ENHANCING THE APPLICATION EFFICIENCY OF LIFE CYCLE ASSESSMENT FOR INDUSTRIAL USES

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"Everything should be made as simple as possible, but not simpler"
(Albert Einstein, 1879 - 1955)

The pithiness of this quote disguises the fact that no one knows whether Einstein said it or not (this version comes from the Reader's Digest, October 1977). It may well be a precis of the last few pages of his "The Meaning of Relativity" (5th edition), where he wrote about his unified field theory, saying *"In my opinion the theory here is the logically simplest relativistic field theory which is at all possible. But this does not mean that nature might not obey a more complex theory. More complex theories have frequently been proposed… In my view, such more complicated systems and their combinations should be considered only if there exist physical-empirical reasons to do so."*

(retrieved, 9.6.2005 from: http://math.ucr.edu/home/baez/physics/General/occam.html)

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Abstract

Sustainable development is an issue that is gaining more and more relevance in all areas of society, though specifically in industry. In order to move towards the goal of sustainability, life cycle thinking is an essential element. For the implementation of life cycle thinking in industry life cycle management (LCM) has been proposed as the general concept. However, the assessment of environmental impacts with the method of life cycle assessment, which is essentially the only tool available for this purpose, has been limited in industrial practice due to involved complexity and the resulting necessary effort and know-how. Therefore, this thesis proposed improved methods that enhance the application efficiency of LCA for industrial uses.

After a short introduction in Chapter 1, new developments for the more efficient application of LCA for environmental assessments are presented in Chapter 2 – both for situations where pre-existing data are available and for studies, where no or very limited information of the involved unit processes and elementary flows exist. This concerns the modeling based on reusable elements as well as limiting system boundaries by recommended cut-offs. Specific and easy to apply recommendations for using cut-off rules are proposed.

Chapter 3 explores another path, namely the usage of LCA models for life cycle costing. A consistent framework of the economic life cycle based assessment in sustainable development is proposed and tested. Essentially, one can conclude that the deployment of the systems approach and underlying model of LCA is extremely useful also for conducting economic analyses, while causing very little additional efforts.

Chapter 4 elaborates case studies, one for an automotive component, and one for the service of waste water treatment. Detailed LCA results are presented and discussed, and the aforementioned methods in regards to a more efficient LCA 'from scratch' and relating to life cycle costing are tested and demonstrated.

The thesis concludes with some concise recommendations for future research and development activities in relation to a better usage of LCA and related life cycle approaches.

Zusammenfassung

Das Leitbild der nachhaltigen Entwicklung (engl. sustainable development) nimmt mehr und mehr an Bedeutung zu, auch oder gerade in Zeiten schwieriger wirtschaftlicher Entwicklungen in vielen Regionen der Erde. Insbesondere in der Industrie wird Nachhaltigkeit zunehmend als Chance begriffen. Für die Umsetzung dieses Leitbilds spielen Lebenszyklusbetrachtungen (Ökobilanzen, engl.: life cycle assessments) eine entscheidende Rolle. In diesem Zusammenhang ist auch Einbindung dieser Ansätze in Managementprozesse und Betriebsabläufe von entscheidender Bedeutung. Ohne eine entsprechende Implementierung, die wiederum eine einfache und verlässliche Anwendung der entsprechenden Methoden voraussetzt, kann und wird Nachhaltigkeit nicht erfolgreich umgesetzt werden können. Existierende Ökobilanzansätze haben oftmals das Problem, dass sie sehr komplex und ihre Anwendung mit einem hohen Aufwand verbunden ist, was das Einsatzgebiet stark einschränkt. Deshalb werden in dieser Dissertation Ansätze vorgeschlagen und entwickelt, die die Anwendung der Ökobilanzmethode in der industriellen Praxis vereinfachen sollen.

Nach einer kurzen Einführung (Kapitel 1) werden in Kapital 2 neue Ansätze zur Vereinfachung der Ökobilanzanwendung eingeführt und entwickelt. Dabei ist zu unterscheiden zwischen einem Ansatz (Modular LCA), der das Ziel hat, die Anwendung der Methode auf Basis vorhandener Daten zu erleichtern und Ansätzen, die sich mit den sogenannten Abschneidekriterien befassen. Letzteres ist insbesondere relevant, wenn man das zu modellierende Produktsystem nicht kennt und entscheiden muss, wann die Datenaufnahme abgebrochen werden kann. Einfach anzuwendende Vorgehensweisen und daraus abgeleitete Empfehlungen für die Anwendung von Abschneidekriterien werden vorgestellt.

Kapitel 3 dagegen beschäftigt sich mit der Nutzung von Systemmodellen der Ökobilanz für die Lebenzykluskostenrechnung (engl. life cycle costing). Hier wird aufgezeigt, wie das Ökobilanzmodell effizient und einfach für die Berechung von Kosten und die Berücksichtigung der ökonomischen Dimension der Nachhaltigkeit eingesetzt werden kann.

Vervollständigt wird die Arbeit durch Ökobilanzfallsstudien aus dem Automobilbereich und für die kommunale Abwasserreinigung (siehe Kapitel 4). Die vorher entwickelten Ansätze hinsichtlich Abschneidekriterien und die Methoden der Lebenzykluskostenrechnung werden angewandt. Abschliessend werden Empfehlungen für weitere Forschungsarbeiten für den verbesserten Einsatz der Ökobilanzmethodik in der Industrie und im Zusammenhang mit weiteren Aspekten der Nachhaltigkeit abgeleitet.

1 Introduction

1.1 From Sustainable Development to Life Cycle Management

Sustainable development is an essential element in the survival and flourishing of mankind and, therefore, also the basis for present and future industrial and business success [Fussler and James 1996]. This can be exemplified by the activities of the World Business Council for Sustainable Development [WBCSD 2001] or other driving forces in industry. For instance, William Clay Ford, Jr., chairman of Ford Motor Company, declared his vision that future mobility has to be achieved in a sustainable way: "Ford Motor Company once provided the world with mobility by making it affordable. In the 21st century, we want to continue to provide the world with mobility by making it sustainable" [Ford 2000]. Furthermore, chemical corporations have taken a lead in the sustainable development initiative, for example, through the Responsible Care Program of the American Chemistry Council [ICCA 2000].

The focus of sustainable development (or 'sustainability') in industry is shifting from looking at environmental and other impacts as an additional issue with additional costs to an area of opportunity. It is, increasingly, seen as a general concept that should be integrated in all relevant activities. This may be exemplified by the statement of Travis Engen, President and CEO of Alcan Inc.: "Whether it's through the design and application of innovative products or by building long-term partnerships through our stakeholder engagement efforts, we are working to integrate sustainability into all aspects of our business" [Engen 2004]. However, as stated in the last phrase by Engen industry and other stakeholders are *working* on the integration of sustainability into business related and other activities. Sustainable development as defined by [WCED 1987][1] is far from being reached in today's global industrial society: "Sustainable development is development that meets the needs of present without compromising the ability of future generations to meet their own needs". A key aspect of sustainable development is the consideration of ecological, economic, and social 'pillars' or 'dimensions'. The goal of sustainable development is to balance these dimensions, so that long term flourishing of mankind is ensured.

