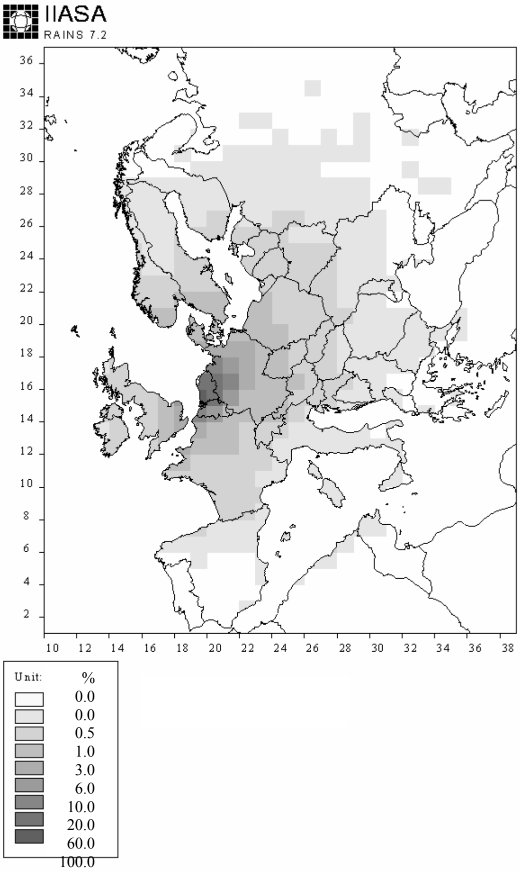
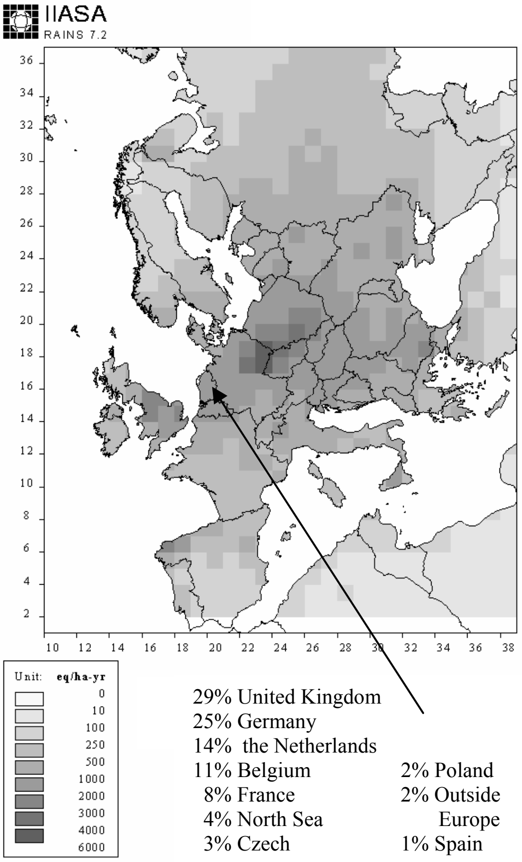


Figure 4.3: Deposition pattern on the European grid resulting from an emission of sulphur dioxide in the Netherlands. The depositions are expressed in percentages relative to the deposition in the sourcing grid element.



29% United Kingdom
25% Germany
14% the Netherlands
11% Belgium
8% France
4% North Sea
3% Czech
2% Poland
2% Outside Europe
1% Spain

Figure 4.4: Deposition pattern on the European grid resulting from the emissions of sulphur dioxide all over Europe and outside Europe. The depositions are expressed in eq/ha-year. Percentages contribution from the main countries to deposition on the Netherlands.

As can easily be imagined from Figure 4.2 and 4.3, there is a heavy overlap in the areas where individual sources have their impact. This leads to the situation where the exposure levels in any point are the result of the contributions of very many sources. The situation is illustrated in Figure 4.4 that gives the deposition pattern on the European grid from the emissions of sulphur dioxide all over Europe and from outside Europe. The percentage contribution from the main countries to deposition on the Netherlands is also given. The Dutch contribution to self-deposition is about 14%, which means that 86% of the total deposition on the Netherlands comes from sources abroad. In the same way, about 88% of total Dutch emissions crosses the borders and is so exported for deposition on other countries (Barret and Berge, 1996).

4.3 Global international agreements

4.3.1 UNFCCC & Climate change

The earliest discussions of greenhouse gas induced climate change at the turn of the 19th century came from northern Europe, where at that time the prospect of warming was not considered to be a problem. The possibility of human-induced climate change resurfaced in the USA and elsewhere in the middle of the 1960's with the publication of the "President's Science Advisory Committee" report (PSAC 1965). This report included a chapter that expressed concern about the increase of carbon dioxide in the atmosphere. Concerns about increasing carbon dioxide concentrations were also featured in the "Study of Critical Environmental Problems" at MIT in 1969 (SCEP 1970). During the 1970's, the focus of concern shifted to potential global climatic cooling induced by industrial and agricultural aerosol. Concerns about aerosol cooling began to abate by the late 1970's and greenhouse warming again came to prominence in the scientific literature. This was marked in 1979 by a report from the National Academy of Science in the USA (National Research Council, 1979).

Discussion on increasing greenhouse gases continued largely in the scientific community through the 1980's. That changed around 1990 with the formation of the Intergovermental Panel on Climate Change (IPCC), a large multinational group of scientists charged with providing scientific input to governments on climate change.

