Human and ecological life cycle tools for the integrated assessment of systems (HELIAS)

Guinée, J.B.; Heijungs, R.; Kleijn, R.; Voet, E. van der; Koning, A. de; Oers, L. van; ... ;
Wegener Sleeswijk, A.

Citation

Guinée, J. B., Heijungs, R., Kleijn, R., Voet, E. van der, Koning, A. de, Oers, L. van, ... Wegener Sleeswijk, A. (2006). Human and ecological life cycle tools for the integrated assessment of systems (HELIAS). *Int J Lca*, *11*(Special Issue 1), 19-28. Retrieved from https://hdl.handle.net/1887/11435

Special Issue Honouring Helias A. Udo de Haes: LCA and Other AssessmentTools

Human and Ecological Life Cycle Tools for the Integrated Assessment of Systems (HELIAS)

Jeroen B. Guinée*, Reinout Heijungs, René Kleijn, Ester van der Voet, Arjan de Koning, Lauran van Oers, Ayman Elshkaki, Ruben Huele, Gjalt Huppes, Sangwon Suh and Anneke Wegener Sleeswijk

Institute of Environmental Sciences (CML), Leiden University, P.O. Box 9518, 2300 RA Leiden, The Netherlands

* **Corresponding author** (Guinee@cml.leidenuniv.nl)

DOI: http://dx.doi.org/10.1065/lca2006.04.008

Abstract

Goal, Scope and Background. CML has contributed to the development of life cycle decision support tools, particularly Substance / Material Flow Analysis (SFA respectively MFA) and Life Cycle Assessment (LCA). Ever since these tools emerged there have been discussions on how these tools relate to each other, and how they relate to more traditional tools. Remarkably little, however, has been published on these relationships from an empirical side: which combinations of tools have actually been used, and what is the added value of combining tools in practical case studies. In this paper, we report on CML's experience in this field by presenting a number of case studies with their related research questions, for which different tools were deployed.

Methods. Three case studies are discussed: 1) Waste water treatment: various options for waste water treatment have been assessed on their eco-efficiency, using SFA to comment on the influence of these options on the flows of certain substances in the water system of a geographical area and a combination of LCA and life cycle costing (LCC) to assess the life-cycle impacts and costs of these options; 2) Prioritization of environmental policy measures: A methodology has been developed to prioritize environmental policy measures and investments within companies based on both the environmental impacts and the costs of these measures; and 3) Environmental weighting of materials: to add an environmental dimension to standard MFA accounts, materials were weighted with cradle-to-grave impact factors based on LCA data and impact assessment factors.

Results and Discussion. For each of these cases, the research questions at stake, the tools applied, the results and the added value, limitations and problems of combining the tools are reported.

Conclusions and Perspective. Based on these experiences, it is concluded that using several tools to address a complicated problem is not only a theoretical proposal, but also something that has been applied successfully in a variety of practical situations. Furthermore, using several tools in combination does not necessarily lead to an increased information supply to decision-makers. Instead, it may contribute to the comprehensibility and ease of interpretation of the information that would have been provided by using a single tool. Finally, it is concluded that there is not one generally valid protocol for which tools to use for which question. The essential idea of using a combination of tools is exactly the fact that research questions are not simple by nature and cannot be generalized into protocols.

Keywords: Integrated assessment of systems; life cycle costing (LCC); life cycle tools, human and ecological; material flow analysis (MFA)

Introduction

CML has contributed to the development of life cycle decision support tools, particularly Material Flow Analysis (MFA) and Life Cycle Assessment (LCA). Amongst others, this is reflected by a number of PhD-theses in this area that have been delivered under the inspiring leaderschip of Helias A. Udo de Haes (Huppes 1993, Guinée 1995, van der Voet 1996, Heijungs 1997, Suh 2004) or that are in preparation (Kleijn, Wegener Sleeswijk, Elshkaki). These life cycle tools take a central position in the rapidly emerging research field of 'Industrial Ecology', as well as in its applications in concrete case studies.

Ever since these tools emerged there have been discussions on how these tools relate to each other, how they relate to more traditional tools such as Environmental Risk Assessment (ERA), Environmental Impact Assessment (EIA), etc. and how these tools could supplement each other in specific cases.

The longest discussion is on the relationship between LCA and ERA, and goes back to the Leuven workshop (De Smet, 1990). The Leuven workshop concluded LCA is not a risk assessment as "[...] an environmental risk assessment can only be carried out with site specific data on dose vs. response rather than typical LCA data which represent energy requirements and releases spread over time and geography." These conclusions have been followed-up by several similar pleas (e.g. Saouter & Feijtel 2000, Wegener Sleeswijk 2001, Olsen et al. 2001, Wegener Sleeswijk et al. 2003), although there have also been pleas for far-reaching integration of LCA and ERA(e.g. Assies 1998, Cowell et al. 2002).

The discussion on the relationship between these tools, however, is much broader than merely LCA vis-à-vis RA. Initial surveys by Beck & Bosshart (1995) and subsequent authors distinguish a dozen or more tools and concepts for environmental decision-support. A panoply of such tools may create a confusing dilemma to a decision-maker: when to use which tool? During the nineties, an EU concerted action was specifically devoted to this topic: CHAINET (Wrisberg 2000, Wrisberg et al. 2002). Two important conclusions were that 1) although some tools are very similar, there is no 'super tool' that embraces the other tools, and that 2) some questions could best be approached by the simultaneous or consecutive application of two or more tools.

Udo de Haes et al. (2004) also acknowledge that LCA cannot address all types of questions that decision-makers face. Starting from this LCA-angle, they summarize the various possibilities of combining tools as follows:

1. extension of LCA – one consistent model;
2. use of a toolbox – separate models used in combination; and
3. hybrid analysis – combination of models with data flows between them.

Here, extension can be regarded as bringing in scientific depth: better accounting for spatial detail, incorporation of more sophisticated models, etc. The use of a toolbox refers to the simultaneous or consecutive application of different tools for the same (or a similar) question. Finally, hybrid analysis takes an intermediate position: it employs aspects from different tools in a not-too-loose but also certainly not-too-integrated way. In this paper, these three possibilities of combining LCA with other tools will not be limited to LCA, but will also include MFA with other tools, ERA with other tools, etc.

So far, the theoretical aspect involving the domain of tools and the possibilities to combine tools. Even though a few explicit studies combining tools are available (e.g. combining LCA and MFA for the case of chlorine by Tukker et al. 1998, combining SFA and ERA for the case of heavy metals (Van der Voet et al. 2000), remarkably little has been published on these issues from an empirical side: which combinations of tools have actually been used, and what is the added value of combining tools in practical case studies (cf. Sonneman et al. 2001). In this paper, we report on CML's experience in this field by presenting a number of case studies with their related research questions, for which different tools were deployed. In each of these case studies at least one life cycle tool was applied beside at least one other tool.

