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The Netherlands

Regional LCA in a global perspective

Wegener Sleeswijk, A.

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General discussion and conclusions

7.1 Introduction

This final chapter provides an integration of the previous chapters, which were written as independent journal articles, into one coherent PhD thesis. The topics discussed so far – actual versus potential impacts, life cycle assessment versus risk assessment, metals in the marine environment, regionalisation of fate, exposure and effect models, and normalisation – may seem to be quite different in character. In this chapter, it will be shown that all these topics are in fact intimately related, and that they can be integrated both conceptually and in a practical sense.

First, I will revisit the main arguments of the various preceding chapters, but in a different order, and with cross-references added. Then, I will list the main conclusions and new findings and insights. The chapter also includes some personal views and historical lines.

7.2 Actual versus potential impacts in connection to LCA and RA

The issue of actual versus potential impacts, which takes a central position in chapter 2, has served as a leading theme throughout a large part of this thesis. Potential impacts are based on a chemical's relative toxicity, environmental distribution and human exposure characteristics as such, without accounting for environmental thresholds or environmental sensitivity. In 1995, White *et al.* stated that the assessment of chemicals, based on threshold exceeding, was not applicable to LCA. Although shortly thereafter, this statement became outdated in practice by the work of Hogan *et al.* (1996) and Potting *et al.* (1998), the tradition of assessing potential impacts with LCA has still remained in most LCIA-methods (*cf.* popular impact assessment methods, such as CML 2002 (Guinée *et al.*, 2002), IMPACT 2002 (Pennington *et al.*, 2005), TRACI (Bare, 2002), and Eco-indicator 99 (Goedkoop & Spriensma, 2003)). This is in part due to the fact that environmental impacts in LCA cannot be described in terms of risks (*cf.* Owens, 1997), at least not the type of risks that have traditionally been assessed with conventional risk assessment methods (Udo de Haes & Owens, 1998). Another ground is that envi-

ronmental thresholds and sensitivity are location-dependent issues while most LCA characterisation methods are location-independent.

The fact that LCA cannot be used for the assessment of risks has two main reasons. The first reason is that most industrial processes – and the connected risks – are only partially included in the life cycle of a given product. A power plant, for instance, may cause a number of environmental risks, that may be analysed and quantified in the context of chemical risk assessment. For the production of a product – for example, a can of paint – only a certain amount of electricity is needed, not that of the whole plant. Although both the risks of the power plant and the amount of electricity needed for the production of a can of paint may be well described, neither risk assessment nor LCA provides the context for describing the risk of one can of paint. In more general terms: risks are connected to (continuous) economic processes in their full extent, while LCA accounts for the impacts of partial processes in a product life cycle, for example, 'paint production' in a certain paint factory. This applies not only to upstream processes in the product life cycle, but also to the direct production process itself. This incongruousness can only be solved if we express the contribution by the given can of paint to the risk that is caused by the plant as a whole in terms of time (Heijungs & Guinée, 1994), for instance, during its production time of 2 seconds, the can of paint was responsible for the full risk of the plant. Or, alternatively, during a year, the can of paint was responsible 1/1,500,000 part of the total risk of the plant (with a production capacity of 1,500,000 cans of paint a year). This is a modification that is not part of the conventional risk assessment procedure. The second reason why LCA cannot be used for risk assessment is the fact that upstream processes in the product life cycle are often dedicated to several downstream processes, for example, steel production is dedicated not only to the production of paint cans, but also to the production of cars, dust bins, cutlery and all other kinds of steel objects. The contribution of a paint can to the environmental risks, caused by a steel production plant, could again be expressed in terms of time – *e.g.*, the time that the full process is needed for the production of the amount of steel, used for the production of one paint can – but it should be noted that the full risk is connected not only to paint cans, but to many other products as well.