Greenhouse warming was a major topic at the United Nation Conference on Environment and Development (UNCED) in 1992 in Rio de Janeiro in Brazil. The Framework Convention on Climate Change (UNFCCC), established by the United Nations and aimed at reducing the rate of climate change, was opened for signature at the UNCED. Various governments, including the USA and industry groups, opposed taking definitive action through the UNFCCC at this point on the ostensible basis that they felt the underlying science and models used to investigate climate change were inadequate.

The Kyoto Protocol to the UNFCCC and negotiated in Kyoto (Japan) in December 1997 sets out targets for reduction of greenhouse gas emission in the industrialised nations in the vicinity of 10% below 1990 levels by 2010. The exact targets vary from country to country.

European nations originally pushed for larger reductions in the vicinity of 20%. In the lead up to the Kyoto Protocol to the UNFCCC in Kyoto in December in 1997, the USA Congress made clear that it would not ratify any treaty that did not include significant participation by developing countries. The USA has therefor signed but not ratified the Kyoto Protocol.

The Kyoto Protocol also introduced a variety of mechanisms with the potential to offset or avoid emissions reductions in the emitting country. These included counting of “sinks” of greenhouse gases through forest and conservation programs, and “activities implemented Jointly” between industrialised and lesser-industrialised nations to reduce the growth of greenhouse gas emissions in lesser-industrialised countries. The details of these mechanisms were left for subsequent Conferences of the Parties (COP) to the Protocol. The most significant of the subsequent meetings to address these issues was the COP 6 meeting in The Hague in the Netherlands in November 2000. The USA and other nations, including Japan and Australia, sought to obtain a suite of credits for emission reductions from domestic agricultural and forestry activities. They were generally opposed in this effort by the European countries. No agreement was reached on the details of these schemes at COP 6. The next effort to craft an agreement on the details will be in Germany in 2001.

The success of the UNFCCC process is hard to gauge at this point. The USA and many other industrialised countries have not significantly reduced emissions of greenhouse gases. The voluntary schemes for emissions reductions introduced in the USA appear to have had little direct affect. A nascent industry in facilitating greenhouse gas emission reductions and offsets has developed, but it is probably still too early to judge its potential. The USA Congress has not showed signs of softening its hard line position on emissions reductions. However, the general trend among multinational corporations has been for a shift away from denying that climate change is a problem. In the past five years the underlying science and models have receded somewhat into the background in the debate on climate change, to be replaced by more discussion on policy instruments and economic issues.

While climate models now figure less prominently in the rhetorical debate over climate change, they are still widely used to address this issue. The three-dimensional climate models are used to help project regional consequences of climate changes (Houghton, 1996). Simpler climate models are typically used in Integrated Assessment (IA) models linking climate, ecological, and economic processes. The degree of sophistication of IA models varies, along with their emphasis on different parts of the causal chain and focus on ecological or economic impacts. The method of translating climate changes into economic impacts in IA models has been the subject of extensive debate, as has been the issue of whether and how to discount impacts in future time periods.

The UNFCCC process is attempting to account for emissions of a variety of greenhouse gases beyond carbon dioxide (methane, nitrous oxide, CFCs, etc.). In order to compare the effects of different greenhouse gases using a standard metric, the IPCC adopted the concept of global warming potentials (GWPs) (Houghton et al., 1996). Using the GWP concept, the effects of different gases are typically compared in terms of a carbon dioxide equivalent. The GWP concept is fraught with difficulties and is less straightforward than the Ozone Depletion Potential counterpart used in the Montreal Protocol. The difficulties relate to the interactions between different greenhouse gases (which affect one another's lifetimes) as well as to the problem of attributing a unique lifetime to CO_2 and other gases that are recycled on a variety of different time scales. This in turn gives rise to a number of somewhat arbitrary choices in attempting to define GWPs. Further, these choices have important consequences in attributing the fractional responsibility of different greenhouse gases in any formal accounting system (Kandlikar and Milind, 1996). Thus, while GWPs are potentially useful, and are indeed being used, one must exercise considerable caution in interpreting greenhouse gas accounting frameworks deduced using GWPs.

4.3.2 Montreal protocol & Stratospheric ozone depletion

Concerns about human impacts on the ozone layer, particularly with regard to the impacts of super-sonic flight, date back to the 1960's and 1970's. The scientific community first raised concerns about possible ozone depletion from Chlorinated Fluor Carbons (CFCs) and related compounds in the 1970's. Two scientists in the USA, Sherwood Rowland and Mario Molina were particularly instrumental in outlining this connection (e.g. Molina and Rowland, 1974). They have gone on to receive a Nobel Prize for this work.

CFCs were first manufactured in 1931 as substitutes for ammonia and sulphur dioxide, the toxic refrigerants then in use, because they were very low in toxicity, non-flammable, and chemically stable. The initial response in the USA on the identified connection between CFCs and ozone depletion led to a phase out of the use of CFCs as a propellant in spray cans in the early 1980's. The debate on the connection between CFCs and ozone depletion continued in the early 1980's between scientists such as Rowland and Molina and scientists in the chemical industry under the rubric of the Alliance for Responsible CFC Use.