Perhaps the most relevant weakness of the generally accepted framework of sustainable development is the fact that it is neither directly applicable, nor can it be

[1] While this is the most widely used definition for sustainable development, there have been other definitions and concepts before (see e.g. [Royston 1979], who already developed a life cycle oriented concept of sustainable development in the 1970s). However, it is important to note that there might be differences in the coverage/extend of the different definitions, but there are usually no general contradictions.

easily measured and traced, via indicators. Furthermore, it can contain ‘everything and nothing’. Some critics have, also, referred to sustainable development as a ‘buzzword for zero content’. Though these are provocative and extreme point of views, they certainly contain some aspects of truth, or at least concern.

Within the area of product oriented environmental and sustainability management, life cycle management (LCM), attempts to address these (perceived) weaknesses. LCM is the application of ‘life cycle thinking’ to modern business practice, with the aim to manage the total life cycles of an organization’s goods and services[2] towards a more sustainable consumption and production [Jensen and Remmen 2004]. It is an integrated framework of concepts and techniques to address environmental, economic, technological, and social aspects of products and organizations. LCM, as any other management pattern, should be applied on a voluntary basis and can be adapted to the specific needs and characteristics of individual organizations [Hunkeler et al. 2004].

Briefly, LCM, with its toolbox and decision-oriented goals, seeks to render sustainability accessible, quantifiable, and operational [Hunkeler et al. 2004]. LCM aims at integrating environmental concerns into industrial and business operations by considering off-site, or supply chain, impacts and costs. LCM seeks to increase the competitiveness of new, and existing, products by examining advantages, and business risks, associated with the environmental and social aspects of a product, throughout its life cycle. Therefore, LCM can be seen as a means of putting sustainable development to work within a firm, given its temporal and financial constraints.

While being much more focused and operational than the general concept or vision of sustainability, LCM, however, also poses challenges which limit its application in practice. These challenges are further elaborated in the following Section, leading to the identification of research topics that are later addressed in this thesis.

[2] Goods and services or their utilities can be summarized under the term “product” [WCED 1987].

1.2 Challenges of Implementing Life Cycle Management

While the roots of life cycle management are more than 25 years old and can be traced back to several sources (e.g. [Royston 1979; Öko-Institut 1987]) it has gained momentum in the recent years, both from a scientific point of view as well as in regards to the implementation in industry (see [Heinrich and Klöpffer 2002; Hunkeler et al. 2004; Jensen and Remmen 2004]).

The potential benefits of implementing life cycle approaches are generally accepted, though in regards to the level and scope of implementation LCM is still in its infancy. This is in particular evident, if one compares LCM for instance with activities related to site-oriented environmental management (ISO 14001, EMAS, and others). Site-oriented environmental management is considered standard business practice in many sectors and regions of the world, while equivalent activities related to the product focus of life cycle management are very scarce, may they be termed LCM, product stewardship, product oriented environmental management, etc.[3]

A major reason for this low level of implementation is the complexity that is invariably involved when the holistic life cycle view related to products is targeted. Central are questions in regards to the measurement of the sustainability performance of products. Only if products and related measures taken can be assessed and analyzed, can progress or negative developments be reported and used for actual decision-making and management purposes.

Looking uniquely at the environmental dimension, life cycle assessment (LCA), the only internationally standardized environmental assessment method [ISO 14040: 1997; ISO 14041: 1998; ISO 14042: 2000; ISO 14043: 2000], is the primary and established tool for assessing the environmental performance of a good or service within LCM. However, the application of LCA is limited, because it is a rather sophisticated method, and the direct usage of the method and employment for decision-making is absolutely non-trivial and needs expert support. In addition, the effort needed can be quite high, which poses additional barriers for its application (for a further elaboration of the specific challenges related to the application of the LCA method see Section 2.1).

Standardized assessment methods and procedures for the economic and social dimensions are still lacking [Klöpffer 2005], rendering a wide-spread and generally accepted implementation even more difficult. Current research e.g. within SETAC [Rebitzer and Seuring 2003] and the UNEP/SETAC Life Cycle Initiative [Töpfer 2002]

[3] Various organizations use different terminologies without addressing inherently other issues. In this thesis, life cycle management is used as the general term, covering equivalent or similar concepts as well.

aims to close these gaps, though specifically work on the social dimension is at the outset [Klöpffer 2005].

To address some of the aforementioned challenges and to *enhance the application efficiency of LCA*, i.e. to make LCA more easy and rapidly usable with less resources, this thesis attempts to further analyze and develop the LCA method in the light of the aforementioned application problems (see Section 2) in conjunction with the associated case studies (see Section 4).

In addition to dealing with the question on *how the use of LCA can be facilitated by improved methodological procedures of LCA itself*, this thesis also examines the *economic pillar of LCM and introduces a life cycle costing (LCC) method that is based on the life cycle inventory of an LCA and embedded within the LCM framework* (see Section 3 and the case studies of Section 4). For this second of this work part the goal is to advance both the assessment of the economic dimension within sustainability and to also make additional use of data and models created by the LCA methodology. If LCA data and models can find further use as a basis for economic assessments, this, likely, will also highly *enhance the overall application efficiency of life cycle approaches* and thus LCM and sustainability.

2 Development of a Consistent Framework for the Simplified Application of LCA

2.1 Background and Rationale

Life cycle assessment (LCA) [ISO 14040: 1997] has evolved into a powerful and robust methodological framework for conducting environmental assessments of products[4]. However, for many uses, specifically in industry, the time and costs for an LCA study are often judged not to correspond to the possible benefits of the results [SETAC Data WG 1999]. This is particularly relevant if it concerns 'one-time studies', where the collected data and results or elements of these are not reused for further applications. The efforts associated with reusing elements of LCA models for new and rapid LCAs, however, are also often seen as barriers for the widespread and routine deployment of the LCA methodology in industry. There is even concern "whether the LCA community has established a methodology that is, in fact, beyond the reach of most potential users" [Todd and Curran 1999].

The aforementioned limitations are particularly acute within contexts where a rapid decision is required, such as during a Design for Environment (DfE) process [Brezet and van Hemel 1997, p. 200; de Beaufort-Langeveld et al. 1997, p. 10]. Therefore, in order to provide efficient and reliable decision support in the available period of time and with the available resources (manpower, know-how, money), the application of LCA has to be as simple as possible, while being as accurate as necessary.

