Regional LCA in a global perspective
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Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000*

Abstract

In the methodological context of the interpretation of environmental life cycle assessment (LCA) results, a normalisation study was performed. 15 impact categories were accounted for, including climate change, acidification, eutrophication, human toxicity, ecotoxicity, depletion of fossil energy resources, and land use. The year 2000 was chosen as a reference year, and information was gathered on two spatial levels: the global and the European level. From the 860 environmental interventions collected, 48 interventions turned out to account for at least 75% of the impact scores of all impact categories. All non-toxicity related, emission dependent impacts are fully dominated by the bulk emissions of only 10 substances or substance groups: CO_2, CH_4, SO_2, NO_x, NH_3, PM_{10}, NMVOC, and (H)CFCs emissions to air and emissions of N- and P-compounds to freshwater. For the toxicity-related emissions (pesticides, organics, metal compounds and some specific inorganics), the availability of information was still very limited, leading to large uncertainty in the corresponding normalisation factors. Apart from their usefulness as a reference for LCA studies, the results of this study stress the importance of efficient measures to combat bulk emissions and to promote the registration of potentially toxic emissions on a more comprehensive scale.

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Keywords

LCA, normalisation, economic system, impact categories, global assessment

6.1 Introduction

Life cycle assessment of products (LCA) has become a widely recognised method for quantifying the environmental performances of products (c.f. EC, 2001; Curran, 2006). Numerical scores make it possible to compare product alternatives on the aspects of climate change, ozone depletion, acidification, eutrophication, toxicity, fossil energy resource depletion, and more environmental impact categories. All environmental releases, fossil energy resource extractions and land use activities that belong to a product life cycle are translated and aggregated in the right proportions to deliver an environmental profile in terms of the overall contribution of the product to a limited number of impact categories (Guinée *et al.*, 2002). A comparison of environmental profiles reveals the relative environmental performance of product alternatives in the context of every single impact category. Despite their apparent simplicity, however, LCA profiles are not in every respect interpreted so easily. Impact scores are expressed in complex units, and reflect environmental impacts in a way that does not correspond directly to perceptible problems or prevailing threats. Their absolute value as an assessment measure remains difficult to interpret as long as it is not placed in an adequate environmental context. This is what LCA normalisation aims at.

LCA normalisation offers a reference situation of the pressure on the environment for each environmental impact category. Normalisation makes it possible to translate abstract impact scores for every impact category into relative contributions of the product to a reference situation. This reference situation consists of an environmental profile on a higher scale – that is, the environmental profile of an economic system that the product life cycle is considered to be part of. The fact that the normalisation results are expressed in the same unit for each impact score makes it easier to make comparisons between impact scores of different impact categories (Norris, 2001). Since product life cycles often have a global coverage – *e.g.* including resource extractions in diverse geographic regions – the global system is the most promising candidate to act as a reference situation (Guinée *et al.*, 2002).

A number of normalisation methods have appeared during the past 10 years. An overview is given in Table 6.1. Some methods are shown to be specific for a limited region or for a limited number of impact categories.

Table 6.1 Overview of LCA normalisation methods.

literature reference	reference areas	reference years	intervention types	number of impact categories
Wenzel *et al.* (1997)	Denmark	1990	emissions	11
Breedveld *et al.* (1999)	The Netherlands	1993/1994	general	13
	EU_{15+3}	1990/1994	general	12
Huijbregts *et al.* (2003)	The Netherlands	1997/1998	general	13
	EU_{15}	1995	general	13
	world	1990/1995	general	13
Strauss *et al.* (2006)	South Africa	2001	abiotic resource extraction	2
Stranddorf *et al.* (2005A & B)	Denmark	1994	emissions	11
	EU_{15}	1994	emissions	11
	world	1994	emissions	11
Bare *et al.* (2006)	United States	1999	general	10
Lundie *et al.* (2007)	Australia	2002/2003	toxic emissions	5
this chapter	EU_{25+3}	2000	general	15
	world	2000	general	15

This study can be considered as a follow-up of the study by Huijbregts *et al.* (2003), in which 1995 was used as the reference year, and which distinguished economic systems on three levels: the world, Western Europe, and The Netherlands. A follow-up was considered necessary for three reasons:

- Acquiring more up-to-date emission and extraction data (year 2000 instead of 1995) and more up-to-date boundaries for the European region (28 instead of 15 countries);
- Extending the number and improving the quality of emissions and extractions (although the availability and quality of the data for toxic emissions appeared to be still limited);
- Including more up-to-date impact assessment models in the normalisation factor calculations for many impact categories, including global warming, ozone depletion, toxicity and acidification.

The outcomes of this normalisation study can be interpreted as an LCA study of the economic systems as a whole on both the European and the global level. LCAs of economic systems can also be used in a broader context than normalisation, for instance by comparing the impacts of different economic systems with each other and by identifying the most important emissions or extractions within a specific impact impact category or economic system. Both aspects will be discussed and quantified.

6.2 General methodological choices

6.2.1 Reference system

Although the global economic system may be considered as the most justifiable reference system for normalisation from a scientific point of view, policy makers are often interested in reference systems on a lower geographic level, since this provides a more direct link to political goals. In this study, we accounted for the group of 28 European countries, formed by the 25 countries of the European Union in 2006, supplemented with Iceland, Norway and Switzerland, as an alternative reference system besides the world economic system. This European system will be further referred to as EU_{25+3}.

With respect to demarcation in time, different alternatives exist. In the most usual approach, emissions that have occurred during the reference year are used as the reference emissions. In an alternative, approach, the reference system is not the year of emission as such, but the year during which the emission has been caused initially. This last approach accounts for the delay between production and use on the one hand and waste treatment and emission on the other, that occurs for certain emissions. While the first approach has the practical advantages of relative simplicity, transparency and feasibility, the second approach is the most correct one in a theoretical sense, since this approach creates a reference system that corresponds to the product-oriented LCA approach. Differences will be negligible for processes with a small time delay or a constant character, but may become distinct if production processes with long-lived emissions show abrupt changes (*e.g.* CFC production) or for future processes handling current products as waste and differing substantially from current waste treatment processes.