Concern for protection of the ozone layer was first formalised in 1985 with the Vienna Convention for the Protection of the Ozone Layer. The Vienna convention outlined initial responsibilities for protecting the ozone layer and established the framework process that led to the called Montreal Protocol. These treaties were preceded by a series of international "political" ozone conferences (since 1977) where political **and** scientific issues were discussed. It is interesting to note that already at the second conference in Munich (Germany) the ban of freons in sprays was announced by the US delegation. The relatively crude models of that time predicted first drastic decreases of stratospheric ozone, but later far smaller ones, according to Rowland & Molina´s mechanism of homogenous catalytic ozone destruction.

In 1985 the scientific community reported discovery of a growing ozone hole over Antarctica. This had been overlooked in the satellite measurement data (which rejected the data as anomalous) until reported from ground-based measurements. An Ozone Trends Panel (Watson et al., 1986; NASA, 1988) was subsequently established to provide current assessments on the status of the science underlying ozone depletion. Developments in the science went on to demonstrate a strong link between CFCs and ozone depletion. A hallmark of the science throughout the last fifteen years on ozone depletion has been that the observations have tended to confirm the original theories, occasionally providing surprises of ozone depletion in excess of that anticipated. For example, the ozone hole over Antartica was not anticipated by the Roland & Molina theory.

The Montreal Protocol on Substances That Deplete the Ozone Layer was negotiated and signed in 1987, undergoing further amendments in 1990 and 1992 that strengthened the original provisions. The Protocol stipulates that the production and consumption of CFCs, halons, and carbon tetrachloride should be phased out by 2000 and that methyl chloroform should be phased out by 2005. Further impetus to the amendments in 1992 was given by the discovery of sustained ozone reductions of up to 20% in the Northern Hemisphere at that time. The London amendments in 1990 were also strengthened on the basis of the developments of the science and modelling of ozone depletion.

To be sure, it is difficult to separate out the role of science and industrial developments in explaining the Montreal Protocol process. DuPont and other large chemical manufacturers developed non-ozone depleting alternatives to CFCs in 1988, concurrently with the publication of the Ozone Trends Panel report. Industry resistance to the Protocol process reduced substantially in the wake of these developments. The Protocol has been very successful in reducing production of CFCs and related ozone depleting compounds. However, because of the long lifetimes in the atmosphere of these compounds, it will be many more decades before the ozone layer begins to return to levels characteristic of the first half of last century. The political process that led to the ban of the freons and related compounds is nevertheless one of the greatest global success with regard to an environmental threat. One should also consider that it took only 13 years from the first scientific publication by Roland and Molina to the signature of the Montreal protocol!
Since a variety of different substances deplete ozone, the Montreal Protocol has used the concept of Ozone Depletion Potentials' (ODP) to standardise impacts across substances. The ODP is the ratio of the impact on ozone of a chemical compared to the impact of a similar mass of CFC-11. ODP's have been accepted and adopted by the WMO and the Montreal processes and they provide a convenient form in which to assess the ozone damage potential of different substances.

4.4 “Atmospheric” regional international agreements
(Tropospheric ozone formation, acidification and terrestrial eutrophication)

4.4.1 Europe: UNECE Convention on Long-Range Transboundary Air Pollution

The threats of pollution accumulation on a regional scale were already recognised in the 1970’s in Europe with the acidification of lakes and forest in Scandinavia in particular. The awareness grew that co-operation at international level was necessary to solve these transboundary environmental problems. This resulted into the UNECE[17] Convention on Long-Range Transboundary Air Pollution (CLRTAP) that was signed in 1979 and entered into force in 1983. The CLRTAP initially aimed at reducing acidification through control of the emissions of sulphur. Later, its scope was widened to include nitrogen pollutants, volatile organic compounds and photochemical oxidants. Presently, heavy metals and persistent organic pollutants are also covered. (UNEP, 1999)

The CLRTAP was one of the first internationally legally binding instrument to deal with problems of air pollution on a broad regional basis. All European countries, including the European part of the former Soviet Union, have committed themselves in the several protocols[18] signed under the CLRTAP to bring their national emissions to agreed ceilings in the year 2010. This allows few countries to increase their emission compared to the base year 1990, but most of them have to reduce considerably. (UNEP, 1999)

A protocol on financing of the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe was signed in 1984, and entered into force in 1988. The co-operative programme, better known as EMEP, has three main components (UNEP, 1999):

- European inventory of atmospheric emissions of sulphur dioxides, nitrogen oxides, ammonia, non-methane volatile organic compounds and a number of other air pollutants. The collection of these data takes place under the joint responsibility of EMEP and Corinair (the latter through the European Environment Agency).
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- Development and maintenance of one-layer trajectory models for atmospheric dispersion. These models are, amongst others, used to arrive at atmospheric transfer matrices that summarise the long-term average dispersion “from country to grid-elements” of emission

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[17] UNECE stands for United Nations Economic Commission for Europe.

[18] Completed and adopted are the first and second protocol on sulphur emissions, first protocols on emissions of nitrogen oxides, volatile organic compounds, heavy metals, and persistent organic pollutants. Recently, negotiations were completed regarding the new multi-effect/multi-pollutant protocol - addressing photochemical pollution, acidification and terrestrial eutrophication posed by nitrogen oxides and related substances. The draft Protocol to Abate Acidification, terrestrial Eutrophication and Ground-level Ozone is ready for adoption by the Executive Body.

of notably sulphur dioxides, nitrogen oxides, ammonia and non-methane volatile organic compounds.