The LCA framework consists of the steps goal and scope definition, life cycle inventory analysis, impact assessment, and interpretation [ISO 14040: 1997]. "In order to develop effective simplification methods, it is obvious to focus on the life cycle inventory analysis, which is typically the most time consuming phase, with the greatest potential for savings" [de Beaufort-Langeveld et al. 1997, p. 19]. Part of the scope definition and the most significant as well as the most time and cost consuming element in the inventory analysis is the establishment of the 'product system model'[5] [Rebitzer and Fleischer 2000]. A 'product system' is the "collection of materially and energetically connected unit processes which performs one or more defined functions" [ISO 14040: 1997], while the 'product system model' is the model representation thereof [Rebitzer 2000].

To establish the complete system model for a 'detailed' or 'simplified LCA' is extremely non-trivial because all involved processes and the relating inputs and outputs have to

[4] The term 'product' encompasses material goods as well as services, or their utility [WCED 1987]
[5] Terms written in single quotation marks that are not self-explanatory, are defined in the glossary.

be identified, structured, and calculated. If one expands the system model in order to avoid allocation, which is methodological preferable [ISO 14041: 1998, Wenzel et al. 1997, 72], the necessary effort in particular for data collection further increases. The issues data collection, data quality, and the definition of system boundaries, all of which are directly associated with the product system model, are viewed to be the greatest difficulties for the application of LCA within companies [Frankl and Rubik 2000, p. 77]. In short, the effort for modeling the product system is the biggest barrier for the efficient application of LCA and thus for the implementation in industry.

The aforementioned challenges and barriers form the rationale for briefly analyzing proposed (previously existing) approaches to make the application of LCA easier (see Section 2.2). Building on this analysis, a framework and new methods that improve the application efficiency and thus facilitate the widespread and routine use of LCA are proposed and elaborated (Sections 2.3 to 2.4). These developments focus on a more efficient modeling of the product system in the life cycle inventory analysis, targeting both

- LCA applications that are based on previous and existing LCA data and models and
- Applications where no data and models (unit processes, partial life cycle inventory (LCI) models, cradle-to-gate LCI data ('building blocks'), or life cycle impact assessment (LCIA) results, etc.) are available.

In practice these two cases usually do not exist in isolation, but rather in combination. Often for parts of the product system there are data and models available, while for others new elements have to be modeled. However, for purposes of systematic analysis and development, these two cases are treated separately in the presented research. The later combination of the different approaches in practical applications is rather straightforward and trivial and outside the scope of this thesis.

While the deployment of existing life cycle impact assessment (LCIA) methods also poses non-trivial problems for the LCA practitioner, it is less a question of the effort involved, but rather an issue of application rules (for instance regarding the assignment of characterization factors to elementary flows) and nomenclature (see e.g. the discussions in [Frischknecht et al. 2004b]), and the potentially resulting uncertainties due to inconsistencies and incompatibilities. Therefore, and due to the focus on product system modeling, the research presented in this thesis does not further examine application problems of the LICA phase of LCA, but uses LCIA as given by existing conventions, procedures and methods.

2.2 State of the Art in Simplified Product System Modeling

[ISO 14041: 1998] states that “the selection of inputs and outputs, the level of aggregation within a data category, and the modeling of the system shall be consistent with the goal of the study”. This indicates that the processes and inputs and outputs that significantly affect the results of the study should be included in the model of the product system. By trying to follow these recommendations the LCA practitioner faces a paradox: if one does not already have a detailed LCA of the product in question, or knowledge about its results, available, one will not know the most significant processes before the impact assessment has been performed. Identifying the most significant processes concerning the environmental interventions is just the aim of the study. The ISO 14040 series of standards [ISO 14040: 1997; ISO 14041: 1998; ISO 14042: 2000; ISO 14043: 2000] forms the framework and sets the requirements for developing the product system model, but it does not give methodological guidance on how to do it.

The modeling of the product system can be facilitated in principal via two strategies, depending on the goal and scope of the study and existing data and models (unit processes or sub-systems of the product system as for instance cradle-to-gate energy production and gate-to-gate industrial process chains):

- Using existing unit process data sets, existing data models, or aggregated data (building blocks) from previous studies or existing databases. Obviously, the prerequisite is that the required data collection and modeling has been carried out previously and that the results are available and applicable to the specific LCA in question.
- Simplifying the product system model by focusing on the key issues and by eliminating the need for complete data collection and process modeling. Such a strategy is attractive, if no building blocks, unit process data, or models are available (see above).

The former strategy is often not termed ‘simplifying’ in the narrow sense, however it is a way to simplify the application of LCA, often also through the use of generic data or data proxies, and is, therefore, included in this thesis. In practice both strategies are often combined together, i.e. usage of existing data and models where these are available in the desired format and quality together with elements of the product system model where simplification methods in the stricter sense are employed.

In this context one has to mention that this thesis addresses only those modeling issues where simplification is systematically, consciously and explicitly carried out, not the implicit simplification inherent in any LCA model, because it is not possible to simulate reality completely due to the level of complexity of the industrial systems. A truly complete modeling of the product system would, for many LCAs, virtually lead to the modeling of the complete or large parts of the economy, resulting in a simulation of the global man-made system and an almost infinite effort, which is not feasible for decision-making[6].

One could argue that the 'complete' modeling also has to make simplifications, which is the case in reality, where there are always cut-offs. Along this line of argument there might be just different degrees of simplifications in contrary to a view which differentiates between detailed and simplified LCAs. From the viewpoint of the author, however, there is an inherent difference between trying to model 'as complete as possible' without considering systematic simplifications and the use of defined and possibly iteratively changing cut-off rules or other ways to create and apply a model as efficiently as possible while covering the decision relevant environmental impacts. An example of the approach to model as complete as possible without introducing systematic cut-offs or other boundary conditions is the research project that cumulated in the ecoinvent 2000 database [Frischknecht et al. 2004a, 2004c].

In the following first the state of the art in using existing data and models is examined (Section 2.2.1). Subsequently existing methods for the aforementioned explicit simplification of LCA, for cases where data and models do not exist, are analyzed (Section 2.2.2).