1 Cases

This section presents the following three example cases of combining different tools:

- Waste water treatment: various options for waste water treatment have been assessed on their eco-efficiency, using SFA to comment on the influence of these options on the flows of certain substances in the water system of a geographical area and a combination of LCA and life cycle costing (LCC) to assess the life-cycle impacts and costs of these options.
- Prioritization of environmental policy measures: A methodology has been developed to prioritize environmental policy measures and investments within companies based on both the environmental impacts and the costs of these measures.
- Environmental weighting of materials: to add an environmental dimension to standard MFA accounts the materials were weighted with cradle-to-grave impact factors based on LCA data and impact assessment.

For each of these cases, the research questions at stake, the tools applied, the results and the added value, limitations and problems of combining the tools will be reported. Based on the experiences from these three case studies, an attempt is made to derive general guidelines on the relation between tools and questions.

1.1 Wastewater treatment

Research questions. Within a EU FP5 project, various options for wastewater treatment have been assessed on their eco-efficiency, using a combination of SFA, LCA and LCC. SFA was used to assess the influence of wastewater treatment options on the flows of certain substances in the water system around WWTP installations. A combination of LCA and LCC has been applied to assess the life-cycle impacts and costs of these options. The project, known as P-THREE (www.pthree.de), studies Persistent Polar Pollutants (PPP). This group of substances contains many 'new' substances increasingly used in medicine or as a chemical with specific properties in industry. The problem arises mainly because of their polarity: ordinary wastewater treatment is not able to retain these substances in the sludge. This means that they are emitted to the surface water. Especially in areas with a water cycle rather than a water throughput, environmental concentrations might build up. Although little is known yet regarding their harmful potencies, rising concentrations of man-made chemicals is a concern for the future.

In the P-THREE project, various options for a better removal of these substances are investigated. One of them uses a Membrane Bioreactor (MBR) instead of the conventional Activated Sludge Process (ASP). One of the main technical differences between MBR and ASP is the way activated sludge is separated from the cleaned wastewater. In ASP, this is accomplished in large settling tanks. In MBR, separation is accomplished in membrane units. In a MBR, the sludge retention time can be manipulated independently from the hydraulic retention time. An MBR plant may, because of the longer retention time of the sludge, perform better in the removal of PPP. Preliminary results of the P-THREE program indeed point in that direction. However, there is more at stake and a broader system analysis is performed to cover life cycle aspects of these options.

Tools applied. A substance flow analysis (SFA) is performed to put the Wastewater Treatment Plant (WWTP) within the total life cycle of the substance. The implicit assumption is that the substance enters the environment only through the WWTP. This may not be the case. Some substances are emitted directly to the environment, for example pesticides, or methyl tertiary-butyl ether (MTBE) as an addition to gasoline. The WWTP hardly plays a role in their life cycle, therefore a better treatment of wastewater hardly influences environmental flows. This is different per substance. Therefore, a number of PPP of a different nature is selected to analyze with SFA, as shown in **Table 1**. Two geographical areas are examined: Berlin and Barcelona. In Berlin, the water flow is to a large extent a cycle: surface water is taken for drinking and industrial water, which ultimately ends up in the surface water again via the WWTPs. Water intake actually takes place downstream from the emission of the WWTP effluent. In Barcelona, there is hardly a cycle, but there are large fluctuations in the yearly water flow, and indeed between the summer and winter season. In dry periods, there is very little water to dilute the emissions from the WWTPs.

Table 1: Sources, influence of WWTP and solutions for different P3 substances

	Main route	Influence of wastewater treatment	Solution
EDTA	Mixed	+	Reduce use
Diclofenac	Households	++	WWTP eff. Up
MTBE	Cars / atmosphere	-	Substitute
LAS / SPC	Industry	++	WWTP eff. Up

An LCA according to the Handbook guidelines of Guinée et al. (2002) is performed to obtain an insight concerning the side effects of the various wastewater treatment alternatives. MBR may do a better job in removing PPP, but might turn out to perform worse in other areas. To assess this, a cradle-to-grave analysis is performed of an existing ASP plant (Ruhleben in Berlin, including a 15 kilometer pipeline to discharge the effluent beyond a recreational area) and of a hypothetical alternative for Ruhleben, based on MBR treatment (no pipeline needed). Thus, the total impacts should become clear and can be compared. Added to the LCA, an LCC is carried out to get insight into the costs of these alternatives.

Finally, measures of eco-efficiency will be extracted from the SFA and LCA/LCC results. The combination of these tools cover a wide area of aspects. **Fig. 1** shows this. An even broader picture might have arisen when the SFA would have been linked to a risk analysis. Although the SFA results have been translated into concentrations in the surface water, a risk analysis is no option since no environmental standards have yet been made concerning the PPPs, as mentioned in Table 1. Carrying out the risk analysis ourselves, which includes looking up reliable effect data, was beyond the scope of this study.

Some preliminary results are presented below.

Results. For the substance flow analysis, the starting point was the analysis of water flow. An SFA of water was drafted for the Berlin and Barcelona areas. The next step then is to make the SFA of the different substances. In combination with the water flow, this could be translated into concentrations in the different water flows. An example is shown below for the flows of diclofenac in the Berlin area. Diclofenac is a medicine increasingly used, for the treatment of arthritis, etc. Its pathways is therefore through the WWTP, and an improved treatment of wastewater may be expected to have good results. **Fig. 2** shows the concentrations of diclofenac as a function of the wastewater treatment efficiency.

Present removal in an ASP is measured in the order of 5–10 percent. MBR treatment increases the measured removal percentage considerably, to roughly 50 percent. For substances like diclofenac, therefore, MBR helps improving environmental quality. For other substances, this may be different (see Table 1).