In this study, we have headed for a combination of theoretical correctness and practical feasibility. This implies that the second approach is used in those cases where the difference between both approaches is clear and information of future emissions, caused by products of the year 2000, can be obtained with reasonable reliability. For this reason, emissions of ozone-depleting substances are set equal to their production rates in the year 2000, and emissions of the anti-fouling agent tributyltin (TBT) are set equal to releases, estimated to result from TBT that has been applied in the year 2000. For both types of chemicals, these emissions are much lower than the actual emission level in 2000, which is largely caused by leakage from formerly produced products. For processes of which future emissions are unknown, emission levels in the year 2000 are used as a ‘best estimate’. This is the case for emissions from waste incineration and landfill seepage.

6.2.2 Units

The assessment unit in LCA – a *functional unit* of product –is usually expressed in terms of a certain *amount* of product. This implies that emissions and extractions in LCA are also most commonly expressed in these terms: they are emission and extraction amounts (in kg), rather than the continuous emission flows (in kg/year) that form the direct link between economic activity and environmental impacts or risks.

If normalisation is considered as an LCA study – as we propose – it seems almost self-evident that outcomes of the normalisation step should be expressed in terms of fractions. This is not, however, common practice. Heijungs (2005) explicitly defends the use of time units such as years – implying that the reference interventions in normalisation should be expressed in units per year. The apparent advantage of this last approach is the fact that the normalisation outcomes will be independent of the (arbitrarily chosen) time period to which the reference interventions in normalisation apply. We have chosen to follow the approach as proposed by Heijungs (2005), which theoretically implies that in this study, both the functional unit (the economic system) and the corresponding interventions should be considered as rates, rather than the more commonly used absolute magnitudes.

Following this line of reasoning, multiplying emission and extraction magnitudes by the corresponding LCA characterisation factors for each LCA impact category gives:

$$A_{e,s} = \sum_i \sum_x Q_{e,x,i} \times M_{x,i,s} \qquad (6.1)$$

where $A_{e,s}$ (*e.g.* in kg-eq./yr) is the normalisation factor for impact category e in reference system s, $Q_{e,x,i}$ (*e.g.* in kg-eq./kg) is the characterisation factor related to impact category e for substance x released to or extracted from environmental compartment i, and $M_{x,i,s}$ (*e.g.* in kg/yr) is the release or extraction of substance x to compartment i in reference system s.

6.2.3 Impact category classes

In LCA characterisation, a distinction is made between two classes of impact categories: the so-called *midpoint* and *endpoint* categories, respectively (Jolliet *et al.*, 2004). Midpoint categories represent homogeneous classes of impacts with respect to effect type or mechanism. These categories largely correspond to the focus points of environmental policy. *Climate change*, *acidification* and *aquatic ecotoxicity* are examples of impact categories at the midpoint level. Endpoint categories are formulated on a more functional basis: they represent the ultimate environmental values, eventually damaged by anthropogenic interventions. In this study, we focus on the calculation of normalisation scores for impact categories on the midpoint level. Table 6.2 gives an overview of the impact categories considered, and of the characterisation methods used for the different categories. Most of these characterisation factors are updates of the characterisation factors, published in the *Handbook on Life Cycle Assessment* by Guinée *et al.* (2002). The new factors are to be included in the new impact assessment method ReCiPe (Heijungs *et al.*, 2003).

Table 6.2 Overview of the impact categories covered by this study.

impact category	characterisation factor	unit of CF	ref.
climate change	global warming potential	kg CO_2-eq./kg	[1]
ozone depletion	ozone depletion potential	kg CFC-11-eq./kg	[2]
acidification	acidification potential	kg SO_2-eq./kg	[3,4]
freshwater eutrophication	freshwater eutrophication potential	kg P-eq./kg (to freshwater)	[5]
marine eutrophication	marine eutrophication potential	kg N-eq/kg (to freshwater)	[5]
human toxicity	human toxicity potential	kg 1,4-DCB eq./kg (to urban air)	[6-10]
photochemical oxidant formation	photochemical oxidant formation potential	kg NMVOC-eq./kg	[11,12]
particulate matter formation	particulate matter formation potential	kg PM_{10}-eq./kg	[11,12]
terrestrial ecotoxicity	terrestrial ecotoxicity potential	kg 1,4-DCB eq./kg (to industrial soil)	[6-10]
freshwater ecotoxicity	freshwater ecotoxicity potential	kg 1,4-DCB eq./kg (to freshwater)	[6-10]
marine ecotoxicity	marine ecotoxicity potential	kg 1,4-DCB eq./kg (to seawater)	[6-10]
ionising radiation	ionising radiation potential	kBq U-235 eq./kg (to air)	[13]
agricultural land occupation	land occupation potentials	–	[14]
urban land occupation	land occupation potentials	–	[14]
fossil energy resource depletion	fossil energy resource depletion potential	kg oil eq./kg (nat. gas: MJ/kg oil eq./m^3)	[15]

[1] IPCC (2001); [2] WMO (1999); [3] Van Zelm *et al.* (2007b); [4] Van Zelm *et al.* (2007c); [5] Struijs *et al.* (2007); [6] Huijbregts *et al.* (2005b); [7] Huijbregts *et al.* (2005a); [8] Van de Meent & Huijbregts (2005); [9] Van Zelm *et al.* (2007a); [10] Huijbregts *et al.* (2007); [11] Van Zelm *et al.* (*in press*); [12] Van Zelm *et al.* (2007d); [13] Frischknecht *et al.* (2001); [14] De Schrijver and Goedkoop (*in press*); [15] Frischknecht *et al.* (2004).