2.2.1 Using Existing Data and Models

Product system models usually rely on a subset of process types common to nearly all systems, namely energy supply, transport, waste treatment services, and the production of commodity chemicals and materials. As a cause of global markets, many of these process types are even identical, be it oil extraction in the Middle East or steel manufacturing in Asia. Other processes show typical continental, national, or even regional properties such as road transportation, cement manufacturing, and agricultural production, respectively. [Rebitzer et al. 2004a]

In order to increase the efficiency in carrying out an LCA, electronic databases have been created that cover the more commonly used goods and services. Many of these databases provide LCI data on the level of life cycle inventory results (e.g., the

[6] Some argue that this complexity problem can be easily solved by methods based on macro-economic input-output analysis (see e.g. [Lave et al. 1995; Hendrickson et al. 1998; Joshi 1999]). While such methods can assist in identifying areas and sectors of environmental impacts and can give some indication on their significance (see e.g. [Suh 2004a; Yurika et al. 2004]), they are clearly not suitable for differentiation between products of one industrial sector, e.g. for assessing alternatives in product design (see also the discussions by [Heijungs and Suh 2002, p. 124]. It is clear that the necessary level of specificity is not possible with input-output analysis due to its inherent structure. Data quality and data and model uncertainties are other issues of input-output LCA that make this method very difficult to apply for direct decision-support and environmental improvements in industry.

aggregated resource consumptions and emissions per 1 kg of material produced, also called ‘building blocks’). Prominent examples for such databases, which are embedded in modeling software tools, are those of Gabi [IKP/PE 2004], TEAM [Ecobilan 2004] and SimaPro [Pré 2004]. Some databases, such as the Swedish SPINE [CPM 2004] and the Swiss "Ökoinventare von Energiesystemen" [Frischknecht et al. 1996], and its successor ecoinvent 2000 [Frischknecht et al. 2004a], also offer data on a disaggregated unit process level (i.e., LCI data per technological process). An overview of existing public databases in electronic and other forms usable for modeling product systems has been compiled by [Norris and Notten 2002]. Additionally, a survey of existing software tools and some life cycle inventory databases, both commercially and publicly available, has been recently published by [Siegenthaler et al. 2005].

Many industry sectors are also pro-actively meeting requests for data to be used in LCAs. The Association of Plastics Manufacturers in Europe (APME) can be considered the pioneer in making data publicly available [Matthews and Fink 1993], but also other industry associations have been actively collecting and providing data since the early 1990s. An indicative list of trade associations providing life cycle inventory data is given in Table 2-1.

Table 2-1: Indicative, non-exhaustive, list of LCI data collected and published by industry associations (modified from [Rebitzer et al. 2004a])

Database ‘name’ (if any) or designation	**Geographical scope**	**Managed by**	**‘Format’**	**Further information**
Ecobalances of the European plastic industry	Europe	APME	Text-format	http://www.apme.org
Environmental Profile report for the European aluminium industry	Europe	European Aluminium Association (EAA)	Hardcopy	http://www.aluminium.org
IAI report on inventory data for the worldwide primary aluminium industry	Global	International Aluminium Institute (IAI)	Text-format	http://www.world-aluminium.org
FEFCO European database for corrugated board - life cycle studies	Europe	FEFCO	Hardcopy Or ‘Spold’	http://www.fefco.org
Life cycle assessment of nickel products	Global	Nickel Development Institute	Text-format	http://www.nidi.org
LCA of the steel industry	Global	IISI	Hardcopy	http://www.worldsteel.org/env_lca.php

In addition to several national-level database development activities in Japan, USA, Canada, Germany, Italy, Switzerland, and Sweden, activities by software providers, and industry associations, some international coordination projects are under way. For example, one of the goals of the LCI Program of the UNEP/SETAC Life Cycle Initiative is to establish “a peer reviewed and regularly updated database or information system for the Life Cycle Inventory for a wide range of unit processes or subsystems (‘building blocks’) like electricity, transportation, or commonly used materials” [Udo de Haes et al. 2002].

There are also approaches, where the concept of building blocks (see above) is developed further to also integrate the impact assessment phase, usually in a highly aggregated form. The best known examples of this are the Eco-indicator 95 [Goedkoop et al. 1996] and Eco-indicator 99 Manuals for Designers [Goedkoop et al. 2000], which provide LCIA results in the form of one score assessments for the cradle-to-gate production of standard materials, selected manufacturing, transport, energy generation, and disposal processes. The underlying LCI data and models are building blocks from the aforementioned and other LCI databases.

All these presently and in the future existing data, models, and LCIA results of building blocks, can be readily used for LCAs and facilitate the application greatly. This support, however, is obviously limited to those processes where data are available and where the available data and models meet the requirements regarding system boundaries, methodological choices such as allocation, choices in impact assessment (for those cases where this is integrated), and compatibility to data from other databases or to those specifically collected for the study at hand.

In addition to using publicly or commercially available databases containing generic life cycle inventory data and models, it is of course also possible to create in-house databases and models for subsystems and complete systems that are used more than once. This re-use of data and models via commercially available or tailor-made software systems is a common approach mainly of multinational companies. Based on previous products and the respective LCI models are product modifications, new products, and alternatives assessed, using common elements of the product system models. Over time, more and more products can be covered and the modeling of new products and alternatives may be based on more and more pre-existing and updated work. The use of such systems, applied to Design for Environment processes, is elaborated e.g. by [Finkbeiner et al. 2002].

While the approach of tailor made, continuously growing, databases is proven, it is limited to firms that have internal LCA experts available (and/or that have established a strong and continuous cooperation with specialized consultants), who develop the required models and update and maintain the database regularly. In addition, it requires a centralized approach, since the software systems are expert systems and cannot be easily used and maintained by non-LCA practitioners. The update of the system, possibly with interfaces to other internal databases, needs expert support.

Additionally, the direct usability of the calculation results of LCAs for e.g. product development, purchasing, sales and marketing, or site-oriented environmental management is limited, since preceding interpretation and evaluation steps are required. As a consequence, the utilization of the results still requires the involvement of the LCA expert, which limits the areas of application due to the limited resources (even large multinational firms usually only have a small team of less than five persons with LCA know-how, if any).

It is desirable to have a system that allows an easier use of existing LCA data and models, also for the non-LCA expert, so that life cycle thinking can be applied more widely and on different levels within an organization. If this ease of application can be achieved, the internal or external LCA expert can focus on the creation of such models, training, and specific decision support functions. For smaller companies, the same is true, since they also need simple models for non-experts. In these cases the internal or external experts (consultants) have to support the set up of the system and the provision of new data and models.

The application to decision-making ('what to do with the results and interpretations of the LCA study?'), however, should be in the hands of management and should not require direct involvement of the experts in all cases, i.e. not for regular decision-making. Consequently, there is a need to make LCA models and their results better usable in everyday industrial practice.

While the development of powerful software systems and extensive databases in the recent years have contributed highly to a better application efficiency of LCA, there is still a need to make LCA better accessible and usable. Specifically, the efficient application of LCA data and models as well as calculated results by non-LCA experts without or with only minimal support from the regular LCA practitioner needs improvement. The method developed in Section 2.4 ('Modular LCA') aims at this improvement. It presents a new approach to enhance the application efficiency of LCA, for cases where data and models are available.

The following Section continues with an analysis of the state of the art for the second strategy identified in the beginning of Section 2.2, the modeling of the product system if no data or models are available.