The location-independent character of traditional LCA – not allowing for the inclusion of any location dependent threshold exceeding – has raised the question whether LCA could make any sense at all in the context of toxicity assessment (*cf.* Owens, 1997). This issue has been discussed in terms of 'less is better' versus 'only above threshold' (White *et al.*, 1995) and 'general prevention' versus 'risk minimisation' (Barnthouse *et al.*, 1997). Potential impacts in LCA are based on the 'general prevention' principle that the release of toxic chemicals into the environment is undesirable and should be prevented per se, whether or not concentrations exceed environmental threshold values. Although I agree with this statement, I feel that concentrations that do exceed thresholds should be handled with priority; they

should therefore be recognisable in LCIA, and get more emphasis in the assessment. This is the reason for which I worked out a method to distinguish between potential and actual impact in LCA toxicity assessment.

My ideas on risk assessment versus LCA and actual versus potential impacts are reflected in several chapters of this thesis. In chapter 2, a conceptual framework is described for the inclusion of contributions to risks in LCA. These contributions are together considered as the summed 'actual impacts', as opposed to the conventional 'potential impacts'. A detailed discussion on the similarities and differences between LCA and human and environmental risk assessment (HERA) can be found in chapter 3. The conclusion from this chapter is that the functional unit, as the only fundamental difference between LCA and HERA, forms the background of the fact that summed 'actual impacts' or 'risk contributions' in LCA remain different from the 'full risks' assessed with HERA, but that both could be combined in a common software tool that produces both HERA and LCA results in a consistent fashion. Chapter 5 contains a description of how the method for the assessment of 'actual impacts', proposed in chapter 2, has been implemented in the GLOBOX model. To this end, two new factors have been introduced: a *sensitivity factor* (SF) and a *threshold factor* (TF), respectively. The sensitivity factor reflects the fraction of a certain region that contains ecosystems that are sensitive to the chemical to be assessed, while the threshold factor reflects the fraction of the total region where the background concentration of this chemical exceeds the toxicological no-effect level. The user of the GLOBOX-model can choose whether actual or potential impacts, or both, should be calculated by the model. 'Actual' impacts are then expressed as the impact scores for the sensitive, above-threshold part of the region. A problem that remains, however, is the limited availability of information, required for determining the sensitivity and threshold factors. This will be illustrated by an example below.

Around the year 2000, I was involved in a project, commissioned by the Netherlands Oil and Gas Exploration and Production Association (NOGEPA), that was meant to assess the emissions of oil platforms in the North Sea with LCA. To this end, we developed location-specific LCA characterisation factors for North Sea water, and alternative characterisation factors for ocean water (Wegener Sleeswijk *et al.*, 2003). The guidelines for the NOGEPA study included that we should assign zero values to the characterisation factors for metals in ocean water, since metal concentrations in oceans were well known to be far below environmental threshold values, and it was regarded useless to include non-existing impacts in the assessment of the companies' activities. What was asked, in fact, was to deviate from our convention to assess potential impacts, and to switch to the assessment of actual impacts. With the newly developed GLOBOX model, this seems to be easy: just set the threshold factor to a zero value for metal emissions to oceans. It should be kept in mind, however, that the user should be keen to make a consistent choice. If the choice is made to account for actual impacts, this should be

done for all chemicals involved in the study, and for all emission and distribution compartments in all regions. This requires a substantial amount of information, which is not always easily available. For LCA, this is much more complex than for risk assessment, because a product life cycle is not limited to one single process, neither to one single substance, to one single region or to one single year.

The GLOBOX model contains specific settings to account for actual environmental impacts in LCA toxicity characterisation. The development of a data set concerning regional environmental sensitivities and background concentrations would be a desirable next step.

7.3 LCA characterisation factors for metals

In the 1992 CML Guide on LCA (Heijungs *et al.*, 1992), characterisation factors for toxicity assessment were based on toxicity as such, combined with human intake for human toxicity assessment. In the years that followed, the possibilities of for the inclusion of multimedia transport and environmental degradation in LCA toxicity assessment were investigated (*cf.* Guinée & Heijungs, 1993). This had consequences for the ratios between characterisation factors, putting relatively more weight on persistent chemicals. In 1996, Guinée *et al.* published a study that included a list of LCA toxicity characterisation factors for about 100 chemicals, based on the Dutch RIVM multimedia fate model USES 1.0. On the basis of this same model, Huijbregts *et al.* (2000) developed a specific LCA multimedia fate model called USES-LCA. With this model, LCA toxicity characterisation factors for 181 substances were calculated, which were later on included in the LCA Handbook of CML (Guinée *et al.*, 2002). Although the USES 1.0 model has originally been designed for organic chemicals, characterisation factors for metals were calculated with the same USES and USES-LCA models, respectively, by the way of specific parameter settings. As a result of the non-degradability and very low rates of disappearance of metals from the environmental system, the calculations showed a strong accumulation of metals, especially in the seawater compartments, which acted as a sink. This resulted in extremely high characterisation factors for metals, and as a consequence, in environmental profiles which in most LCA cases studies were fully dominated by metal emissions.