6.2.4 Boundary between inventory and characterisation

Normalisation largely consists of a two-step procedure: *inventory analysis* of environmental interventions (Rebitzer *et al.*, 2004) and numerical *characterisation* of these interventions by multiplying them with the characterisation factors in the context of the applicable impact categories (Pennington *et al.*, 2004). Although environmental fate analysis is considered as a part of the LCA characterisation, it is not always clear where the inventory analysis ends and the characterisation starts – that is, in which terms the interventions should be expressed. Ammonia emissions from manure and fertiliser to air, reported extensively in literature, are in this study considered as emissions of nitrogen compounds to soil, with subsequent volatilisation in the field situation. Emissions of nitrous oxide – again reported extensively in emission databases – result from environmental fate processes in the soil after the application of manure or fertiliser. In this study, these emissions are accounted for as direct emissions, since the process of denitrification is not yet included in the characterisation factors used. In general, we accounted for fate in the inventory analysis only if the fate process concerned was not included in the characterisation factors (see Table 6.3).

Table 6.3 Level of emissions at the boundary between *inventory* and *characterisation*.

substance	emission source	original release compartment	inventory level emission compartment	possible subsequent (characterisation level) residence compartments
ammonia/ ammonium	manure, fertiliser	air or agricultural soil	agricultural soil	air
nitrous oxide	manure, fertiliser	agricultural soil (as N)	air	–
minerals	manure, fertiliser	agricultural soil	agricultural soil	freshwater, crops
heavy metals	manure, fertiliser	agricultural soil	agricultural soil	freshwater, crops
pesticides	direct application	agricultural soil	agricultural soil	air, freshwater

6.2.5 Boundary between economic system and environment

When soils are used for economic activities, the boundary between the economic and the environmental system is not always clear cut. For normalisation, we need to define the environmental soil system in order to be able to distinguish between soil emissions and transfer processes within the economic system. For industrial soil, we considered landfills as part of the economic system, but the underlying soil as part of the environmental system. Agricultural soil was assumed to be part of the environmental system, but crops were considered as part of the economic system. This corresponds with the calculation routine for fate and exposure factors of toxic pollutants, as elaborated in Huijbregts *et al.* (2005b).

6.2.6 Emissions, caused by physical interventions

Besides resource extraction, production, use, and waste treatment, physical interventions too are part of many product life cycles. Physical interventions include forest clearing, mine exploitation and other landscape-transforming processes that are performed for the sake of economic production. Besides direct landscape transformation, these interventions can also cause emissions. While the emission of carbon dioxide by forest burning is easily recognised, more hidden, long-lasting emissions may occur as well, especially as a result of transformed soil physics and chemistry. These emissions include increased levels of soil carbon dioxide release (Marland and Schlamadinger, 1995; Schulze *et al.*, 2000) and increased phosphorous compound loss as a result of a boosted soil erosion level (Smil, 2002). The magnitude of this type of emissions is expected to be considerable, but estimates are still uncertain. The question in which way such interventions should be allocated to an LCA functional unit has been discussed by Reijnders and Huijbregts (2003). For reasons of present uncertainty on the emission magnitudes, emissions that are caused by physical interventions have not been included in this study.

6.3 Guidelines for data source prioritisation and data estimation

6.3.1 Procedures

It occasionally happens that different literature resources are available on the same releases and extractions. In these cases, it is necessary to make a choice between different data sources. Additionally, data on environmental releases and extractions are most often not available in the form or at the geographical scale that exactly corresponds to the specific needs for an LCA normalisation study. Therefore, the use of data estimates is inevitable in LCA normalisation.

For reasons of consistency, it is important to use explicit criteria for the prioritisation of data sources and the estimation of data. The criteria developed for use in this project can be considered as a proposal for the establishment of a general set of guidelines for data source prioritisation and data estimation in LCA normalisation. A detailed overview of impact categories, intervention data collected, data types used, data estimation procedures applied, and literature resources consulted, can be found in the *Supporting information* (part 1).

6.3.2 Data source prioritisation

The following criteria for data source prioritisation have been used in this study, and are proposed as general guidelines:

1. Authorativeness of literature sources

Authorative literature sources were preferred to less authorative sources. Independent scientific journals were considered as the most authorative data sources, followed by databases provided by independent, large international organisations, particularly if the databases concerned were updated regularly and were generally recognised as a basis for international environmental policy.

2. Number of emission sources covered

Data covering a large proportion of emission source categories were preferred to data which covered only a limited number of source categories.

3. Measured values versus scenario estimates

Measured values were preferred to scenario estimates.

4. Measured values versus expert estimates

Expert estimates on the basis of measured values were preferred to measured values as such.

5. Independency of data sources

Independently collected data was preferred to data, reported by stakeholders.

6. Number of countries covered
Data sources, covering a large number of countries, were preferred to sources with a smaller spatial coverage.

7. Normalisation as a 100% reference
Data, linked to emissions *caused by* production in the reference year, chosen for the normalisation at stake, were preferred to emission data as such for the same year.

6.3.3 Emission factors

Emission factors can be used if the relationship between the magnitude of a source and the release from this source is known or can be derived. For some source-substance combinations, emission factors are readily available. In this study, we used ammonia emission factors for different types of fertiliser, as defined by FAO (2006a), Bouwman *et al.* (1997, 2002), and IFA (2006), and an emission factor for the release of the anti-fouling agent tributyltin (TBT) from ship surfaces (Van de Plassche and Van der Aa, 2004). If emission factors are not readily available, they can be constructed from information on the magnitude and composition of releases. In this study, we constructed emission factors for the release of ammonia from animal manure and for the release of phosphorous compounds and heavy metals from both animal manure and fertilisers on the basis of livestock statistics (FAO, 2006b), manure excretion data (Bouwman *et al.*, 1997, Van Bruggen, 2004), the use of fertiliser and manure (FAO, 2006a,b,c) and the assumed heavy metal content of manure and fertilisers (Delahaye *et al.*, 2003).