2.2.2 Modeling Without Existing Data and Models

2.2.2.1 Methodological Context

The use of existing data, as outlined in the previous Section, is one foundation of efficient product system modeling. However, if suitable data do not exist for (parts of) the system, the respective models have to be established from scratch. In these cases the question 'what to include in the product system model without having the effort for the study exploding?' is of crucial importance. This question has been a central issues from the beginning of the development of the LCA methodology (see [Fava et al. 1991; Consoli et al. 1993; Hunt and Franklin 1996; Weitz et al. 1996]). However, it was only systematically addressed much later in working groups of SETAC Europe [de Beaufort-Langeveld et al. 1997] and SETAC North America[7] [Todd and Curran 1999]. The publication of [Todd and Curran 1999] mainly examines specific approaches and discusses their advantages and disadvantages, while [de Beaufort-Langeveld et al. 1997] developed a systematic procedure for the simplification of LCA, which is outlined in Figure 2-1. In the Figure the term simplification is used for the overall procedure and the term simplifying for the second step within simplification (see below).

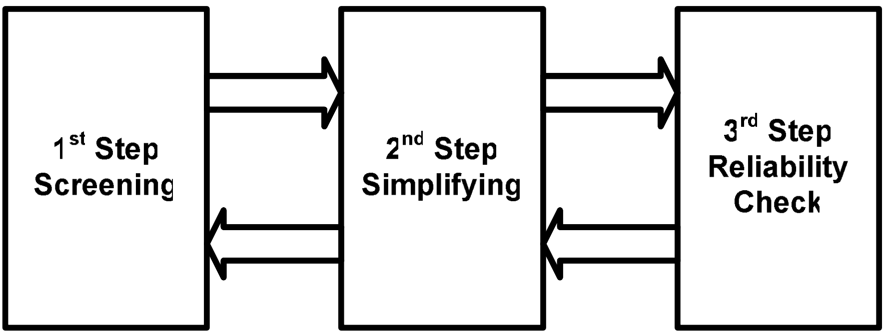

Figure 2-1: Simplification procedure for LCA
(based on [de Beaufort-Langeveld et al. 1997])

This procedure starts with a screening step, which is a pre-assessment of the system that guides the further data collection and modeling steps. The goal of the screening step is to identify those areas of the product system and/or key aspects of the life cycle that contribute significantly to the environmental impacts of the overall system. It is not

[7] In the report of [Todd and Curran 1999] the term 'streamlining" is used, which is considered synonymous with simplifying [de Beaufort-Langeveld et al. 1997].

meant to quantify the aspects, but rather identify the hot-spots and areas that should not be neglected in the LCA.

Screening is followed by the simplifying step in the narrow sense, which eliminates parts of the product system model (processes or flows) or can also be targeted at the life cycle impact assessment phase (which is not elaborated further here). The simplifying step leads to a quantification of the environmental impacts of the assessed product system. An alternative terminology for this second step could be 'targeting', which avoids the confusion of using both 'simplifying' and 'simplification' (see above). However, in this thesis the established terminology of [de Beaufort-Langeveld et al. 1997] is used in the following.

Finally, a reliability check concludes the procedure. This can involve sensitivity and dominance analyses as well as comparisons to other studies, analogy considerations, expert judgments, uncertainty assessments, etc. It has to be stressed that the procedure as illustrated in Figure 2-1 is not a linear step-by-step method, but that the different steps interact and that results of one step might lead to an adapted repetition of another, i.e. iteration (hence, the arrows in Figure 2-1 point in both directions).

The generally iterative nature of LCA has additional functions for simplification, including a formalized way to end the iterations faster [Schmidt 1996]. The goal is to get results that are 'good enough' for the goal and scope of the study, if higher uncertainties compared to results from detailed LCAs are acceptable (see also the short discussion on reliability checks in Section 2.2.2.4).

Because the research on the efficient application of LCA in this thesis employs the simplification procedure of [de Beaufort Langeveld et al 1997] as a starting point and concept, the relevant steps and the state of the art for the different methodological steps are analyzed in more detail in the following.

2.2.2.2 Screening

For screening purposes the following concepts exist [Rebitzer et al. 2004a]:

- Qualitative approaches: ABC hot spot screening [Fleischer and Schmidt 1997]; matrix methods, representing life cycle stages and stressors [Graedel et al. 1995; Todd 1996; Hunkeler et al. 1998]; checklists and expert panels [de Beaufort-Langeveld et al. 1997].
- Semi-quantitative methods: ABC/XYZ assessment, a statistically weighted hot-spot screening according to [Fleischer et al. 2001].

- Quantitative approaches: input-output LCA (see e.g. [Hall et al. 1992; Hendrickson et al. 1998; Suh and Huppes 2002]); assessment of single key substances; calculation of the cumulative non-renewable primary energy demand [de Beaufort-Langeveld et al. 1997; Fleischer and Schmidt 1997].

Qualitative matrix approaches and the use of non-renewable primary energy demand as a screening indicator are the most widely applied screening approaches. Matrix methods are especially preferable if detailed LCAs of similar product systems exist, with which conclusions can be derived based on the identification of differences to a well-known system. Non-renewable primary energy demand can be useful, because energy related data are readily available for many single processes as well as in aggregated forms and several environmentally important impacts are strongly linked to energy generation and consumption processes. For instance, on an average global scale, energy generation and consumption are responsible for about 90% of impacts on acidification, more than 80% on eutrophication, about 65% on global warming, and 60% on summer smog, all figures in relation to overall man-made impacts [Fleischer and Schmidt 1997]. These figures have been calculated based on statistical data [OECD 1989; Deutscher Bundestag 1990] that are not related to LCA studies, ensuring that the results are not biased by a possible focus of LCA data on energy generation processes[8]. However, care has to be taken, if significant human- or eco-toxicological emissions from non-energy related activities can be expected. In such cases, at least a qualitative screening of the emitted substances should be added (see above).

In the recent years, input-output LCA has gained in importance and has been developed to a stage that it is easily usable for screening purposes. Tools that can be used for screening are for instance LCNetBase [Sylvatica 1999], MIET/CEDA [Suh and Huppes 2002; Suh 2004c], OpenLC [Norris 2003], or eiolca.net [Carnegie Mellon 2005].

With input-output LCA those sectors and therefore the potentially associated processes within the sectors, that are of significance to the overall LCA result and which should be included in the product system model, can be identified. Some claim that input-output LCA is much more than a screening tool, but rather a much more comprehensive method that has less problems with setting up the complete product system model than process oriented LCA according to [ISO 14040: 1997] (see e.g. [Lave et al. 1995; Hendrickson et al. 1998; Joshi 1999]). However, for applications in an industrial context, where product improvements and specific product and design alternatives are

[8] One might argue that impacts of energy generation processes are often very relevant in regards to the overall result of LCAs, because energy generation and consumption are two of the areas with the best data situation and therefore a high level of completeness compared to other processes such as materials processing or manufacturing activities. This could essentially lead to the argument of a self-fulfilling prophecy. Therefore, the correlation was determined based on national and international data related to overall emissions and their share on global man-made impacts.

in the focus, the direct application of input-output LCA is not possible, due to its limitations in regards to level of specificity, material and energy flows across national boundaries, etc. (see the discussion of [Heijungs and Suh 2002, p. 124]). With input-output LCA studies it is possible to show the sectors where impacts occur on a macro-scale, but not to identify improvement potentials for specific goods or services.