- Measurement of air quality and precipitation by some 100 monitoring stations in 24 European countries. The air quality and precipitation data are, amongst others, used for calibration and validation of the trajectory models owned by EMEP.

In addition to the activities under EMEP, the CLRTAP appointed in 1988 a working group to develop a common understanding of the critical load approach. A critical load is defined as “a quantitative estimate of an exposure to one or more pollutants below which significant harmful effect on specified elements of the environment do not occur according to present knowledge”. The critical load approach was discussed, developed and applied in a number of international meetings held under the CLRTAP. This resulted first in 1993, but revised in 1996 in a manual on methodologies and criteria for the mapping of critical loads (UBA 1996).

The CLRTAP requires each country to establish and map the critical loads for sulphur and nitrogen for various areas and receptors in its own territory. The impact categories covered are acidification (sulphur and nitrogen) and terrestrial eutrophication (nitrogen). The resulting data is being assembled by the Co-ordination Centre for Effects (CCE), and used in the production of maps for the critical loads all over Europe. The latest CCE Status-report makes mention of critical loads being provided by 24 European countries covering all together over a million ecosystems (Posch et al., 1999/2001; Hettelingh et al., 2001).

For troposhperic ozone formation, the CLRTAP uses the concept of Accumulated Ozone exposure above Treshold (AOT) to evaluate the risks for human beings and ecosystems posed by elevated concentrations of thropospheric ozone. The AOT40 (= 40 ppb ≈ 80 $\mu g/m^3$) is used for crops and forest. Critical AOT40 ozone levels have been established at 3,000 ppb.h for crops during daylight hours over the period from May to July, and at 10,000 ppb.h for forests during daylight hours over the period from April to September. The AOT60 (= 60 ppb ≈ 120 $\mu g/m^3$), and a critical AOT60 ozone level of zero is used for human beings. (EEA , 1998; Posch et al. 1999/2001; Hettelingh et al., 2001).

Model calculations have been an important input to the negotiations about, and the ceilings finally agreed in the several protocols. Also the analytical framework for these calculations are established and agreed upon under the CLRTAP. This gives the framework, besides a scientific basis, also political credibility (UNEP, 1999).

The EMEP/Corinair atmospheric emission inventory, the EMEP atmospheric transfer matrices, and the CCE critical load and level data were used in the several calculations. The Abatement Strategies Assessment Model (ASAM) created at Imperial College in London, UK, the Coordinated Abatement Strategy Model (CASM) from the Stockholm Environment Institute (SEI) at York, UK and the Regional Air Pollution INformation and Simulation model (RAINS) developed at the International Institute for Applied Systems Analysis (IIASA) in Laxenburg, Austria provided the model calculations being at the basis of the several protocols under the CLRTAP. They are a 'easy-to-use' computer tools for analysing abatement strategies to reduce acidification, terrestrial eutrophication and tropospheric ozone formation in Europe.

4.4.2 North America

4.4.3 Asia

4.4.4 Other continents

4.5 “Aquatic” regional international agreements
(Aquatic eutrophication)

Transboundary aquatic eutrophication is bound to the catchment through which an emission is transported before it reaches marine waters. International agreements to protect the inland and marine waters will typically relate to such catchment area that are usually far smaller than the scale of regional atmospheric problems. It is not feasible to give a world-wide coverage in this section. The situation in Europe is therefore here described as an example only.

Several international measures to protect the marine and coastal environment have been introduced on a regional basis in Europe. They include (Gibson, 1999):
- Mediterranean Action Plan and the Barcelona Convention
- Helsinki Conventions on the Baltic Sea
- OSPAR Convention
- North Sea Conferences
- Trilateral Wadden Sea Co-operation
- RAMOGE Agreement
- the proposed Council of Europe Model Law on Sustainable Management of Coastal Zones.

The OSPAR Convention covers parts of the Atlantic and Arctic Oceans, but excludes the Baltic and Mediterranean Seas. RAMOGE is a sub-regional Mediterranean agreement between France, Monaco and Italy on the protection of the coastal waters of the Côte d’Azur.

The agreements have different contents intending to assist national governments in the formulation and implementation of effective anti-pollution policies. All of them cover one way or another land-based sources causing aquatic eutrophication. The agreements have different objectives and targets for eutrophication issues because of differences in political willingness and conditions of the water environments.

As a result the requirements of a more global and coherent approach to water policy a proposed Water Framework Directive of EU will provide a new framework for protection of water resources. It includes inland surface water, transitional waters, coastal waters and groundwater. A key feature of the proposed Directive is that it requires Member States to manage and co-ordinate administrative arrangements at the catchment areas.