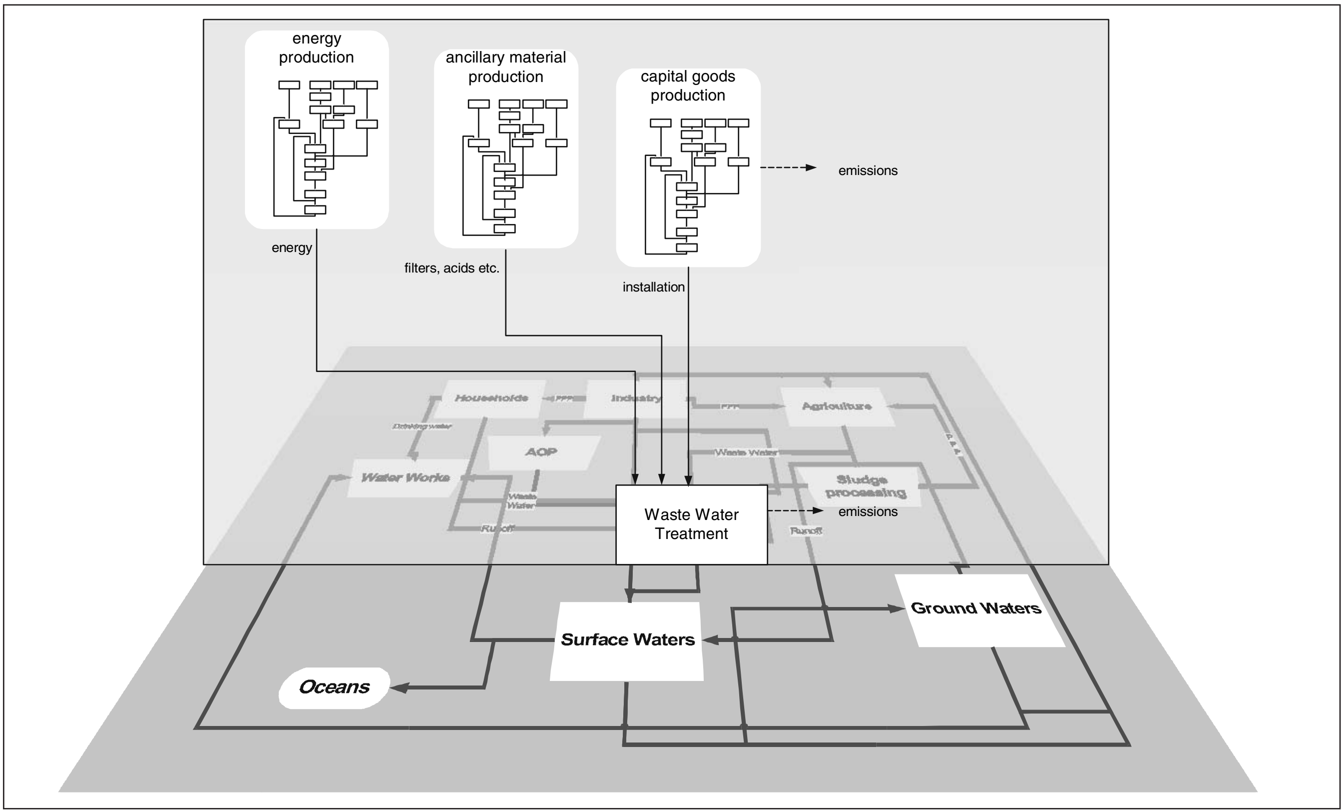

Fig. 1: Life cycle based assessment of P-three removal in WWTP from two perspectives: SFA and LCA

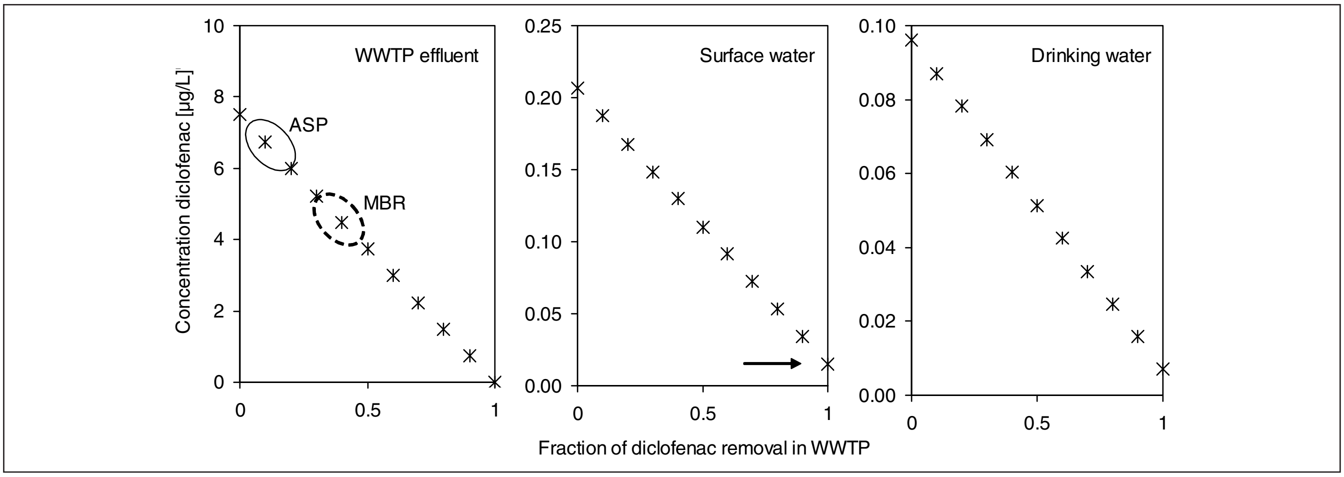

Fig. 2: Concentrations of diclofenac in the Berlin area as a function of the fraction of diclofenac removed in the WWTP. The average removal efficiency of ASP is indicated by the oval with a solid line. The expected removal efficiency of MBR is indicated by the oval with a broken line. The arrow indicates that some diclofenac remains in the surface (and drinking) water, even when all diclofenac is removed in the WWTP, since not all sewage is treated in WWTPs

The actual concentrations and flows calculated using SFA might be used in as input for a risk assessment, although this line of research has not been carried out. The LCA and LCC results are summarized in **Fig. 3**

The MBR does a slightly better job in removing N, P and micronutrients from wastewater. Still, the MBR scores worse in the freshwater aquatic toxicity impact category. In the whole cradle-to-grave analysis, more micro-pollutants (mainly metals) are emitted from the energy production sector due to the higher energy use of the MBR offsetting the lowered impact on freshwater toxicity by the MBR plant itself. In the category involving eutrophication, MBR scores better than the ASP, even when the higher energy demand by MBR is taken into account from a cradle-to-grave perspective.

In most other respects, MBR scores worse than the traditional ASP. This is due to the higher energy use of the MBR, especially the electricity needed for the coarse bubble aeration (anti-fouling measure for the membranes). A further interesting insight provided by the LCA is that N_2O emissions from the biological stage (both in ASP and MBR) are significantly contributing to the global warming impact category.

Knowing that electricity use is such an important driver in the environmental assessment of the MBR, which level of