Complementing these systematic methodological options for screening, one should also mention the experience of the LCA practitioner as an invaluable asset for conducting screening procedures (predicting the key elements of the product system model based on knowledge, experience, and educated guesses). However, even if sufficient know-how and experience for a product group is available, it is hardly possible to predetermine all the important environmental issues without risking to neglect relevant hot-spots or trade-offs.

An additional issue is the fact that the appropriate screening methods may highly depend on the study objects, i.e. the reference flows, of an LCA. Based on experience and know-how regarding the product group or type, specific screening methods are more or less suitable for identifying areas of concern and improvement opportunities and to select those issues that need to be further studied quantitatively in the following steps. Examples are for instance products that require high amounts of fossil fuels for the production and use phases or products where toxicity effects might have the highest importance. For the former product group such as automobiles non-renewable primary energy demand can be suggested, while for the latter group such as pesticide products, a qualitative screening for toxic substances might be the better choice.

Often it is also advisable to combine the analysis of several screening indicators in order to avoid that important impacts are neglected in the LCA. Different indicators, may they be of qualitative or quantitative nature, can be combined and used within a matrix approach as outlined above, where the indicators are representing environmental stressors.

While the screening step is a very important component of the overall simplification procedure (see Figure 2-1), it is much more developed than the subsequent simplifying step, which is further elaborated in the following.

2.2.2.3 Simplifying

After screening, the goal is to simplify the model of the product system (see Figure 2-1), for which data have to be collected and compiled. As the area of simplifying is still in its infancy, there are no validated and accepted standard methods available. The U.S. Environmental Protection Agency (EPA) and the Research Triangle Institute (RTI) cooperated to examine various LCA simplification possibilities [Hunt et al. 1998]. Due to the aforementioned reasons (see Section 2.1), the analysis mainly looked at techniques that reduced the effort for the LCI by applying different cut-offs (i.e., deliberately excluding processes of the system from the inventory analysis). It was concluded that universal recommendations for 'horizontal cut-offs' (based on the image of a flow chart where the flows start with resource extraction at the top and end with the final disposal at the bottom), leading to the elimination of complete life cycle phases or major parts thereof, cannot be given. The success rate of the simplification by different horizontal cuts, expressed as delivering the same ranking as detailed LCAs, was found to be rather arbitrary and depending on the single application and reference flows (see Table 2-2).

Table 2-2: Analysis of LCA simplifying methods [Hunt et al. 1998]

Cut-off method	**Description (applied to packaging, industrial chemicals, household cleaners, etc.)**	**Success Rate (same ranking as detailed LCA)**
Removal of upstream components	All processes prior to primary material production (e.g. polymerization) are excluded	**58%**
Removal of partial upstream components	As above, but the one preceding step is included (e.g. monomer production)	**70%**
Removal of downstream components	All processes after primary material production are excluded (manufacturing, use, end-of-life)	**67%**
Removal of up- and downstream components	Only primary materials production is included (e.g. only polymerization)	**35%**

[Hunt et al. 1998] concluded that the application of 'vertical cut-offs' (in contrary to the horizontal cut-off defined above), whereby data are collected for all relevant stages and impacts, but in lesser detail (e.g. in regards to ancillaries), is generally preferable to eliminating major parts of life cycle phases at any given stage. This also implies that a screening, or pre-assessment, of the LCA is required prior to commencing a simplified inventory (and confirms the findings of [de Beaufort-Langeveld et al. 1997], see Section 2.2.2.2).

Horizontal cut-offs can be applied for comparative LCAs, if the alternatives have identical life cycle stages (type and quantity of material and energy processes and resulting elementary flows involved) [de Beaufort-Langeveld et al. 1997]. While this is quite obvious ('the low hanging fruit'), this strategy has to be mentioned as possibly the only one reliable simplifying method that can be always used, where applicable. An example for such a cut-off is the comparison of two passive automotive components with the same function and service life and identical weight, where the use phase can be neglected. In such cases, however, one has to be careful in the interpretation phase of LCA [ISO 14043: 2000], because the relative differences in impacts might seem much more relevant than those for the equivalent comparison that includes the identical parts. This has to be considered if the interpretation aims at decision-making in regards to the complete life cycle.

As already briefly discussed in Section 2.2.2.2, LCA approaches based on macro-economic input-output analysis (for the fundamentals of this methodology see [Leontief 1936]) have also been proposed as simpler alternatives for modeling complete product systems. Repeating and extending the arguments from the beginning of Section 2.2 and of Section 2.2.2.2, one has to state that input-output LCA is also not a solution for the simplifying step. For most products, input-output LCA is not sufficiently detailed and specified and not complete enough, since it is based on industry sectors and commodities (one commodity per industry sector) and restricted to industrial activities in single economies (therefore excluding imports and exports) [Heijungs and Suh 2002]. For instance, for Europe there is one sector (i.e. one unit process [Rebitzer et al. 2004a]) that encompasses all non-ferrous metals (see e.g. MIET according to [Suh and Huppes 2002]. As a consequence the calculated environmental impact per monetary unit of purchased non-ferrous metal material (e.g. primary aluminum, gold, platinum, magnesium, zinc, etc.) for a non-ferrous metal based product (e.g. a complete aluminum body-in-white structure for a passenger automobile or a piece of gold jewelry) are all identical, independent of the specific material and product. Even if the sectors are more specific (e.g. in the more detailed input-output tables of the U.S. economy primary aluminum is one specific sector [Carnegie Mellon 2005]), the variations of products produced by one sector are still immense. In addition, processes that only take place outside Europe or North America on a major scale (e.g. mining of precious metals) are completely neglected.

Other problems of input-output LCA concern the sometimes questionable economic and environmental base data, with the many pitfalls of tabulating economic flow data and especially environmental data (e.g. the toxic release inventory (TRI) in the US, which is often used for input-output LCA, does not account for emissions of small and medium sized enterprises (SMEs) [Suh et al. 2004]). Price inhomogeneity is also a very relevant cause of distortion in input-output LCA [Suh et al. 2004], since the impacts are linearly linked to prices, whereas price variations over time or in different regions are usually not connected to environmental causes. If they were linked to prices they might even point in the opposite direction: higher prices for products with improved environmental performance.

The deficiencies of input-output LCA described here are not conclusive or completely covered. [Suh et al. 2004], for instance, describe additional problems of input-output LCA. However, the issues elaborated above demonstrate that this approach cannot be used for simplifying the modeling of the product system in the context of industrial applications, where specific product life cycles have to be assessed. However, input-output LCA can be used for purposes of guiding the simplifying step, which is an idea proposed and elaborated in this thesis (see Section 2.5).