These results drew the attention of specialists on the environmental chemistry of metals, who remarked that metal-specific environmental chemistry was not included in the model. At an expert meeting with experts from the fields of both LCA and environmental metal chemistry in Apeldoorn (The Netherlands) in May 2004, it was agreed upon that characterisation factors for metals had to be adapted, as was briefly laid down in the so-called 'Apeldoorn Declaration' (Aboussouan *et al.*, 2004). In the mean time, a workshop had been organised in Montréal (Canada) where experts from both fields could exchange ideas. It is for this workshop that

chapter 4 of this thesis has been prepared, as a proposal for the improvement of LCA characterisation factors for the ecotoxicity assessment of metals in seawater.

It was clear from the international discussion that the process of a better, science based inclusion of metals in LCIA would be a laborious and time-consuming task. The goal of the study in chapter 4 was to start this process by making a simple, transparent method with which the most urgent problems would be solved. Two metal-related issues were addressed:

- metal speciation in freshwater and seawater;
- vertical transport of metals to deeper water layers in seawater.

Free metal ion fractions of different metals in water were collected from literature and introduced in a preliminary version of the GLOBOX model. For metals, emitted in inorganic forms, bio-availability was assumed to be limited to these free ion fractions, according to the so-called *free ion activity model* (FIAM) (Morel, 1983). Regarding the vertical transport of metals, the seawater compartment was split into two parts: an upper mixed layer of 100 m and a lower layer, which was considered not to be part of the environmental system. As a consequence, the sedimentation process resulted in a reasonably fast elimination of metals from the environmental system. These two relatively simple modifications brought about that the values of metal characterisation factors dropped dramatically, thus no longer unduly causing metals to dominate environmental profiles because of their environmental persistence.

The original calculations reported in chapter 4 were executed with a preliminary version of the GLOBOX model. The fully differentiated version discussed in chapter 5 contains the same adaptations: temperature-dependent speciation, the inclusion of activity coefficients and the distinction between an upper mixed layer and a lower layer in the seawater compartment.

Further improvement of metal characterisation factors could be achieved by accounting for metal-specific environmental processes in other environmental compartments too. Especially for soil compartments, it will be necessary to adapt model parameters to local circumstances. How this should be fitted into a global LCA model would be a subject for further study. Within the water compartments, modelling could be improved by replacement of the FIAM by the biotic ligand model (BLM) (Di Toro, 2001), which accounts for bioavailability in a more refined way. An advanced model that has been specifically designed to account for the behaviour of metals in both aquatic and terrestrial environments is the Canadian TRANSPEC model (Bhavsar *et al.*, 2004 and 2008). In a project, commissioned by the International Council on Mining & Metals (ICMM) and executed by a number of collaborating institutes in The Netherlands and Canada, these issues are now being elaborated in the context of LCA (*cf.* Gandhi *et al.*, 2008).

7.4 Regional differentiation

Characterisation factors connected to substances released to the environment aim to reflect the relative environmental harmfulness of emissions of these substances in comparison to each other, assuming they are released in equal amounts. Environmental characteristics of the specific regions where the emissions take place are generally not accounted for in LCA characterisation factors. It has been demonstrated in this thesis that for the test substance nitrobenzene, when region dependent characteristics are included, the toxicity characterisation factor may show differences of up to 5 orders of magnitude for different countries. These findings are in good accordance with those of other authors, who report differences of up to 1 or 2 orders of magnitude for differentiation at the level of continents (Huijbregts *et al.*, 2003b, for fate and exposure factors; Rochat *et al.*, 2006, for intake fractions), and differences of up to around 3, 5 or 8 orders of magnitude for differentiation at the level of smaller regions within continents (MacLeod *et al.*, 2004; Pennington *et al.* 2005; Humbert *et al.*, 2009; all for intake fractions). Manneh *et al.* (2009) report a noticeable increase of differences with increasing level of spatial model resolution, which may explain why the differences, found at the level of small regions, are considerably higher than the differences, found at the level of continents.