6.3.4 Production and consumption data

If emissions and emission factors are unknown, production or consumption data can be used instead. This is only possible for chemicals of which the largest part of production of consumption will eventually be released to the environment. In this study, we used consumption data of CFCs, HCFCs, halons, 1,1,1-dichloroethane, and tetrachloromethane (UNEP, 2002) as an approach for the corresponding (future) emissions. For aldrin, dieldrin, DDT and cyclohexatin, we used production data from a UNEP/FAO study (1995) as estimates for global emissions to agricultural soil. For bromomethane, consumption data (UNEP, 2002) was used as an estimate for the emission to agricultural soil.

6.3.5 Estimation factors

If emissions and emission factors are unknown and production or consumption data are not applicable, so-called estimation factors can be used instead. In contrast to emission factors, estimation factors are not applied to individual emission source types, but directly to emission totals. Estimation factors should preferably bear a clear functional relationship to the emission or extraction to be estimated. Depending on the type of emission or extraction to be assessed, many different types of estimation factors are possible. In this project, we used the following estimation factors:

- *GDP* (gross domestic product), assumed to be related to industrial production and release
- *crop production area*, assumed to be related to crop-dependent ammonia emissions and pesticide use
- *population magnitude*, assumed to be related to human waste production of N and P
- *nuclear power capacity*, assumed to be related to the release of radio-active substances
- *estimated length of polluted coastline*, assumed to be related to metal releases in the marine environment

The estimation factors were applied to extrapolate or interpolate in space or time.

6.3.6 Estimation types

Space

Extrapolation in space can be used if information is available on just a part of the assessment region. As an example, information on emissions or extractions in a number of European countries can be extrapolated to the level the European region as a whole, and information on emissions and extractions in a number of individual countries can be extrapolated to the global level. In this study, extrapolation in space was mainly applied to industrial emissions of toxic chemicals, in particular on the global scale. The national PRTRs (Pollutant Release and Transfer Registers) of the United States, Canada, Japan and Australia (US-EPA, 2006; EC, 2006; NITE, 2006; AG-DEH, 2006) and the European emission databases EMEP (2006) and EPER (2006) were the most important data sources used for this purpose.

Interpolation in space can be used if information is available on a higher spatial level than the assessment region. As an example, information on emissions or extractions on the global level can be interpolated to the level of the European region. In this study, data on the emissions of HFCs on the European scale was interpolated from global scale data (AFEAS, 2006), while data on the emissions of phosphorous compounds to freshwater on the European scale was interpolated from the global scale information, provided by Smil (2002).

Time

Extrapolation in time can be used if information is available on an earlier year than the assessment year. In this project, we extrapolated air emissions of ammonia in the year 1990 to the reference year 2000.

Interpolation in time can be used if information is available on a later year than the assessment year, *e.g.* on emissions and extractions in the year 2005 instead of 2000. In this project, we did not use interpolation in time.

Parallel equivalency assumptions
Parallel equivalency assumptions can be used if information is available on another region than the assessment region, *e.g.* on emissions and extractions in the United States instead of Europe. In this project, we used parallel equivalency assumptions to estimate European pesticide emissions from estimated pesticide emissions in The Netherlands (CBS, 2006), the United Kingdom (CSL, 2006), and the United States (Gianessi and Marcelli, 2000; Kiely *et al.*, 2004).

6.3.7 Statistical analysis

The reliability of estimation factors depends on the strength of the correlation between the emission or extraction on the one side and the specific estimation factor on the other. If information is available on a number of emissions/extractions, it can be valuable to calculate correlation factors for two or more alternative estimation factors. In this project, we applied statistical analysis to the emission data of PM_{10} and to pesticide consumption data.

For the emission of PM10, we had emission data to our disposal for most European countries and a small number of Asian countries, provided by EMEP (2006). The correlation between GDP values and PM_{10} emissions for these countries turned out to be rather weak ($r^2 = 0.291$). An analysis of alternative extrapolation bases showed that it was possible to find a better correlation alternative by dividing countries in three different income groups, based on a classification by the World Bank (2000): *high*, *upper-middle* and *lower-middle* plus *low* ($r^2 = 0.544$ by the application of ANOVA).

Outliers were eliminated by use of Chauvenet's criterion (Bevington and Robinson, 1992). Within each income group, the ratio between PM_{10} emission and GDP turned out to be reasonably constant, with large differences between income groups. The ratio between PM_{10}/GDP ratios was 3:20:600 for high, upper-middle and lower-middle plus low income groups, respectively, suggesting that the investments in cleaner technologies are larger in the more prosperous countries. Average PM_{10}/GDP ratios for the 3 income groups were applied to calculate estimates for PM_{10} emissions for all world countries on the basis of the combination of their respective GDP values and income groups.

Specific attention was paid to the estimation of pesticide emissions. FAO (2006d) provides data on the consumption of different pesticide groups for a reasonably large number of European countries. Although these group data could not be used as emission estimates for individual pesticides, they were useful for checking the validity of alternative estimation factors. The correlation between the data for overall pesticide use and GDP values turned out to be rather weak ($r^2 = 0.318$), suggesting that another variable might be more determinative. Crop production showed a better correlation with total pesticide use ($r^2 = 0.507$). Still better became the image when we used a more specific approach by dividing pesticides into different classes, and calculated correlations separately for each class. For the use of different classes of pesticides, we analysed correlation coefficients between pesticide use on the one hand and GDP values and crop production areas on the other, respectively. It turned out that for insecticide consumption, the correlation coefficient with the crop production area ($r^2 = 0.62$) was much higher than the correlation coefficient with the GDP value ($r^2 = 0.075$), while for herbicide, fungicide and bactericide, and other pesticide consumption, it was the other way around: higher correlation coefficients with GDP values ($r^2 = 0.701$, $r^2 = 0.336$ and $r^2 = 0.562$, respectively) than with crop production areas ($r^2 = 0.67$, $r^2 = 0.163$ and $r^2 = 0.101$, respectively). Outliers were eliminated by use of Chauvenet's criterion (Bevington and Robinson, 1992) for all correlation calculations. Estimation factors were chosen in correspondence with the results of the correlation analyses for the individual pesticides in each of the pesticide classes. The use of individual pesticides on both the global and the European scale was derived from pesticide consumption data in The Netherlands (CBS, 2006), the United Kingdom (CSL, 2006), and the United States (Gianessi and Marcelli, 2000; Kiely *et al.*, 2004).