An interesting approach for completing product system models are hybrid LCA approaches as proposed by [Suh 2004a] and others, where input-output LCA data are used as proxies for parts of the product system that are unknown. Similarly, one should also mention the use of surrogates (usage of other, but similar data and models for missing elements of the product system model) as a very common approach to simplifying [de Beaufort-Langeveld et al. 1997]. Both of these approaches can be summarized as methods that use proxies for vertical cut-offs (see also the conclusions of [Hunt et al. 1998]).

The use of proxies for simplifying is a very relevant and useful approach in practice, specifically it is combined with a targeted sensitivity analysis (see Section 2.2.2.4). These are approaches, where the general strategy is to complete the product system model as in a detailed LCA, but to simplify in regards to the data requirements. Validity and ability to enhance the application efficiency of LCA then largely depend on the availability and specificity of proxies.

The research related to modeling without existing data and models in this thesis, however, targets more the methodology of product system modeling itself, focusing on the question 'Which processes to include in the product system model and which to neglect by introducing cut-offs?' (see Section 2.2.2.1). The application of hybrid and other approaches that utilize proxies is therefore not further elaborated.

Promising research aiming at simplifying the product system model by introducing systematic vertical cut-offs has been presented by [Raynolds et al. 2000a and 2000b]. Their Relative Mass-Energy-Economic (RMEE) method uses relative contributions of mass, energy content, and economic value to the process product of a unit process as criteria for cutting off input flows and thus the process chain(s) supplying this input flow. While this approach has been demonstrated for energy and combustion related air emissions, it is not generally sufficient to lead to correct conclusions [Lenzen 2001]. As a consequence, the applicability is limited.

One can conclude that the simplifying step for cases where no data and models are available as part of the overall simplification procedure (see Figure 2-1) is the most critical but least developed step when striving for the goal to model a product system with minimal effort but high validity. Therefore, one goal of the research presented in this thesis is to contribute to the development of new approaches and concepts for a more efficient modeling of the product system, in essence a way of simplifying.

2.2.2.4 Reliability Check

After one or more simplifying procedures have been selected and carried out, the reliability check is rather straightforward. From a methodological point there is no difference to conducting uncertainty, dominance, or sensitivity analyses, or to using expert judgments, analogy considerations, and plausibility checks, etc. in a detailed LCA.

Particularly important for simplification, however, is the condition that the reliability check should address both methodological choices as well as the quality of the data used [de Beaufort-Langeveld et al. 1997]. This is necessary in order to find those issues, which need refinement and for which fast feedback is required, leading to iterations and pointing to recalculations of the simplified LCA [Schmidt et al. 1995] (compare also Section 2.2.2.1).

Uncertainty and sensitivity analyses can range form simple procedures, based on intuition, to complex systematic methods. A systematic methodology for sensitivity analysis has been developed by [Heijungs 1996], which works on the basis of calculated or estimated uncertainty margins for all model input parameters. If the model has been implemented in modern LCA software, then more sophisticated reliability tests, also including stochastic models such as Monte Carlo analysis can easily be applied (see e.g. [Huijbregts et al. 2000; Maurice et al. 2000]). However, a lack or an insufficient application specifically of sensitivity analysis in LCAs can generally be observed, whether for detailed or for simplified LCA studies. This is an area, which clearly needs more attention and diligence.

For simplified LCAs it is important to stress that special care has to be taken in the interpretation phase. Though very small differences should not be over-interpreted in any LCA due to the involved uncertainties, the 'significant difference' should be larger in a simplified LCA. Some practitioners use a rule of thumb that differences of less than 20% for any given impact category, when comparing alternatives with the same functional unit, cannot be considered as significant. However, the exact range of differentiation depends on the method chosen, the data employed, and also largely on the goal and scope of the LCA. A general recommendation cannot be made [de Beaufort-Langeveld et al. 1997].

While the 'reliability check' is an important methodological element, the same procedures and methods as for any specific detailed LCA can be employed. Therefore no specific development in regards to simplification is required.

2.2.3 Conclusions and Resulting Methodological Focus of Research

From the previous Sections it is evident that the concrete modeling of the product system is an issue far from being scientifically solved, specifically from the perspective of an LCA practitioner in industry. The modeling of the product system and the involved data compilation and computation are still the probably highest barriers for the widespread and regular application of LCA. Therefore, methodological improvements for this critical step of LCA are necessary, both for cases where data and models are already available (see Section 2.2.1) and where the modeling has to start from scratch (see Section 2.2.2).

From the analysis of existing methodological attempts to facilitate the use and application of LCA, it is clear that there will never be a 'one fits all' approach, but that a toolbox of approaches is necessary, where the appropriate approach can be chosen based on the specific study object, pre-existing work, the goal and scope definition, etc. As part of this thesis four principle approaches to enhance the application efficiency of LCA, for different areas of applications, have been identified (see Section 2.3).

In order to give guidance on when to apply which of these approaches and when to employ a conventional LCA[9] method, the areas of application are differentiated by the following criteria:

- Availability of data and models
- Usage of product system models for one study only or additional uses for further analyses and other applications
- Level of influence on the complete life cycle of the product, i.e. the degree of influence that can be exerted by the decision maker on the product life cycle, e.g. regarding the selection of alternative physical or chemical processes and process chains
- Availability of reliable screening methods for the specific study object, i.e. knowledge on the key parameters and processes

These criteria and the resulting proposed approaches, however, are not conclusive, rather they are trying to offer new tools and methods for the aforementioned toolbox in order to enhance the application efficiency of LCA for industrial uses.

[9] The term 'conventional LCA' refers to the methodological procedure exactly as described in [ISO 14040: 1997; 14041: 1998; 14042: 2000], relating to the single steps as well as the order of steps. It is a detailed LCA that complies to the ISO standards without specific modifications or adaptations.

In the following Section a new framework for simplified product system modeling, based on the analysis of the state of the art and the proposed criteria for the area of application (see above) is presented. This is followed by an elaboration of the specific methods that have the goal to contribute to the improvement of the application efficiency of LCA (see Sections 2.4 and 2.5).

2.3 Framework for Simplified Product System Modeling

Figure 2-2 shows the developed systematic framework for simplified product system modeling, derived from the needs identified in the analysis of existing and proposed methods as elaborated in Section 2.2. The focus of the following Sections is on those elements, which are shown in hatched boxes in the figure. In these areas in particular, improved methods and procedures are necessary in order to facilitate the use of LCA for industrial applications. The differentiation of these methods, also in relation to product system modeling for a conventional LCA is based on the criteria described in Section 2.2.3, which are formulated as decision points in Figure 2-2. Additional explanations of the elements in Figure 2-2 are given in Table 2-3.