Leaving out spatial information from LCA characterisation may cause the introduction of major inaccuracies in environmental profiles of product alternatives – and thus in the environmental ranking of product alternatives – since the emissions of the product life cycles to be compared may take place at different locations, not only for the different product alternatives, but also for the different processes within one product life cycle. Product life cycles usually consist of hundreds of processes, that may together span the world with respect to their regional origin. The lack of spatial differentiation may well reverse the outcome of an LCA study with respect to the question which product alternative should be preferred.

Spatial differentiation of toxicity assessment refers to three different issues: environmental fate, human intake, and effect. The environmental fate of chemicals depends on a large number of parameters: geographic parameters (for example, the relative surface area covered by water), geophysical parameters (*e.g.,* the organic carbon content of soils), climatological parameters (*e.g.,* temperature, rain rate and wind speed) and flow parameters (*e.g.,* river in- and outflows). Human intake depends on population density, food and drinking water consumption patterns, drinking water origin, drinking water purification characteristics, and body weight. Ecotoxicological effects depend on regional species sensitivities, and both human and ecotoxicological effects depend on background concentrations in relation to toxicological threshold values. The degree to which a certain environmental parameter influences LCA outcomes will be chemical-dependent; for instance, there will be a relatively high influence of drinking water related parameters on LCA on

characterisation factors of chemicals that tend to reside in the water compartments.

In order to test whether spatial differentiation matters or not, three conditions should be met:

1. All possibly influential parameters should be spatially differentiated with respect to environmental fate, human intake and human and ecotoxicological effects.
2. Fate, exposure and effect models should contain equations that reflect all possibly relevant spatial dependencies.
3. The environmental system tested should be sufficiently broad and varied to represent the given spatial variation of model parameters.

The GLOBOX model for LCA toxicity assessment has been designed according to these conditions. A central characteristic of the model is the combination of its large number of spatially differentiated parameters with a global range and a relatively high level of spatial differentiation. The exemplifying calculations with nitrobenzene as a test chemical show that many of the spatially differentiated parameters do matter, and that the influence of spatial differentiation is comparable to the influence of chemical characteristics.

A question that can be posed is whether the traditional, non-differentiated toxicity-related characterisation factors make any sense at all if spatial differentiation is indeed so important. I think the answer is that they do make sense if they are handled with care, that is: if they are used only for the assessment of product life cycles of which the main processes are located in the area to which the model parameters apply, *e.g.,* to Western Europe for the USES-LCA factors (Van Zelm *et al*, 2009) or to North America for the CALtox factors (McKone *et al.*, 2001). This implies that for other countries, the use of spatial differentiation *or* the replacement of region-dependent parameters by the applicable values is necessary for acquiring reliable characterisation factors.

It should be kept in mind that a regionalised model can address both actual impacts and potential impacts, as discussed above. Potential impacts can thus be calculated either with a non-regionalised model or with a regionalised model. A regionalised model is expected to improve the results for potential impacts, without pretending to make them more actual. For actual impacts, a regionalised model is indispensible.

I believe that the development models like USEtox (Rosenbaum *et al.*, 2008), resulting from the harmonisation of a number of existing models, are very useful for the enhancement of model reliability with respect to their selective power considering model structure, equations and parameter construction. A parsimonious approach, defended by the designers of the USEtox model, can be useful in this

context. In my view, however, parsimony should be considered as an intermediate stage, that serves the goal of disposing the model of bugs and flaws. After this, the improved model should be tooled up again with spatial characteristics and other information that makes the model reflect reality as well as possible. The ideal model should reflect the world as it is, in its full complexity. The challenge for the modeller is to develop sufficient knowledge on relevant details to allow for building a construction that produces perhaps unpredictable but yet plausible results in relevant model situations.