4.6 Conclusions

The chapter attempts to give a global overview of the state of the environment for the impact categories covered here. The incompleteness of this review with regard to the non-global impacts basically illustrates – except that the involved authors were predominantly from Europe – that these categories also have varying degrees of importance on the several continents. This, together with cultural differences in problem solving and environmental management may explain deviating levels of sophistication of regional models to characterise the several impacts. So does to our knowledge exist no model for assessment of acidification in Australia and New Zealand where this is not perceived as a problem. North America on the other hand employs sophisticated emission inventories and atmospheric transport models, but appears not to have effect measures like critical acidifying loads. Europe on its turn keeps sophisticated integrated models like for instance the RAINS model that is established under, and used as an input to the several protocols negotiated in the context of the UN/ECE convention on Long-Range Transboundary Air Pollution (CLRTAP). The large differences in impact modelling between continents pose problems for characterisation in LCA since the processes making up a given product system may take place all over the world (see Section 3.3.3). Characterisation factors should therefore preferably be based on models relating to the whole world. This would unfortunately force the category indicator to be chosen relative early in the causality chain, however, since this is commanded by the continental model with the lowest level of sophistication (characterisation factors for other continents should adapt to this in order to arrive at a globally consistent set.

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Appendix 1: Choice of indicator and spatial resolution

(submitted as example for ISO EN TR 14047, Technical Report of the International Standard Organisation on life cycle impact assessment)

A 1 Examples of selecting impact categories in relation to the study goal and scope

A 1.1 Overview – Selecting Different Acidification Impact Category Indicators

The example illustrates the importance of ISO 14042 recommendations and the criteria for environmental relevance by comparing two very different indicators (see Table 1). There are very significant differences between the indicator results, e.g., over 5000-fold (Table 3), even when the same inventory results are used. Due to the concentration on a single impact category, section 5.2 is omitted.

A 1.1.1 Describing the environmental mechanism for an impact category

Two alternative impact category choices for acidification are used. The first alternative is a single impact category for the total emission burden or load of acids and acid precursors to the environment. The single category combines several separate effects using value-choices, e.g., aquatic impacts, terrestrial impacts, and deterioration of materials in buildings and other structures. The category indicator reflects system environmental burden or the total flow of possible acid emissions crossing the system boundary or the system burden. The indicator, however, provides no environmental information, e.g., condition, intensity of impact, reversibility, etc. The second alternative uses a specific endpoint, possible effects on terrestrial plants. The characterization model is intended to provide environmentally relevant information and:

- uses the spatial location of inventory emissions,
- characterizes the conversion of each emission to acid,
- characterizes the spatial transport of each acid to different receiving locations, and
- characterizes the change in sensitive ecosystems at each receiving locations.

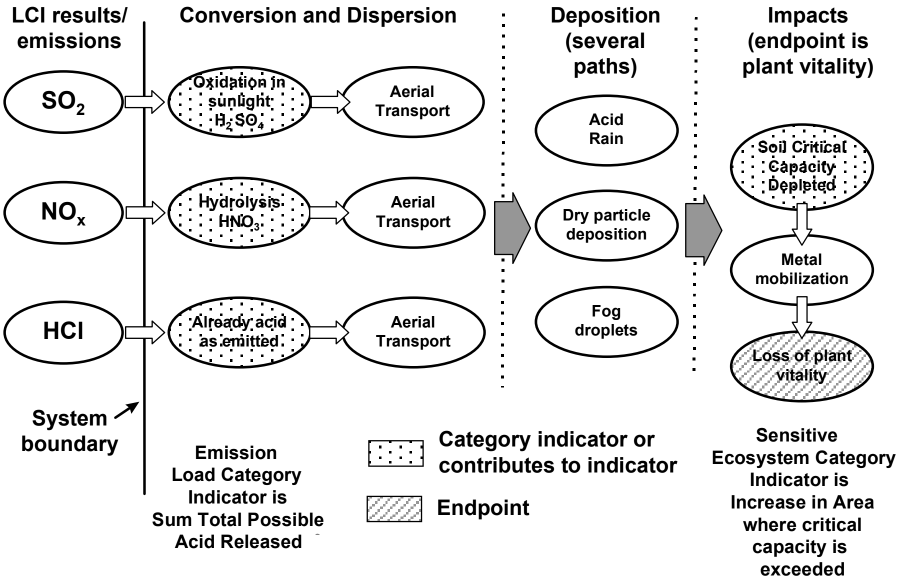

Figure 1: Simplified environmental mechanism for acidification.

A simplified environmental mechanism for acidification is shown in Figure 1. The Figure shows the flow of emissions across the system boundary, their conversion to different acids, their dispersion to remote spatial locations, their deposition as acids in spatially remote locations by several paths, and, if the critical capacity of the soil to neutralize acids is exceeded, the effects on terrestrial plants. The location of two different indicators in the environmental mechanism is shown. The steps are described to illustrate the differences in these indicators.

Emission or release

Acidification begins with the emission of compounds such nitrogen (NO_x and NH_3) and sulphur (SO_2). These emissions are LCI inventory results that flow across the system boundary to the environment. NO_x, NH_3 and SO_2 are not emitted as acids and are converted to acids in the environment. Other emissions, such as hydrogen chloride (HCl), are emitted directly as acids and need no conversion.

Conversion, Dispersion, and Deposition

NO_x, NH_3 and SO_2 are converted to acids in the atmosphere and undergo long-range transport and dispersion to distant receiving locations several hundreds to a thousand kilometers from the emission source. The acids are deposited in remote locations by several possible means (e.g., acid rain, dry particles, and in fog droplets). Several factors determine the acid amounts that reach a specific receiving area. For environmental relevance, these factors must be included in spatially specific characterization models, such as:

- emission conversion into acid has its own chemical reaction and depends on temperature, weather, etc.;.
- transport distance and direction depends on source location, stack height, weather, etc.; and
- deposition depends upon each acid's characteristics, e.g., particle size, and weather conditions, e.g., rain.