6.4 Results

A total number of 860 environmental interventions were collected for both the European and the global system: 854 emission totals, 4 fossil energy resource extraction totals and 2 land occupation totals. An emission total consists of the summed emissions of a substance to a specific compartment in one of the geographic systems. Fossil energy resource extraction totals refer to summed extractions of a certain fossil energy resource. Land occupation totals comprise the summed area occupied for a certain type of use. The results of LCA normalisation are acquired by multiplication of each reference intervention for each environmental system with the applicable characterisation factors for all impact categories. The resulting normalisation factors are listed in Table 6.4. This table also gives an overview of the relative environmental load of the European economic system, compared to the world economic system, for each separate impact category.

Table 6.4 Normalisation factors.

impact category	unit	EU$_{25+3}$	world	EU$_{25+3}$ (% of world)
climate change				
TH = 20 yr	kg CO$_2$ eq.	6.57E+12	5.76E+13	11
TH = 100 yr	kg CO$_2$ eq.	5.21E+12	4.18E+13	12
TH = 500 yr	kg CO$_2$ eq.	4.49E+12	3.36E+13	13
ozone depletion	kg CFC-11 eq.	6.79E+06	2.10E+08	3
acidification				
TH = 20 yr	kg SO$_2$ eq.	2.23E+10	3.01E+11	7
TH = 100 yr	kg SO$_2$ eq.	2.36E+10	3.18E+11	7
TH = 100 yr	kg SO$_2$ eq.	2.49E+10	3.36E+11	7
TH = 500 yr	kg SO$_2$ eq.	2.84E+10	3.78E+11	8
freshwater eutrophication	kg P eq. (to freshwater)	3.47E+08	3.77E+09	9
marine eutrophication	kg N eq. (to freshwater)	5.89E+09	5.71E+10	10
respiratory effects				
photochemical oxidant formation	kg NMVOC eq.	2.80E+10	3.51E+11	8
particulate matter formation	kg PM$_{10}$ eq.	8.12E+09	9.92E+10	8
human toxicity				
TH = 100 yr	kg 1,4-DCB eq. (to urban air)	1.24E+11	1.20E+12	10
TH = infinite	kg 1,4-DCB eq. (to urban air)	2.27E+12	8.86E+12	26
freshwater ecotoxicity				
TH = 100 yr	kg 1,4-DCB eq. (to freshwater)	5.83E+09	2.94E+10	20
TH = infinite	kg 1,4-DCB eq. (to freshwater)	6.03E+09	3.07E+10	20
marine ecotoxicity				
TH = 100 yr	kg 1,4-DCB eq. (to seawater)	8.98E+09	2.85E+10	32
TH = infinite	kg 1,4-DCB eq. (to seawater)	1.78E+12	6.24E+12	29
terrestrial ecotoxicity				
TH = 100 yr	kg 1,4-DCB eq. (to industrial soil)	4.07E+09	3.72E+10	11
TH = infinite	kg 1,4-DCB eq. (to industrial soil)	6.37E+09	5.09E+10	13
ionising radiation	kBq U-235 eq. (to air)	2.90E+12	7.97E+12	36
agricultural land occupation	m^2×year	2.10E+12	3.30E+13	6
urban land occupation	m^2×year	1.89E+11	4.71E+12	4
fossil energy resource depletion	kg Sb eq.	7.23E+11	7.78E+12	9

An overview of the most important environmental interventions with respect to each impact category is shown in Table 6.5 and Table 6.6, which list all interventions that contribute to a joint share of at least 75% of the total result. Total contributions of important substance groups are listed as well. For impact categories with different sets of factors for different time horizons, a default time horizon was chosen for this table: 100 years for climate change, 500 years for acidification, and 'infinite' for al toxicity-related impact categories. A complete representation of all data for all impact categories and time horizons can be found in the *Supporting information* (part 2). It turns out that a relatively small number of interventions are responsible for the dominant results: 52 emission totals, 4 fossil energy resource extraction totals, and 2 land occupation totals.

Table 6.5 Contributions of individual impacts to normalisation factors for the EU25+3/world: emission related impact categories (TH = time horizon).

substance	emission compartment	climate change (TH 100 years)	ozone depletion (infinite TH)	acidification (TH 500 years)	freshwater eutrophication	marine eutrophication	photochemical oxidant formation	particulate matter formation	human toxicity (infinite TH)	freshwater ecotoxicity (infinite TH)	marine ecotoxicity (infinite TH)	terrestrial ecotoxicity (infinite TH)	ionising radiation
CO_2	air	77/68											
CH_4	air	10/18											
HCFC-22	air		41/7										
CFC-12	air		26/48										
CFC-11	air		11/19										
Halon-1211			2/14										
HCFCs (total)	*air*		*57/10*										
CFCs (total)	*air*		*39/70*										
halons			*3/20*										
SO_2	air			31/35				22/26					
NO_x (as NO_2)	air			30/22		27/27	42/33	32/26					
N from pig manure	agricultural soil			8/3									
N from dairy cattle manure	agricultural soil			7/3									
NH_3 (direct)	air			3/12									
manure-N (total)	*agricultural soil*			27/20		*13/15*		*11/9*					
fertiliser-N (total)				9/11		*14/11*							
NH_3 (total)	*air/agricultural soil*			40/44		*28/29*		*14/17*					
P-total	freshwater				66/80								
P-total	agricultural soil				34/20								
N-total (direct)	freshwater					47/43							
N from non-specified fertiliser	agricultural soil					6/1							
urea	agricultural soil					3/6							
NMVOC	air						42/48						
PM_{10}	air							31/29					
selenium	air								31/21		17/13		
lead	air								17/9				
arsenic	air								15/12				
mercury	air								5/3				
zinc	agricultural soil								4/13			5/7	
barium	industrial soil								3/3				