7.5 Normalisation

The primary goal of LCA normalisation is to translate the abstract impact scores of the environmental profile into more imaginable indicators, that relate each score to a concrete reference situation. A typical choice for the reference is the collection of global emissions of potentially harmful substances in a certain year. A normalised environmental profile gives an image of the relative contributions of a functional unit of product to the different environmental problems as they are occurring.

The goal of the normalisation study described in chapter 6 of this thesis was to update the formerly published normalisation factors for the reference year 1995 (Huijbregts *et al.*, 2003a), to the new reference year 2000. Normalisation factors were determined at two levels: the EU (plus Switzerland, Norway and Iceland) in the year 2000, and the world. This has been done for all impact categories which had been elaborated in the LCIA ReCiPe-project (Goedkoop *et al.*, 2009) by that time, in the context or which this normalisation study was performed.

It should be noted that the ReCiPe-characterisation factors for the toxicity-related impact categories differ from the characterisation factors that could be derived with the GLOBOX-model. Differences largely apply to environmental modelling parameters – especially in relation to spatial differentiation – and to the way in which the characterisation factors for metals have been derived. This last issue also influences the relative weight of the normalisation factors for metals.

When I started collecting data on emissions in this new reference year, I encountered the discrepancy between production data and emission data for CFCs, caused by the recent ban on CFC. On the one hand there were the low emission rates, related to the adaption of production processes to the new legislation, and on the other hand the still high rates of actually emitted CFCs, due to the use and waste treatment of formerly produced refrigerators, air conditioners, etc. The question was whether for ozone depletion, the actual CFC emissions in the year 2000 would form the right reference if the goal of normalisation were to assess the contribution of the environmental impacts of a certain product to the impacts of all products together. CFC emissions in the year 2000 formed an inheritance of past economic systems, rather than resulting from the life cycles of products pro-

duced in the chosen reference year. Inversely, it was the question whether it was correct that new chemicals, produced in the year 2000 but released in later years, would not be taken into account for products produced in the year 2000 by traditional normalisation factors. In the traditional approach, normalisation scores (or 'contributions') of single products could in theory even become more than 100% of the total. Thus, the question presented itself whether LCA normalisation should take environmental interventions *taking place* in the reference year, or, alternatively, the interventions *caused by economic activities* in this same year, should be quantified as reference interventions. Personally, I felt we had not been right by choosing the first of these two options in our former normalisation activities. If we wanted to use normalisation as a reference for assessing the relative contribution of the environmental impacts of a product to 'total' environmental impacts, this 'total' should reflect the impacts of all product life cycles together, in their proper magnitudes. And this corresponds in my view to the second option: that of the interventions, caused by economic activities in the reference year. This is what we aimed at in our updated normalisation study.

Different sources of uncertainty and bias were identified in our normalisation study. Data gaps in LCA inventory were identified as the most important cause of both uncertainty and bias. If normalisation factors are uncertain, this enhances the risk of biased results in case studies (Heijungs *et al.*, 2007). With respect to missing emission data, the foundation of an independent organisation that aims to register emissions in a systematic way, as suggested by Scheringer (2007), could offer the possibility to make a step forward in this respect.

Another type of bias can be caused if a discrepancy exists between the reference region for normalisation (used for the construction of a normalisation factor) and the 'ecological footprint' (Wackernagel & Rees, 1996) of this same region (in which product life cycles may extend). 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 fact, a footprint approach seems to offer the most complete figure. Most normalisation studies – including the present study – use geographic boundaries as system boundaries, thus being prone to this type of bias. A possible future implementation of the 'ecological footprint' principle could be reached by the use of economic input-output models (*e.g.*, Hubacek, & Giljum, 2003; Suh, 2004) to estimate the emissions due to a region's consumption.