Note: transport and deposition can be annualized from environmental models for the characterization factors.

*The role of the receiving ecosystem's **critical capacity** to neutralize acid*

Deposited acids may decrease the receiving water or soil pH. The pH decrease depends both on the amount of acid deposited from the LCA system, the background acid load from other human and natural sources, and the receiving site's neutralization capacity. Each site has a given capacity to neutralize acid, i.e., the critical capacity. When the critical capacity of a ecosystem is exceeded, the pH decreases and impacts (e.g., lost plant vitality) are likely. For acidification, the critical capacity is then the level at which impacts begin to occur. This is similar to the World Health Organization (WHO) noting that human health impacts may occur when substances exceed the Allowable Daily Intake (ADI) for people over a 70 year lifetime of exposure. When the critical capacity is not exceeded, acidification impacts do not occur. When the WHO ADI is not exceeded, no disability or death would be expected, i.e., real world DALYs would be zero. For environmental relevance, It is then essential to identify when measures such as the critical capacity or ADIs are exceeded. Compared to a total emission load indicator, one must recognize:

- only a small percentage of the total emissions are actually deposited in sensitive ecosystems where the critical capacity is exceeded, causing impacts, and
- the percentage varies substantially depending upon the spatial locations of the emission source and the receiving ecosystems.

Thus, a total emission load indicator, ignoring these environmental details will have very different indicator results from a sensitive ecosystem indicator, even if the starting LCA inventory results are the same (see Table 3).

A 1.2 Indicator Models and Characterization Factors

The models and characterization factors for two category indicators are described. Section 5.4.4 is omitted.

Emission-Loading Category Indicator Model (hereafter EL indicator)

The EL indicator model characterizes the total emission-load released by the LCA systems using a chemical equivalence calculation. The model omits spatial information on fate, dispersion, or the amount of acid deposited into sensitive areas. The model assumes complete conversion to acid, complete deposition to sensitive regions, and occurrence of environmental effects in every location. These are worst-case assumptions and lack environmental information and relevance (see Table 1). However, the EL indicator results are often referred to as 'potential environmental impacts' by some parties.

Acid Deposited in Sensitive Ecosystems Category Indicator (hereafter the SE indicator)

The SE indicator characterization incorporates spatial aspects and fate and transport and addresses environmental relevance as recommended by clauses 5.3.4 and 5.3.5 of ISO 14042. This also illustrates the importance of the goal and scope selection process in Annex A of ISO 14042. The SE model is more complex and includes the emission conversion and dispersion from a given country, the acid amounts deposited in receiving countries, and the area of sensitive ecosystems in the receiving countries whose critical load is exceeded. The results of the SE indicator provide information on the environmental performance of the system, while the EL indicator does not.

The SE model adapts the European RAINS model[19]. The RAINS model uses 150 by 150 km grids or cells for both emissions and receiving ecosystems. These cells allow the mathematical accounting for emissions from each cell, the percentage conversion to acid, transport and deposition from each source cell to each possible receiving cell, the different areas and their critical capacities of soils within each receiving cell, etc. The LCA adaptation converts the cells to countries, so the inventory must only record the country of an emission. Each country has a characterization factor (e.g., AF_{NOx} and AF_{SO2} – see Table 2) to calculate for each emission conversion to acid, transport, and deposition and then calculate area at each receiving sties where the critical capacity is exceeded. Each emission is converted with the characterization factor from kilotons (or grams) of emission to the increased area in hectares (or square meters) where the critical load is exceeded. For the complete derivation of the SE indicator, see Potting et al. Journal of Industrial Ecology, volume 2, pages 63-87 (1998).

To have environmental relevance, a characterization model must then incorporate certain information, such as:

- The geographical location of releases from the inventory.
- Ability to calculate the conversion, transport and deposition from each source location to each of the many different receiving areas.

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[19] RAINS is an integrated assessment model that combines information on national emission levels with information on long range atmospheric transport in order to estimate patterns of deposition and concentration for comparison with critical loads and thresholds for acidification, terrestrial eutrophication-via-air and tropospheric ozone creation.

- The ability to sum the acid load for each receiving area from a number of different emitting sites.
- The ability to relate the acid load in each receiving area to critical neutralizing capacities in the receiving areas and whether the critical capacity is exceeded.
- The ability to calculate the marginal increase in the area where the critical load is exceeded. This provides a direct linkage to the damage to which a system may be contributing. This final step adds important information on the condition of the receiving environment to the model.