substance	emission compartment	climate change (TH 100 years)	ozone depletion (infinite TH)	acidification (TH 500 years)	freshwater eutrophication	marine eutrophication	photochemical oxidant formation	particulate matter formation	human toxicity (infinite TH)	freshwater ecotoxicity (infinite TH)	marine ecotoxicity (infinite TH)	terrestrial ecotoxicity (infinite TH)	ionising radiation
bromomethane	air								2/8				
cadmium	agricultural soil								3/7				
chlorine	freshwater									68/60			
atrazine	agricultural soil									15/10		17/8	
nickel	air										22/11		
copper	air										16/23	6/4	
vanadium	air										16/17		
zinc	air										6/3	5/1	
cypermethrin	agricultural soil									3/7		16/26	
copper	marine water										3/3		
nickel	agricultural soil										3/3		
copper	agricultural soil										3/3	9/2	
copper	industrial soil											5/2	
aldicarb	agricultural soil											5/7	
chloropicrin	agricultural soil											4/6	
terbufos	agricultural soil											4/6	
pesticides (total)	*agricultural soil*								*1/8*	*24/30*		*59/68*	
carbon^{14}	air												73/73
cesium^{137}	freshwater												25/25

Data in *italics* refer to intervention groups.

Table 6.6 Contributions of individual impacts to normalisation factors for the EU$_{25+3}$/world: non-emission related impact categories.

land use intervention or substance extracted	intervention type	agricultural land occupation	urban land occupation	fossil energy resource depletion
agricultural land occupation	agricultural land use	100/100		
urban land occupation	urban land use		100/100	
natural gas	fossil energy resource depletion			37/31
crude oil	fossil energy resource depletion			38/44
hard coal	fossil energy resource depletion			14/23
lignite	fossil energy resource depletion			11/2

For *climate change*, CO_2 and CH_4 together account for around 85% of the effect on both scales. CO_2 alone is responsible for almost 70% of the impact on the global scale and for more than 75% of the impact on the European scale.

For *ozone depletion*, CFC emissions are responsible for 66% of ozone depletion on the global scale in the year 2000. Halons are the second cause, with a contribution of 19%. On the European scale, however, HCFCs take the leading role with a contribution of 65%, and the contribution of halons is less than 5%.

Characterisation factors for *acidification* on both the global and the European scale are defined for SO_2, NO_x, and NH_3. Each of these substances contributes significantly. Emissions of NH_3 have been split up between different sources: Emissions of all different types of manure and fertiliser are considered separately, and distinguished from direct ammonia emissions from other sources. Manure and fertilisers together form the largest sources of NH_3 on both scales. Total NH_3 emissions form the dominant cause of acidification on both scales.

Eutrophication is split up into *freshwater* and *marine eutrophication*. While *freshwater eutrophication* is caused by emissions of phosphorous compounds, either directly to the freshwater compartment or to agricultural soil, *marine eutrophication* is caused by a number of different nitrogen compounds to different compartments. Human waste emissions of phosphorous compounds and point emissions of nitrogen compounds to the freshwater compartments are identified as the dominant causes for freshwater and marine eutrophication, respectively.

Photochemical oxidant formation is caused for more than 80% by the combination of non-methane volatile organic chemicals (NMVOCs) and NO_x in both geographic systems. *Particulate matter* formation is dominated not only by the direct emission of PM_{10} (particles with a diameter of 10 µm or less), but also to a considerable extent by the emissions of SO_2, NO_x and NH_3 in both systems.

Human toxicity impacts are fully dominated by heavy metals in both geographic systems. Air emissions of heavy metals are specifically large contributors to human toxicity, with selenium air emissions as the largest contributor for both the global (21%) and the European (31%) system. Apart from heavy metals, only pesticide emissions to agricultural soil contribute for more the 5% to one of the environmental profiles: 8% for the global system. This last contribution is almost fully determined by bromomethane.

For *freshwater ecotoxicity*, chlorine emissions to the freshwater compartment play a dominant role with a contribution of more than 50% for both systems. Pesticide emissions to agricultural soil are the other important factor. *Marine ecotoxicity* is fully dominated by emissions of heavy metals, largely to air. *Terrestrial ecotoxicity* is dominated by pesticide emissions to agricultural soil.

For *ionising radiation*, emissions of carbon-14 are fully dominant, with a contribution of 73% to the total impact on both geographical scales.

For LCA characterisation and normalisation purposes, the impact categories of agricultural and urban land occupation have been split up here. On the global scale, direct data for both categories was available from the World Resources Institute (WRI, 2000). On the European scale, these categories have been composed from classes in the CORINE system (EEA, 2006).

Finally, for *fossil energy resource depletion*, the extractions of natural gas and crude oil form the determinative impacts.

6.5 Discussion

6.5.1 Interpretation of results

The results of this study show that only a relatively small proportion of the total number of interventions is responsible for a large proportion of potential environmental impacts. From the 858 environmental interventions collected, 46 interventions account for at least 75% of the impact scores of all impact categories. All non-toxicity related, emission dependent impacts are fully dominated by the bulk emissions of 10 substances or substance groups: CO_2, CH_4, SO_2, NO_x, NH_3, CO, PM_{10}, NMVOC, and(H)CFCs emissions to air and emissions of N- and P-compounds to freshwater.