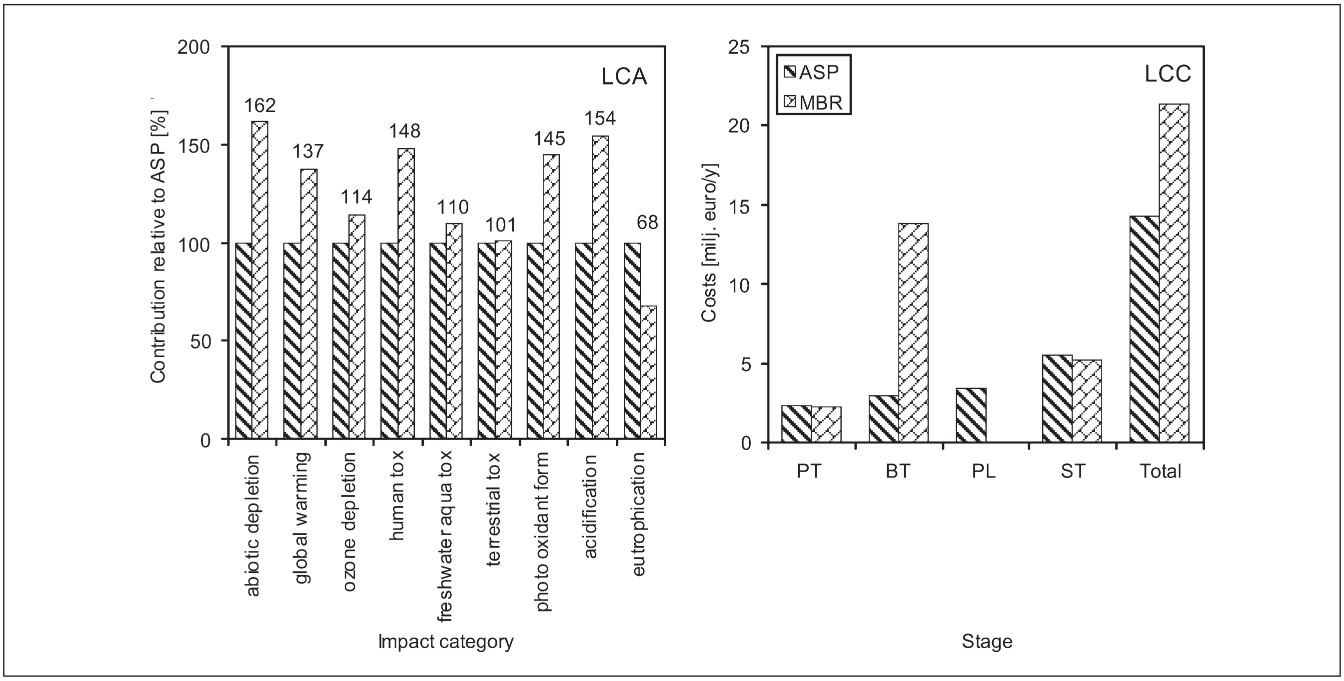

Fig. 3: Comparison of cradle-to-grave impacts of ASP and MBR (left) and comparison of the life cycle cost of ASP and MBR (right). The cost of running the MBR and ASP has been divided into four stages: PT = primary treatment, BT = biological treatment, ST = sludge treatment, PL = pipeline. Notice that the MBR alternative does not use a long pipeline for the discharge of effluent

energy use is allowed for the MBR, and still has a better environmental performance than the ASP, was examined. This energy level was found to be about 5–6% higher than the ASP. The first MBR installations required about twice the amount of energy of ASP installations having the same treatment capacity. However, MBR is new technology and evolving fast. It is expected that the energy requirements of MBR installations will drop considerably in the near future to a level of about 30% higher than ASP.

The LCC indicates that the total cost of the MBR installation are higher. These higher costs are the result of the high cost of the biological treatment step. Both capital cost and operational cost (especially energy cost) are higher for the biological treatment. Again, this is not a definitive conclusion about MBR. Besides the expected further improvement in energy efficiency, the lower cost of membranes and increased lifetime of membranes will reduce the cost of MBR installations considerably in the near future.

Added value, limitations and problems of combining tools. In this case study, the tools are used separately and therefore no methodological difficulties arise. Nevertheless, the conclusions from applying these tools may point in different directions. For example, from the SFA for diclofenac, it appears that MBR is the WWT technology to be preferred. From the LCA, this is by no means clear; if anything, the results point in the opposite direction as that observed for the SFA. On the one hand, this reflects the added value of applying a combination of tools, but, on the other hand, this also provides a problem in the interpretation of the results. Which option is to be preferred does not depend on the P3 removal only, and even the life-cycle impacts are not the only basis for the decision-making. The local situation can be very important as well. For example, the WWTP may be located in a sensitive or protected area, where meeting surface water standards is imperative even at the cost of energy related problems elsewhere.

Moreover, the much smaller area occupied by an MBR is also a very favorable aspect, especially relevant in densely populated areas. Therefore, even the combination of LCA, LCC and SFA cannot provide the complete answer.

1.2 Prioritization of environmental policy measures

Research questions. The local authorities in the Netherlands province of Zeeland asked the Institute of Environmental Sciences (CML), Leiden University and the Fuels and Raw Materials Bureau (B&G) to develop a methodology to prioritize environmental policy measures and investments within companies taking into account both its environmental impacts and its costs. From a series of interviews with decision-makers in local governmental organizations and local industries, it became clear that almost none of the decision-makers were experienced in using methodologies to prioritize environmental policy measures taking into account emissions in the life-cycle of the products and services offered by the companies. Emission reduction measures at the industrial plants, which could be implemented at reasonable costs, had already been taken. A further reduction of emissions from companies would mean that the costs per unit of environmental gain would strongly increase. Therefore, a well-balanced assessment of costs and environmental gain, which can possibly take into account emissions in the life cycle of products, had become a necessity to prioritize environmental investments and policy measures.

Tools applied. The basis of the methodology was formed by existing LCA and cost analysis methods. A set of guidelines for practitioners in industry and local governments in the Dutch context was developed. This form was chosen because local industries and policy makers in the province of Zeeland were the target group for the method. The guidelines were split up in guidelines for the cost part and for the environmental analysis part. For costs, one specific cost accounting method was chosen which was developed by the Netherlands Ministry of Housing, Spatial Planning and the Environment (VROM 1994, VROM 1998). In the environmental analysis, a simple stepwise working plan was presented to the users in which six subsequent steps were distinguished:

1. Definition of the scope of the study
2. Distinguishing location specific and generic environmental impacts
3. Choice for a specific method or tool
4. Data collection
5. Translation of the data into a list of environmental interventions
6. Calculation of the environmental impacts

In the first step, it is decided what the system boundaries are which part of the life cycle or production and consumption chain will be part of the quantified analyses. It is also decided which environmental aspects will be taken into account. The second step has been made explicit because only generic impacts are considered in most tools. If location specific impacts are to be considered, additional tools should be used. In the third step, a choice for a specific method is made from a short list of seven methods that are considered the most promising operational methods. In the fourth step, data is collected for the different policy measures and investments. Depending on the specific method that has been chosen in step 3, these data include the material inflows and outflows of the processes that are involved, including the emissions and waste flows. In step five, the data collected in the previous step are used for the compilation of a complete list of environmental interventions connected to the different alternatives. In the sixth and final step, the environmental impacts of the interventions are calculated, resulting in a list of scores related to the aspects that are chosen in step 1. These environmental scores are then combined with the costs in order to determine the cost effectiveness of each alternative.

Results. The main results of the project are a brochure in which the Guidelines are presented as guidelines for practitioners (Provincie Zeeland 2001), and a report which can be used as a more detailed reference (De Koning et al. 2001). The method was tested on six cases, three at a phosphorous production plant and three at a starch production and processing plant.

At the phosphorous production plant, the following three cases have been used:

- reduction of phosphine emissions during the production of phosphorous with the aid of inert gas and a sparger;
- changing from natural gas to phosphorous oven gas for calcination in the phosphorous plant;
- treatment of the of gypsum waste in order to make it useful as a building material.