Table A1: Coverage of 14042 recommendations and criteria

ISO 14042 Notes	EL Indicator	SE Indicator
LCI Results – Both indicators use the same LCI parameters, but spatial detail needed for SE indicator		
ISO 14042 - clause 5.3.4		
spatial and temporal differentiation of the characterization model relating the LCI results to the category indicator should be considered	**NO**	**YES**
fate and transport of the substances should be part of the characterization model	**NO**	**YES**
ISO 14042 - clause 5.3.5		
reflect the consequences of the LCI results on the category endpoint(s), at least qualitatively	**NO**	**YES**
condition of the category endpoint(s)	**NO**	**YES**
spatial aspects, such as the area and scale	**NO**	**YES**

A 1.3 Selection of the characterisation model and characterisation factor

The EL indicator results are expressed as proton equivalents or grams of a major emission, usually SO_2. The conversion and combination of acids is scientifically valid and contrasts with attempts to combine different human toxicities. Combining different human toxicities has been described as a subjective or value-choice score like combining global warming, acidification, and eutrophication (see International Life Sciences Institute. 1996 *Human Health Assessment and Life-cycle Assessment: Analysis by an Expert Panel.* T.A. Burke et al. Washington, DC.). For the EL indicator, the necessary LCI parameters are direct acids, such as hydrochloric acid, and substances possibly converted to acids, such as sulfur dioxide, nitrogen oxides, and ammonia. The characterization factors for several substances in addition to those in the simplified inventory calculations below are: 0.88 for HCL emissions, 1.00 for SO_2, 0.80 for SO_3, 0.70 for NO_x, 0.70 for NO_2 and 1.88 for NH_3.

Table A2. Characterization factors for several substances and countries according to the SE model.

Country or Region	**AF(SO_2)**		**AF(NO_x)**		**AF(NH_3)**		**AF(HCl)**	
	ton/ha	g/m^2	ton/ha	g/m^2	ton/ha	g/m^2	ton/ha	g/m^2
Albania	0.02	0.0002	0.00	0.0000	0.01	0.0001	0.00	0.0000
Belgium	1.28	0.0128	0.82	0.0082	1.10	0.0110	0.02	0.0002
Denmark	5.56	0.0556	2.02	0.0202	5.28	0.0528	0.06	0.0006
Finland	15.14	0.1514	2.42	0.0242	13.40	0.1340	0.07	0.0007
Germany	2.17	0.0217	0.90	0.0090	1.89	0.0189	0.02	0.0002
Netherlands	1.24	0.0124	0.97	0.0097	1.55	0.0155	0.03	0.0003
Portugal	0.02	0.0002	0.01	0.0001	0.01	0.0001	0.00	0.0000
United Kingdom	1.94	0.0194	0.92	0.0092	4.32	0.0432	0.03	0.0003

The SE indicator is expressed in hectares or square meters of area where the increased load of the LCA causes the critical capacity to be exceeded. The characterization factors for several countries with their spatially specific characterization factors (e.g., AF_{NOx} and AF_{SO2}) are given in Table 2 clearly shows how spatial differences will result in large differences in the characterization factors. For the acid SE category indicator, the collection of LCI parameters is more detailed. In addition to the hydrochloric acid, sulfur dioxide, nitrogen oxides, ammonia, etc., noted above, the region where each emission takes place must be recorded.

Calculation of the LCI results into the indicator result.
This section calculates category indicator results for the EL indicator and the SE indicator. The outcome in the values of the two indicator results can differ by over 5000 fold (see Table 3)! This reinforces the need to carefully evaluate choices in the study goal and scope and reinforcing the statement in ISO 14042 that: *The usefulness of the indicator results for a given goal and scope depends on the accuracy, validity, and characteristics of the characterization models and characterization factors. A trade-off often exists between characterization model simplicity and accuracy.*

The inventory is highly simplified using only NO_x and SO_2 and is based on the electrolytic refining of primary copper. The details of the mining, the drawing of copper wire, the production of PVC, the disposal and recycling of the wire with incineration of PVC are omitted. The functional unit is a kiloton of electrolytic refined copper produced and the parameters used are 10 grams of NO_x and 100 grams of SO_2. Identical processes and the same emission quantities are assumed to exist in three different locations. For the EL model a straightforward calculation is made using chemical characterization factors. For the SE model, the production process is calculated for three different emitting locations (Albania, Belgium, and Finland). The example calculations for the EL indicator results are:

10 g NO_x * 0.70 = 7 SO_2 g equivalents/kt copper, and
<u>100 g SO_2 * 1.00 = 100 SO_2 g equivalents/kt copper.</u>
= 107 SO_2 g equivalents/kt copper

Thus, whether the smelter was in Albania, Belgium and Finland, the same total burden is released and EL indicator results would be the same: 107 SO_2 g equivalents/kt of electrolytic refined copper.

The calculations on a site-dependent basis for the SE indicator result are shown in Table 3. The characterization factors are country specific so that the indicator results for the same quantities of emissions now differ considerably (1 to 769). The indicator results differ dramatically from the EL indicator results. For comparison,m these are also included on a relative basis in the right hand column of Table 3.

Table A3. Calculations for Indicator Results using SE Model

	NO_x (g * AF)	**SO_2** (g * AF)	**Indicator result** (m^2)	**Relative Comparison Comparison**	
Calculation of SE indicator results for same levels of emissions from three different spatial locations (countries)				**To SE result for Albania**	**To EL result**
Albania	10 * 0.00 = 0	100 * 0.0002 = 0.02	0.02	1	5350
Belgium	10 * 0.0082 = 0.008	100 * 0.0128 = 1.28	1.29	64	83
Finland	10 * 0.0242 = 0.242	100 * 0.1514 = 15.14	15.38	769	7
Maximum	7	100	107		1

Thus, on a relative basis, the two models yield results have dramatically different results! This clearly illustrates the effect of category model and indicator choices between a study goal and scope that only needs general screening results (EL indicator) and one that needs accuracy and environmental relevance (SE indicator).