Comparing our results with the study of Huijbregts *et al.* (2003), it was found that for climate change, acidification and resource extraction, the overall results for year 2000 have remained largely the same as for 1995. For ozone depletion, the toxicity-related impact categories and ionising radiation, the results have substantially changed. The total impact score for ozone depletion has been largely reduced between 1995 and 2000. CFC emissions still dominate the global normalisation score. On the European level, however, HCFCs have taken over the leading role of CFCs as the dominant cause of ozone depletion. It should be remarked, however, that the normalisation results for 1995 and 2000 are not fully comparable. Especially for these ozone-depleting emissions, the methodological choice to shift from emissions, taking place in the reference year to emissions, caused by production in this year, results in a considerable reduction.

For eutrophication and photochemical oxidant formation, the results between the two studies are difficult to compare, because the characterisation method has been adapted for these impact categories. Particulate matter formation is a new impact category that was not accounted for in the 1995 study.

The present human toxicity image, with a dominant role for heavy metal emissions, is different from the 1995 image, in which non-halogenated hydrocarbons were fully dominant. For aquatic ecotoxicity, chlorine emissions to freshwater, that dominate the results in the present study, were not accounted for in 1995, and pesticide emissions to agricultural soil then dominated the impact score. For marine ecotoxicity, heavy metal emissions were dominant as well in 1995, but differences exist in the relative importance of individual metals, and direct emissions to seawater played a more important role. For terrestrial ecotoxicity, heavy metal and pesticide emissions to agricultural soil changed places between 1995 and 2000: a dominant role for the pesticides and a secondary role for heavy metals in the present study, and the inverse image in the 1995 study. Here too, the relative importance of individual metals differs between the 1995 and the 2000 studies. The large differences between 1995 and 2000 results indicate that the uncertainties for the toxicity-related impact categories are probably very high. For these impact categories, the dominance of an emission should be considered only as an indication that further investigation could be interesting. Whether the dominant emissions (*e.g.* chlorine emissions to freshwater and bromomethane emissions to agricultural soil) actually play a dominant role in potential environmental toxicity is a question for more specific research.

For ionising radiation, the results for year 2000 are similar to the results for the year 1995: a dominant role for the emission of carbon-14 to air.

With respect to the LCA results of both economic systems as a whole, it turns out that the European economic system is responsible for relatively large contributions to most of the toxicity-related impact categories, and to ionising radiation. For ionising radiation, this is due to the relatively high nuclear capacity of the European region, which has been used as an estimation parameter for emissions. For the toxicity-related impact categories, the ratios found are most probably a reflection of the percentage of 27% of world GDP for which the European region is responsible, since GDP has been used as an extrapolation factor for many toxic emissions. This finding emphasizes the relatively high uncertainty of the normalisation factors for toxicity in both Europe and the world. Considering the relatively high European GDP, the European contribution to other impact categories – mostly around 10% or less – is rather low. For the emission-related impact categories, this may be due to the use of cleaner technologies in this region.

6.5.2 Uncertainty

Uncertainty in normalisation results is caused by a combination of uncertainty in emission data and uncertainty in characterisation factors. Establishing quantitative estimates of uncertainty in normalisation factors is considered out of the scope of this study, as uncertainty margins for most emissions, extractions and characterisation factors were not readily available. Instead, we focus on a qualitative assessment of the most important uncertainties in the normalisation factors.

The highest level of uncertainty applies to emission data that is extrapolated from emissions in a limited number of countries. This is the case for virtually all data that refer to *toxicity*-related impact categories, to the impact category *ionising radiation*, to emissions of the substances 1,1,1,-trichloroethane (methyl chloroform) and tetrachloromethane (carbon tetrachloride) – that contribute to *ozone depletion* – and to emissions of PM_{10} – that is important in the context of the impact category *particulate matter formation*. An intermediate level of uncertainty is expected to apply to all other types of estimates that are based on a combination of reported data and estimation factors: emission estimates for CFCs, HCFCs and halons, European emission estimates for HFCs, estimated point emissions of nitrogen compounds, and emission estimates for phosphorous compounds. The lowest level of uncertainty is assumed to apply to the emission and extraction data, taken directly from official data reports: European data on *climate change*, European SO_2 and NO_x emission data, land occupation data, and resource depletion data. Emission estimates or scenarios, derived from reported emission data in earlier years or calculated from manure and fertiliser application, are also considered to have a relatively low level of uncertainty: global, non-European data on *climate change* and on SO_2 and NO_x emissions, as well as all NH_3, CO and NMVOC emission data.

Apart from emission data, characterisation factors can also be a source of uncertainty for normalisation. The largest uncertainties with respect to characterisation factors probably apply to fate modelling of heavy metals, resulting in uncertainty in the corresponding toxicity-related characterisation factors (Heijungs *et al.*, 2004). Since the emission data for heavy metals have a high factor of uncertainty as well, the relative importance of the role of heavy metals with respect to different toxicity-related impact categories is still very difficult to assess.

The most important overall cause of uncertainty in emission data is most probably the absence of toxic release inventories in most countries. The absence of characterisation factors for some substances further adds to uncertainty of the results of normalisation.

Expansion of emission registration by international policy can be considered as one of the most effective measures to diminish uncertainty. The foundation of an Intergovernmental Panel on Chemical Pollution, as recently suggested by Scheringer (2007), would be a way to implement this in a formalised and unambiguous way. An independent reviewing procedure for all types of release data could improve data quality and thus further reduce uncertainty.

6.5.3 Bias

Systematic over- or underestimation are considered as bias, rather than uncertainty. Heijungs *et al.* (2007) pointed out that data gaps in the inventory or in the list of characterisation factors may cause not only uncertainty, but also bias in normalisation. The authors give a systematic overview of all possibilities of over- or underestimation, according to the type of data that is lacking.