At the starch producer and processor, the following three cases have been used

- choosing between two possible environmental measures: install a deNOx installation at a combined heat power installation or the reduction of sulphate emissions with a sulferox installation;
- reduction of odor problems in the surroundings of the waste water treatment plant by solving H_2S by leading it through a water basin;
- using corn instead of wheat as a raw material for the starch production.

In **Fig. 4**, the results are given for the case of phosphine emission reduction. It shows that, although the effect of the measures to reduce the phosphine emissions have a clear positive effect on the local emissions, this effect is largely compensated for by the impacts of processes further in the chain.

The main conclusions from the cases are that the methodology can provide insights in possible environmental improvements of different measures. However, it is also clear that the method is not useful for measures with a large impact on the production process, such as changing to a completely new production process. Furthermore, it was found that there are no procedures that can be used to determine the system boundaries (e.g. which emissions are taken into account and where is the boundary between environment and economy), while this choice can influence the results of a study to a large extent. The cases made it also clear that different methodologies give different results. In order to understand and interpret these differences, it is important that the methodologies become more transparent, especially if they include some sort of weighting between environmental problems. Next to that, it is difficult to include local environmental problems in LCA-type methodologies and almost none of the methodologies is complete in the set of environmental problems included.

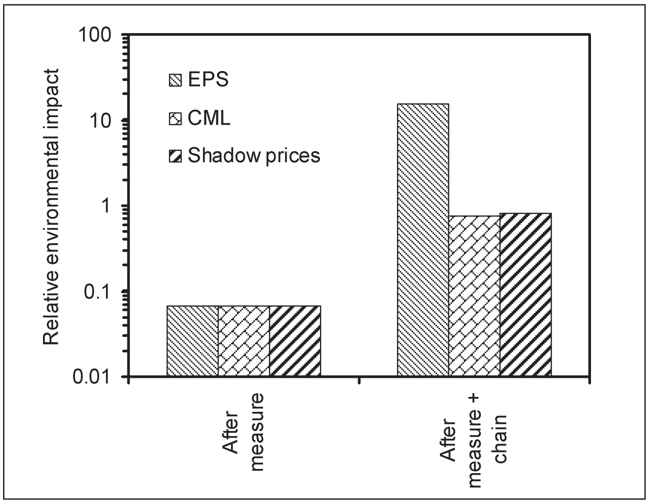

Fig. 4: The relative environmental impact of the emissions before and after the environmental measure. The first set of bars reflect the effect on the on-site emissions of the production plant, while in the second set of bars the impacts are shown when also the electricity production chain is included in the analysis. The data in this graph should be used for illustration purposes only. The relative environmental impact before the measure is set at 1. Note that a log scale is used on the y-axis

The method was tested in a session with around 50 participants from local governmental organizations and local industries. The results from this session were quite promising and many participants found quite some eye-openers in the cases that were presented to them. Especially the integrated framework, considering all environmental impacts and the life cycle perspective made it clear that there is often unexpected problem shifting. It was also clearly appreciated that the guidelines provide some help with the use of tools that are often regarded as complicated and very academic. However, there are also clearly some difficulties left to overcome if one wants to attain the achievement that these methodologies are used routinely. The methods are still seen as time consuming and too many black boxes with underlying assumptions that are not always explicit.

Added Value, limitations and problems of combining tools. The added value of using combinations of tools in this case is that local to global environmental impacts and costs can be taken into account together in the decision-making process. For example, the odor problems that are solved by leading the H_2S through the water basin solve the local odor problems. However, since it requires the use of extra electricity, there is a problem shift from local problems towards national and global impacts. By using both simple inventory data for the regional odor impacts together with LCA for the national global impacts to support the decision, both levels of impacts can be taken into account. By including the costs, this specific measure can be compared to other measures on the basis of environmental gain per •. Both decision-makers in industry and governmental agencies benefit here from using a combination of tools because it facilitates a more integral decision. One of the limitations of a combination of tools is that no combination of tools will ever cover all aspects that decision-makers are concerned with. Things like safety and social aspects are not included in the methodology that was developed. Furthermore, a combination doubling the number of tools used also means doubling the amount of work that is needed to get results and doubling complexity that has to be studied by the user and decided upon by the decision-maker.

1.3 Environmental weighting of materials

Research questions. A topic in which policy-makers are highly interested is decoupling of environmental pressures from economic activities. Decoupling can be defined as economic growth combined with less-than-proportional growth of environmental pressure, or even with a decline of environmental pressure. Within the framework of a study for the Resource Strategy in development at EU DG Environment, a study was carried out on how to measure decoupling and how to monitor progress on the decoupling road (Van der Voet et al. 2005). For monitoring, indicators or measurements are required that encompass the following characteristics:

- they should be applicable at the (supra)national level; they should indicate a total level of environmental impacts, related to the use of materials or resources; and
- they should enable creating time series in order to monitor progress.

Tools applied. In earlier studies (Moll et al. 2003), the Domestic Material Consumption (DMC) over the Gross Domestic Product GDP (DMC/•) has been put forward as such an indicator. The DMC is an indicator made up of MFA accounts as drafted according to the Eurostat methodology (Eurostat 2001). The DMC measures the material resources that are directly consumed within a national economy. The DMC has been put forward as an indirect indicator for environmental pressure. The reasoning behind this is that each kilogram of material entering an economy has to come out at some moment as waste or emissions in the end. While this is undoubtedly true, it is at the same time true that there are large differences in environmental impacts between different resources or materials. A kilogram of sand does not have an equal impact as a kilogram of copper, or meat, or coal. The potential environmental impacts of the different materials or resources should be considered as well as the weight or volume of their use. In the end, it is their environmental pressures and impacts that should be decoupled from economic growth, not their use per se. In this study, we developed an indicator combining information on material flows (with MFA) with information on environmental impacts (using LCA). We called this indicator EMC, Environmentally weighted Material Consumption.

The idea behind the EMC is quite simple (Van der Voet et al. 2004): *multiply the material flows with a factor representing their environmental impact*. Material flows are available through DMC and the accompanying MFA account. To specify the environmental impacts of a material, a Life Cycle Impact approach is taken. For every considered material, an estimate is made of its contribution to environmental problems throughout its life cycle. This includes not only the impacts related to the material itself, but also the impacts of auxiliary materials, energy used for its extraction and production, emissions of impurities and pollutants included in the material during use or waste treatment, etcetera. Energy use in the consumption phase is not allocated to the materials' chains. We consider this energy use - for example, gas in cars or electricity for computers - to be related to products rather than materials. It is difficult to allocate the use of energy to the individual materials a product is composed of, and quite often the energy use is hardly related to these materials. Energy use in the consumption phase, however, is not excluded from the EMC: it is included in the chains of fossil fuels, and any change due to shifts to less energy-intensive products will be visible in the EMC.