It should be noted that, using the EL indicator results in the Interpretation phase, a lower level of total emissions from Belgium would at first appear to be environmentally ‘better’ than a somewhat higher level of total emissions from Albania. However, the environmentally relevant SE indicator would clearly show that emissions from Albania would cause a much lower area to exceed its critical capacity. Thus, decisions making important comparisons should consider selecting environmentally relevant indicators whose models incorporate spatial information on the emission source, the fate and transport processes, and the sensitive ecosystems.

A 1.4 Examples of the transformation of indicator results using several selected reference values, and how these transformations may yield different outcomes (Normalization)

This section illustrates several possible normalization procedures, including a per captia approach and a reference approach. The illustration shows how these procedures change the original category indicator results from the mandatory sections of ISO 14042 both in absolute terms and in relative terms. This illustration illustrates the cautions and recommendations regarding normalization and other optional procedures: *The selection of the reference system should consider the consistency of the spatial and temporal scales of the environmental mechanism and the reference value.* and *The normalization of the indicator results changes the outcome of the mandatory elements of the LCIA phase. It may be desirable to use several reference systems to show the consequence on the outcome of mandatory elements of the LCIA phase. A sensitivity analysis may provide additional information about the choice of reference.*

Normalization can use several reference values as selected by the goal and scope, such as, population, area, emission proportions, and historical emission baselines. Table 4 provides three values for several countries that can be used for reference values illustrating the large variation. Such different values will shift and alter the relative standing of the indicator depending upon the country used for the normalization reference. In addition, if only industrial processes were chosen for normalization, then only 2% of the Albanian, 27% of Belgian, and 24% for Finnish total SO_2 emissions would be used (e.g., 2,400 to 85,600 to 62,400 tons for a reference value, respectively). This would further increase the differences in the resulting normalized indicators.

Table A4. Reference and baseline values for normalization

Country	**Population (thousands)**	**Area (sq km)**	**Emission quantities per yr (tons)** SO_2	NO_x
Albania	3,119	27,000	120,000	30,000
Belgium	10,141	33,000	317,000	352,000
Finland	5,154	305,000	260,000	300,000
Germany	82,133	349,000	4,520,000	2,376,000
Spain	39,628	499,000	2,265,000	1,178,000
UK	58,649	242,000	3,751,000	2,701,000

If the normalization reference is the denominator, those countries with smaller populations, areas, or emissions will increase relative to larger countries when normalized. Table 5 applies both population and emissions baseline references to the SE indicator results derived in section 5.4.5. Relative changes due are shown in the right hand column of the table. Significant changes in the results occur in the outcome of the analysis by the choice of the normalization reference.

Table A5. Calculation of normalized indicator results using different reference and baseline values

<table>
<tr><th colspan="11">Example of per capita population normalization</th></tr>
<tr><th rowspan="2">Country</th><th colspan="2" rowspan="2">Indicator Result (m^2)</th><th colspan="2" rowspan="2">Population (thousands)</th><th colspan="2" rowspan="2">Normalized Indicator Result</th><th colspan="4">Relative size</th></tr>
<tr><th colspan="2">Before</th><th colspan="2">After</th></tr>
<tr><td>Albania</td><td colspan="2">0.02</td><td colspan="2">3,119</td><td colspan="2">0.0641×10^{-6}</td><td colspan="2">1</td><td colspan="2">1</td></tr>
<tr><td>Belgium</td><td colspan="2">1.29</td><td colspan="2">10,141</td><td colspan="2">0.127×10^{-6}</td><td colspan="2">64</td><td colspan="2">20</td></tr>
<tr><td>Finland</td><td colspan="2">15.38</td><td colspan="2">5,154</td><td colspan="2">2.98×10^{-6}</td><td colspan="2">769</td><td colspan="2">465</td></tr>
<tr><th colspan="11">Example of reference emission baseline normalization</th></tr>
<tr><th rowspan="2">Country</th><th colspan="2" rowspan="2">Indicator Result (m^2)</th><th colspan="2" rowspan="2">Emissions (tons)</th><th colspan="2" rowspan="2">Normalized Indicator Result</th><th colspan="4">Relative size</th></tr>
<tr><th colspan="2">Before</th><th colspan="2">After</th></tr>
<tr><th></th><th>SO_2</th><th>NO_x</th><th>SO_2</th><th>NO_x</th><th>SO_2</th><th>NO_x</th><th>SO_2</th><th>NO_x</th><th>SO_2</th><th>NO_x</th></tr>
<tr><td>Albania</td><td>0.02</td><td>0.00[20]</td><td>120,000</td><td>30,000</td><td>1.67×10^{-7}</td><td>3.33×10^{-10}</td><td>1</td><td>1</td><td>1</td><td>1</td></tr>
<tr><td>Belgium</td><td>1.28</td><td>0.008</td><td>317,000</td><td>352,000</td><td>4.04×10^{-6}</td><td>2.27×10^{-8}</td><td>64</td><td>800</td><td>24</td><td>68</td></tr>
<tr><td>Finland</td><td>15.14</td><td>0.242</td><td>260,000</td><td>300,000</td><td>5.82×10^{-5}</td><td>8.07×10^{-7}</td><td>757</td><td>24200</td><td>329</td><td>2420</td></tr>
</table>

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[20] A value of 0.00001 was used to conduct the normalization so that values from Belgium and Finland would not be divided by zero.