In this study, we had to deal with two types of data gaps: missing characterisation factors for expectedly toxic substances on which emission data was available, and missing emission data for potentially toxic substances that were included in the list of characterisation factors. Despite a large increase in the number of available characterisation factors for toxic impacts, data gaps are still considerable. 30% of the total load of emissions to air and freshwater, reported by the national PRTRs (Protocols on Pollutant Release and Transfer Register) of Canada, Japan and the United States (EC, 2006; NITE, 2006; US-EPA, 2006) could not be covered by characterisation factors for at least one toxic impact category. Almost all reported emissions are expected to refer to potentially toxic substances. As a result of this gap in toxicity characterisation factors, both characterisation and normalisation will often apply to a limited part of total emission – depending on the exact emission profile of the product under study. This should be kept in mind if conclusions are drawn from the results. Especially if a substance causes a high impact score in characterisation, while this substance has not been accounted for in the normalisation factor, the normalisation score may become disproportionally high. This bias can be prevented either by leaving the characterisation score of the substance out of the impact score in the normalisation step, or by the (*ad hoc*) construction of a new normalisation factor that includes the total environmental emission of the substance at stake.

A possible cause of bias that has not been mentioned by Heijungs *et al.* (2007) is connected to the demarcation of non-global reference systems. Geographical boundaries do not automatically correspond to the system boundaries in LCA. The life cycle of a European product, for instance, will often exceed the geographic European boundaries. In other words: the environmental impacts, connected to the European *ecological footprint* (Wackernagel and Rees, 1996) will probably be different from the environmental impacts occurring within the European geographic boundaries. If normalisation should be consistent with the LCAs of the composing product systems, it should reflect the impacts, connected to this ecological footprint, rather than to the European geographic region. This issue has been addressed by Breedveld *et al.* (1999), who defined normalisation factors for the Netherlands on two different bases: geographic boundaries versus final consumption – representing 'all environmental interventions in one year (1993/1994) related to the consumption of Dutch end consumers, including the total chains of production and waste processes that result from this consumption'. In the present study, we used the geographic boundaries as system boundaries. A possible future implementation of the 'ecological footprint' principle could be reached by the use of input–output models (*e.g.* Hubacek and Giljum, 2003; Suh, 2004).

The methodological choice to prefer emissions, caused by production in the year 2000, above emission during the year 2000 as such, can be an extra cause of bias. Future emissions of CFCs, HCFCs, and halons, are presumed to equal consumption in the year 2000, while in reality, combustion or destruction are other possibilities to handle these substances. Recycling of 'old' CFCs and HCFCs in the year 2000 and the question how this should be analysed and allocated further adds to methodological complexity and further risk of bias. Future emissions, caused by usage and waste treatment of products, were presumed to equal the equivalent emissions in the year 2000, with only a few exceptions for mainly (H)CFCs. This assumption may be biased, especially for those substances that are largely emitted by usage and waste treatment related processes. For the majority of substances, however, differences between both emission types are expected to be relatively small.

6.5.4 Perceived and actual versus potential and prolonged

The results of an LCA study do not always reflect the ratios of perceived environmental problems. This becomes specifically clear if not just one product, but the economic system as a whole is subjected to the analysis. High impact scores may be connected to substances that are perceived as being not that problematic, while substances that are known to be connected to real environmental problems may receive amazingly low scores in the LCA environmental profile. This phenomenon is caused by two specific properties of LCA characterisation: first, the convention to assess potential environmental impacts, and second, the integration of environmental concentrations over time.

The concept of potential problems – or the 'less is better approach' – refers to the fact that emissions in LCA are assessed on the basis of a combination of potential harmfulness, environmental residence time and – if applicable – overall human intake. Environmental concentrations or average daily intake rates cannot be coupled to the summed emission totals that LCA characterisation is about. LCA therefore leaves risks and average human intake rates out of consideration. Environmental emissions are directly combined with environmental residence times to deliver time-integrated exposure totals. This makes LCA results independent of dilution, not only in the spatial sense, but also in terms of time. Environmental threshold values – that are so determinative for risk assessment – are not accounted for in LCA. Low but long-remaining concentrations of potentially toxic substances will deliver the same outcome as concentration peaks, as long as the time-integrated concentrations are the same. In general, this may result in an unexpectedly high LCA impact scores for poorly degradable chemicals, despite their relatively low environmental concentrations. For the normalisation step in an individual case study, it means that normalisation scores for these chemicals, and for chemicals that are released in near-to-threshold concentrations, will be relatively high, compared to their relative risk that they cause. Therefore, the prioritisation of desired environmental measures should not be based on LCA normalisation alone. Normalisation scores form an indication, but should be checked with measurements before conclusions on corresponding risk levels can be drawn.

6.6 Conclusions

This study has been performed in the context of LCA methodology, but the relevance of the results ranges much further. For LCA, this normalisation study is not only an update in the sense that it refers to a time step of 5 years in comparison to our former study (Huijbregts *et al.*, 2003). Much more important is the fact that a large reduction in uncertainty could be reached by the increased data availability, resulting from a growing international awareness of the urgency of emission data reporting. Another important feature of these updated normalisation data is in their correspondence to up-to-date LCA midpoint characterisation factors.

Although intended as a methodological step in LCA, this normalisation study can also be considered as an LCA case study in itself – with the economic system as the functional unit. The results of this study clearly indicate that the environmental profile of the global and European economic systems appears to be dominated to a large extent by only a small number of substances, released in bulk quantities.

Large uncertainties remain to exist with respect to toxic substances. Current toxic release inventories are available for developed countries only. Releases in developing countries are very difficult to estimate, even by approximation. Moreover, characterisation factors are still lacking for many toxic chemicals, most probably because no or insufficient toxicity data is available. This combination makes it impossible to assess the magnitude of overall toxicity of chemicals, released to the environment, and thus to construct reliable normalisation factors for toxicity-related impact categories in LCA. An update of this normalisation study is recommended as soon as either supplemental emission information or new characterisation factors have become available.

With a fast growing world economy, new, strictly controlled procedures for the abatement of bulk releases and the inventory of toxic releases, coupled to the identification of key toxic substances, are recognised to be of prior importance for global environmental management in the context of sustainability.

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