The established impacts in this way provide the total cradle-to-grave impact per kg of the material. This impact factor is then multiplied with the number of kilograms of this material being consumed to obtain an idea of the environmental impact of the consumption of the material. Summed over all materials, a picture emerges of the potential environmental impact of the material consumption of a national economy.

This simple idea, when put into practice, proves not to be that simple. In this case, the tools of MFA and LCA are really integrated, which has methodological complications and constraints:

- Double-counting: cradle-to-grave chains of materials contain the impacts of energy and auxiliary materials as well. For example, fertilizer is used to produce crops: when crops are accounted for, fertilizer will be part of their cradle-to-grave chain and therefore should not be accounted for separately as well, in order to avoid double counting.
- Resources vs. finished materials: the DMC contains resources, finished materials and products, which should all be translated to the level of finished materials in order to avoid omission, miscounting or double counting
- Included and excluded materials: for all materials included, information needs to be available both on material flows and on impacts. Surprisingly, the former was the main limiting factor: especially small-scale (and sometimes high-impact, e.g. some heavy metals like platinum, palladium and rhodium) materials have not been included due to missing data.
- Weighting: the end-result of the exercise was, per material, a score on impact potential on 13 impact categories. Adding over materials is thus easy, but adding over impact categories - mandatory for an overall indicator of environmental pressure - requires weighting, a well-known problem in LCA.

Results. The result of applying the EMC methodology to the 28 countries included in this study shows, in the first place, that there are large differences between countries. The levels of EMC/capita and EMC/• vary by a factor of 2–5. The most important explanation lies in the differences between the structures of the economy. Countries with a relatively large or intensive agricultural sector have the highest EMC score. These are different from countries with a high DMC, except for Ireland. It is, however, difficult to attach a meaning to these differences. Should a country change its economic structure, or copy other countries? This is at least open to debate. While country comparisons suffer from interpretation problems with regard to the absolute value of EMC, the interpretation of time series within a country is less problematic. Given a certain structure of the economy, a development towards a less impact intensive economy can be regarded as positive. Here, too, are clear differences between countries. Some countries show a clear decrease in their EMC/capita, others a clear increase, yet others remain quite stable. The largest increase is visible in Southern European countries as Portugal, Spain and Greece. For the 28 countries in total, the EMC/capita is quite stable.

The EMC/• however shows a clear down-going trend over time, as shown in **Fig. 5**. Most countries also show this trend, with different rates of improvement. This means that the EU economy is becoming more eco-efficient. **Fig. 6** shows that the trend is also down-going for DMC/•. However, a comparison of Fig. 5 and 6 shows that the type of materials that dominate the indicators are different in EMC and DMC. EMC is dominated by the impacts due to consumption of biomass, while DMC is dominated by the consumption of construction minerals.

Added value, limitations and problems of combining tools. The added value of using a combination of tools in this case is that the potential environmental impacts of the different materials or resources can be considered as well as the weight or volume of their use.

The uncertainties of basic MFA data and the derived DMC also apply to the EMC. These are especially large for construction minerals: statistics on sand and gravel, being an important part of DMC, are notoriously unreliable and

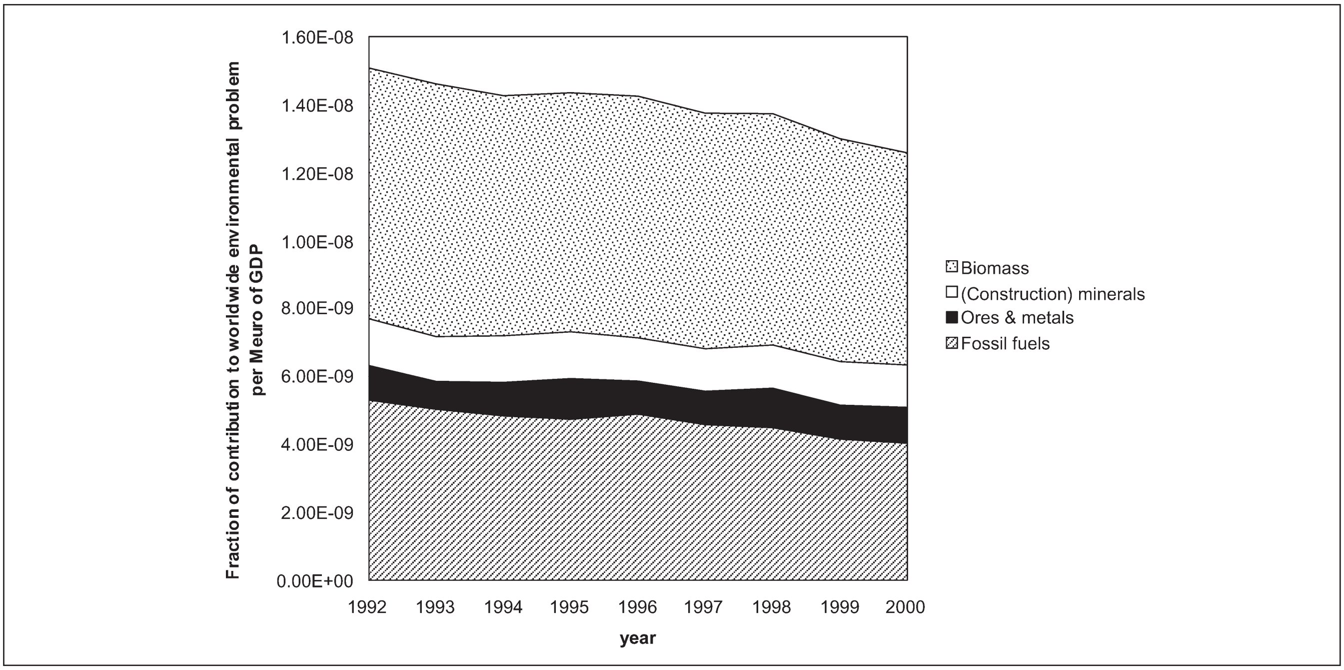

Fig. 5: EMC per million Euro for EU-25 and AC-3, 1992–2000 (AC-3: Accession Countries are, in 2004, Romania, Bulgaria and Turkey)