7.6 Main achievements and conclusions

This thesis reports an attempt to improve the reliability of LCA toxicity assessment and normalisation. In the introduction, five goals were presented:

1. Contributing to an optimal reliability of LCA toxicity assessment by creating a flexible, reasonably detailed system for spatial differentiation of LCA toxicity assessment on a global scale.
2. Enhancing the accuracy of LCA modelling with respect to the behaviour of metals in the environment.
3. The introduction of a method for the assessment of contributions of the product life cycle to toxic risks or *actual impacts*, along with the conventional assessment of *potential impacts*.
4. Analysing the influence of spatial differentiation on LCA characterisation factors for human toxicity and ecotoxity by calculations on a test substance.
5. Creating an updated, global LCA normalisation sytem.

The five goals have been achieved as follows:

With respect to toxicity assessment, the software tool GLOBOX has been developed, in which three concepts have been implemented:

- spatial differentiation on a global scale (1);
- a specific approach for the calculation of the fate of metals in the oceans (2);
- a distinction between potential and actual impacts (3).

The GLOBOX model has been tested by calculations on the test substance nitrobenzene, with an analysis of the relevance of spatial differentiation (4).

Normalisation has been updated at two scales: the Western European scale and the global scale (5).

From the calculations with the test substance nitrobenzene, it has been shown that spatial differentiation can indeed make a large difference, and that lack of spatial differentiation may lead to rank reversal of chemicals with respect to the harmfulness of their emissions.

Metal characterisation factors have been shown to fall dramatically if speciation is accounted for, and if only the upper mixed layer of the seas is regarded as part of environmental system.

Actual impacts have not yet been calculated, but after a discussion of many years about whether this is possible in LCA at all, a first step has now been set by rendering the assessment of actual impacts possible, in the GLOBOX model, both conceptually and implemented in the model software.

Normalisation has been founded on a new principle: the idea that normalisation factors should be based on the production processes in the given year, rather than on the actual emissions taking place in that year, in order to remain in line with the characterisation of a product, which is also about the processes for which this product (by itself being produced in a certain year) is responsible. All products together, produced in a certain year, are thus forming the basis for the normalisation factors.

Apart from the practical achievements, this thesis can also be viewed from a more abstract perspective. It can be considered as a plea for not avoiding complexity in environmental science. Quoting Einstein: *Everything should be made as simple as possible, but not simpler*. Spatial differentiation causes multimedia models to become more complex, and thus less easy to understand, but this enables them to much better reflect reality and lead to relevant results.

In my view, three levels can be distinguished in the relation between environmental science and policy. The highest level is the level of implementation of environmental measures, such as an emission tax or the banning of chemicals. At this level, a balance should be found between scientific environmental information, feasibility in terms of funding and time, and social and economic interests. This is primarily a political issue in which scientific knowledge only plays a role as source of essential information.

One level below is the level of environmental tools like LCA and HERA. At this level, a scientific foundation is an absolute requirement. In the mean time, it is important that the instrument remains manageable for the user. Tools that can be handled by experts only, that require input that is not easily available, or that are very time-consuming in use, will not become popular. Again, a balance will have to be found, this time between scientific accuracy on the one hand and practicability on the other. Since practicability is more easily recognizable than scientific accuracy, safeguarding the latter is a special point of attention. Attractively designed tools of which the results are not sufficiently detailed or even scientifically incorrect are of little use and may even be harmful. Smart software may help to include the necessary scientific details in environmental tools while preserving user friendliness, practicability and elegance.

The basic level on which both environmental measures and environmental tools rest is the level of scientific modelling. At this level, scientific accuracy is the main purpose. Scientific models should reflect reality as well as possible. On this basis, choices can be made about the details that can be omitted from practical tools if desirable – without giving in too much on scientific quality – and about details that are indispensible and should be included anyhow. It is at this level that the GLOBOX model has been designed, built on the conviction that environmental practice should always be based on sound scientific grounds.

One final remark. Despite the importance of the environmental improvement of products, it is my personal belief that the most urgent step forward towards a sustainable economy is in a decrease of industrial production. It may be questioned what the eventual goal of production should be, and how this relates to the fulfilling of human needs, both in a material and in a financial-economic sense. In environmental discussions, the emphasis is often placed on a decrease in consumption, but if consumption levels are largely driven by production and marketing, these should also be the subject of environmental management.

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