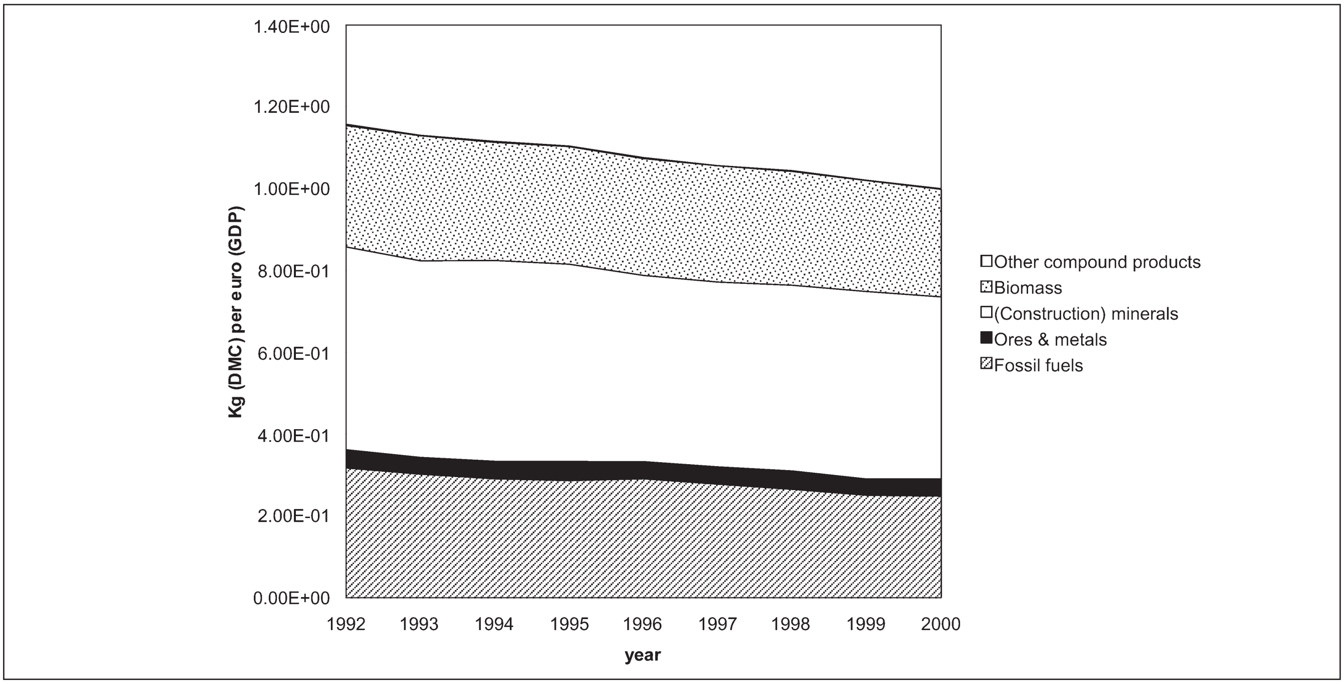

Fig. 6: DMC per Euro for EU-25 and AC-3, 1992–2000 (AC-3: Accession Countries are, in 2004, Romania, Bulgaria and Turkey)

unharmonized between countries. Additional uncertainties and restrictions arise from the use of LCA data. The LCA process data are averages for Western Europe, implying, on the one hand, that differences between countries are not expressed while, on the other hand, efficiency improvements cannot be seen over time that do not result in a lower material consumption (such as the application of end-of-pipe technologies). The LCA database is updated once a decade rather than once a year. Basic assumptions in the LCA database with regard to recycling and allocation are difficult to detect and may be open for improvement. Regarding the LCA impact assessment data, there are large differences in quality between the different impact categories. While global warming potentials are based on internationally agreed-upon studies, large uncertainties exist in the impact categories related to toxicity. The LCA Impact Assessment methodology is not well developed for land use. Depletion of resources of a biotic nature, e.g. wood and fish, is not included at all; at this moment there is no consensus on how to derive impact factors for this impact category. Despite these omissions and uncertainties, the addition of LCA data in our view is still relevant, bringing the MFA based indicator a step further in the direction of potential impacts. Both for MFA and LCA databases, improvements should and probably will be made over time, allowing for more reliable indicators. Both research and development areas are alive and many experts are working on it, which ensures a highly dynamic development field.

2 Discussion

Based on the experiences from these three case studies, an attempt is made to derive general guidelines on the relation between tools and questions.

First, we see that there are indeed questions posed by different types of decision-makers that have been approached by using a combination of tools. That is to say, using several tools to address a complicated problem is not only a theoretical proposal, but also something that has been applied successfully in a variety of practical situations.

A second aspect is that the results of these tools are not always stand-alone results; they are sometimes combined to form indicators, e.g. the EMC, but also indicators of eco-efficiency (cf. Huppes and Ishikawa 2005), and so on. Thus, using several tools in combination does not necessarily lead to an increased information supply to decision-makers. Instead, it may contribute to the comprehensibility and ease of interpretation of the information that would have been provided by using a single tool.

Thirdly, it is natural to seek guidance on combining tools, in the same way that guidance has been provided on how to apply LCA or MFA in concrete cases. We think, however, that the essential idea of using a combination of tools is exactly the fact that the research question is not a simple question. It by definition escapes a simple classification. For a person with a broken leg, there is a clear protocol, and there is another protocol for a person with brain damage. For a patient with both problems, a more individually tailored solution must be found. The same idea applies to the real-life complex problems that contain aspects of products, materials, risks and/or costs. Although we do not deny that the ideas presented in Section 1 may contain ideas or indicators which may successfully be transferred to other problems, we see no general recipe here.

While there may not be a general recipe for combining specific tools, a categorization of the various possibilities of combining tools in general may still be useful, thus allowing for consistent combination choices. Then, a final question is how one may categorize the tools found in practice in a conceptual scheme, like that by Udo de Haes et al. (2004), comprising extension, the toolbox, and hybrid analysis.

- The example of wastewater treatment could be described as using a toolbox: SFA and LCA are applied, and their methods and/or results are not formally integrated, but presented separately.
- The example on prioritization may be seen as an example of hybrid analysis: the results of an LCA and an LCC are combined to form an integrated indicator.
- The example of decoupling could be regarded as a form of extension: MFA is extended with an impact assessment phase (one might refer to this as doing an MFIA), like LCIA is an extension to the LCI. Note that MFA can also be extended in quite another way, e.g. linking it to an ERA approach as applied for a number of heavy metals (Van der Voet et al. (2000).

Considered in this way, we end with the following messages:

- The extension of tools, like LCA and MFA, is an ongoing activity where mutual learning and critical adoption and adaptation of ideas and methods from LCC, ERA, etc., as well from neighboring fields like economics, ecology, decision analysis, etc., can provide an input that will enable the construction of ever more sophisticated tools. This, we speculate, will mainly be a topic of academic concern, and we foresee that new PhD theses will continue to focus on this element.
- The use of a toolbox is an idea that will remain necessary in addition to the extension of tools, particularly if questions are at stake that require different system boundaries and (spatial and temporal) characteristics, as is the case for questions that can only be answered by applying an ERA as well as an LCA. However, it leaves the decision-maker with the problem of interpretation and prioritization. Help must be found here in established techniques, like multi-criteria analysis.
- The development of hybrid analysis has been pursued for some time in combining LCA with IOA. We think, however, that it may provide a useful addition to resolving the limitations of using a single tool, while facilitating the interpretation by the decision-maker in the form of offering indicators of the ratio between the results of two tools, like eco-efficiency.

Helias' brainchild, the Human and Ecological Life cycle tools for the Integrated Assessment of Systems, has started to show its usefulness and, hence, has demonstrated its right for existence. It now comes to nourishing it into maturity.

References

Assies JA (1998): A risk-based approach to life-cycle impact assessment. J Hazard Mat 61, 23–29

Cowell SJ, Fairman R, Lofstedt RE (2002): Use of risk assessment and life cycle assessment in decision making: A common policy research agenda. Risk Anal 22 (5) 879–94

Eurostat (2001): Economy-wide Material Flow Accounts and derived Indicators – A Methodological Guide. Methods and Nomenclature series, Luxembourg

Guinée JB (1995): Development of a methodology for the Environmental Life-cycle assessment of products, with a case study on margarines. PhD thesis, Institute of Environmental Sciences (CML), Leiden

Guinée JB (ed), Gorrée M, Heijungs R, Huppes G, Kleijn R, Wegener Sleeswijk A, Udo de Haes HA, de Bruijn JA, van Duin R, Huijbregts MAJ (2002): Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Kluwer Academic Publishers. Dordrecht (Hardbound, ISBN 1-4020-0228-9; Paperback, ISBN 1-4020-0557-1; see also <http://www.kap.nl/prod/b/1-4020-0228-9>

Heijungs R (1997): Economic drama and the environmental stage. Formal derivation of algorithmic tools for environmental analysis and decision-support from a unified epistemological principle. PhD thesis, Institute of Environmental Sciences (CML), Leiden

Huppes G (1993): Macro-environmental policy: principles and design. With cases on milk packaging, cadmium, phosphorus and nitrogen, and energy and global warming. PhD thesis Institute of Environmental Sciences (CML), Leiden. Amsterdam, Elsevier Science Publishers, 430 pp, ISBN 0-444-81657-7

Huppes G, Ishikawa M (2005): A framework for eco-efficiency analysis. J Ind Ecol 9 (4) forthcoming

Koning A de, Kleijn R, Duin R van (2001): Methodieken voor de prioritering van milieumaatregelen bij Zeeuwse bedrijven (Methods for prioritizing environmental measures at companies in the Dutch province of Zeeland). CML-report 155, 22.69, ISBN 90-5191-134-3

Moll S, Bringezu S, Schütz H (2003): Resource Use in European Countries – An estimate of materials and waste streams in the Community, including imports and exports using the instrument of material flow analysis (Zero Study). Copenhagen, European Topic Centre on Waste and Material Flows <http://waste.eionet.eu.int/mf/3>

Olsen SI, Christensen FM, Hauschild M, Pedersen F, Larsen HF, Tørsløv J (2001): Life cycle impact assessment and risk assessment of chemicals – A methodological comparison. Environ Impact Ass Rev 21, 385–404

Provincie Zeeland (2001): Handreiking Priotering van Milieumaatregelen (Guidelines for prioritizing Environmental Measures). Provincie Zeeland, Directie Ruimte, Milieu en water

Saouter E, Feijtel TCJ (2000): Use of life cycle analysis and environmental risk assessment in an integrated product assessment. Environmental Strategies, Nordic Workshop Vedbaek 1999. ISBN 92-893-0464-2. In: Hauschild M, Olsen SI, Poll C, Bro-Rasmussen F (eds, 2000), Risk Assessment and Life Cycle Assessment, TemaNord 2000:545. Nordic Council of Ministers, Copenhagen, pp 81–97

Smet B de (ed) (1990): Life-cycle analysis for packaging environmental assessment. Proceedings of the specialized workshop, 24–25 September 1990 in Leuven. Procter & Gamble Technical Center, Strombeek-Bever

Sonnemann G, Solgaard A, Saur K, Udo de Haes H, Christiansen K, Jensen AA (2001): Life Cycle Management: UNEP-Workshop. Sharing Experiences on LCM. Int J LCA 6 (6) 325–333

Suh S (2004): Materials and energy flows in industry and ecosystem networks: Life Cycle Assessment, Input-Output Analysis, Material Flow Analysis, Ecological Network Flow Analysis and their combinations for Industrial Ecology. PhD thesis, Institute of Environmental Sciences (CML), Leiden

Tukker A, Kleijn R, Oers L van, Smeets ERW (1998): Combining SFA and LCA: The Swedish PVC Analysis. J Ind Ecol 1 (4) 93–116

Udo de Haes HA, Suh S, Heijungs R, Huppes G (2004): Three Strategies to Overcome the Limitations of Life-Cycle Assessment. J Ind Ecol 8 (3) 19–32

Voet E van der (1996): Substances from Cradle to Grave: development of a methodology for the analysis of substance flows through the economy and the environment of a region, with case studies on cadmium and nitrogen compounds. PhD thesis, Institute of Environmental Sciences (CML), Leiden

Voet E van der, Guinée JB, Udo de Haes HA (eds) (2000): Heavy Metals: a Problem Solved? Methods and models to evaluate policy strategies for heavy metals. Kluwer Academic Press, series Environment & Policy, Dordrecht

Voet E van der, Oers L van, Nikolic I (2004): Dematerialization, not just a matter of weight. J Ind Ecol 8 (4) 121–137

Voet E van der, Oers L van, Moll S, Schütz H, Bringezu S, Bruyn S de, Sevenster M, Warringa G (2005): Policy Review on Decoupling; Development of indicators to assess decoupling of economic development and environmental pressure in the EU-25 and AC-3 countries. CML report 166, Department Industrial Ecology, Leiden

VROM – Ministry of Housing, Spatial Planning and the Environment (1994): Kosteneffectiviteit van milieumaatregelen in de industrie, beschrijving van de methodiek, deel A. VROM, publicatiereeks lucht & energie, nr 119

VROM – Ministry of Housing, Spatial Planning and the Environment (1998): Kosten en baten in het milieubeleid: herziene uitgave van de standaardmethode voor het definiëren en berekenen van de kosten van milieubeheer. VROM, publicatiereeks milieustrategie, nr. 1998/6

Wegener Sleeswijk A, Heijungs R, Erler ST (2003): Risk Assessment and Life-cycle Assessment: Fundamentally Different yet Reconcilable. Greener Management International 41, 77–87

Wegener Sleeswijk A (2001): General Prevention and Risk Minimization in LCA. A Combined Approach. ESPR – Environ Sci & Pollut Res 8 (1) 1–9

Wrisberg N (2000): CHAINET: Integrating environmental evaluation tools Environmental Strategies, Nordic Workshop Vedbaek 1999. ISBN 92-893-0464-2. In: Hauschild M, Olsen SI, Poll C, Bro-Rasmussen F (eds, 2000), Risk Assessment and Life Cycle Assessment. TemaNord 2000:545. Nordic Council of Ministers, Copenhagen, pp 99–112

Wrisberg N, Udo de Haes HA, Triebswetter U, Eder P, Clift R (2002): Analytical Tools for Environmental Design and Management in a Systems Perspective. The Combined Use of Analytical Tools. Kluwer Academic Publishers, Dordrecht

Received: December 8th, 2005
Accepted: December 8th, 2005
OnlineFirst: December 9th, 2005