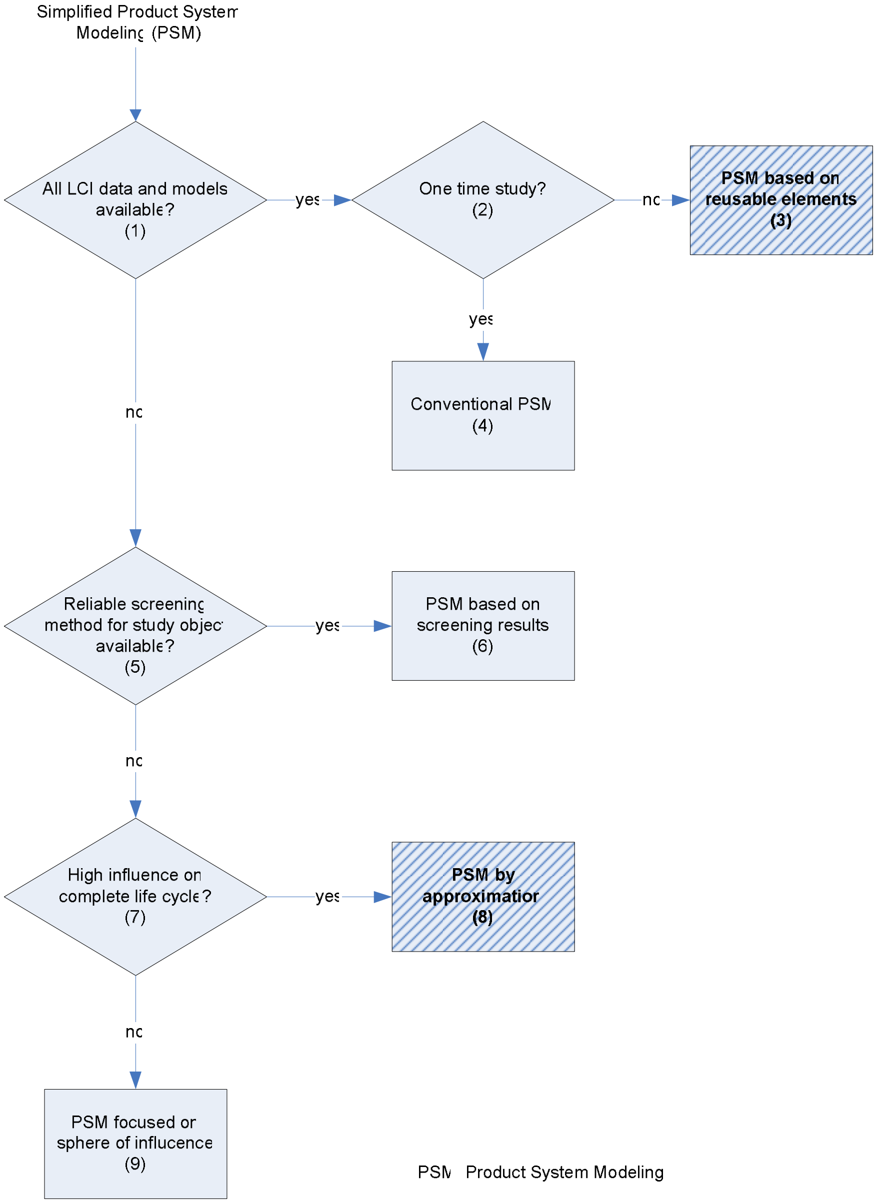

Figure 2-2: Framework for simplified product system modeling (numbers (1) to (10) refer to the explanations in Table 2-3); the hatched boxes refer to methods that are further elaborated in this thesis

Table 2-3: Components of the developed framework for simplified product system modeling (PSM)

No	Framework components	Description
(1)	All LCI data and models available?	This distinguishes studies where the relevant data and models are available from those where data have to be collected and computed from scratch (see the explanations in Section 2.2).
(2)	One time study?	Here, one has to decide if the data and models are only used for one study or if they or elements of these will be used for other studies and purposes in the future.
(3)	**PSM based on reusable elements**	The product system modeling based on reusable elements assumes that the data collection and modeling steps should lead to elements that are used again for other studies and calculations. The goal is to make this reuse as simple and effective as possible. **The 'Modular LCA' methodology elaborated in Section 2.4 is such an approach that makes the continued use of LCA data and models more simple and effective, also with its focus on decision-relevance and versatile usability within an organization.**
(4)	Conventional PSM (product system modeling)	If the collected data and the developed models of an LCA are not reused for other purposes or future LCAs, conventional modeling (see 'conventional LCA' in Section 2.2.3) is suggested, where the goal is to be as efficient as possible for this one study. This is usually done by using LCA software, tailor-made databases, or spreadsheet models.
(5)	Reliable screening method for study object available?	As discussed in Section 2.2.2.2 there are various screening methods available, which are more or less suitable for different study objects. E.g. if comparable LCAs of similar products are available, the key parts of the system can be pre-selected. Similarly, if it is known that the system is likely to be dominated by one parameter, such as energy consumption, the product system model can focus on the flows relating to energy generation.
(6)	PSM based on screening results	If there is a screening method for the study object available (see (5)), the model of the product system can be restricted to those elements which have been proven to be significant. Unit processes and the associated impacts whose influence has been proven to be negligible can be omitted.
(7)	High influence on complete life cycle?	This specifies whether the company or other organization commissioning the study, i.e. using it for decision-support, has a high influence on the life cycle, e.g. via material or complete product choices. There is no high influence if only parts of the complete system can be changed, e.g. due to the role of infrastructure constraints or generally limited influence on the system. The influence can be limited for instance if there is a constraint to use a specific technology due to recent investments. Essentially, it is a question if certain changes are options to be considered for the decision at hand, or if they are outside the scope.

No	Framework components	Description
(8)	**PSM by approximation**	If there is a high degree of freedom and therefore a high influence on the complete life cycle (see (8)), e.g. in choosing a product design (material, joining technologies, structure, etc.) or in selecting a supplier, it is important to cover all relevant elements of the product system. Then the question is what are the relevant elements in regards to environmental impacts. **An approach for the approximation of an unknown system and the resulting modeling consequences is developed in Section 2.5. This approach, coined 'Limiting system boundaries by baseline approximation' deals with the cut-off problem in LCA (see Section 2.5) and is, therefore, concerned with 'micro boundary' selection as defined below.**
(9)	PSM focused on sphere of influence	In cases, where the organization commissioning a study does not have influence on the complete life cycle, the product system modeling should focus on those parts, where activities of the organization can lead to changes in the system. It points to a much closer connection between the goal and scope definition of LCA and the modeling of the product system than commonly done in LCA. While this is an area where further research is needed, this is not the focus on this thesis. However, some initial research on this topic has been started (see below). Approaches focusing on the influence are concerned with 'macro boundary' selection as defined below.

The aim of this framework is to improve the usability of the LCA methodology for the given constraints and areas of application in order to complement the aforementioned toolbox (see Section 2.2.3). However, other approaches with equal or similar goals are also possible.

In the following Sections 2.4 and 2.5 the developed specific approaches for

- PSM based on reusable elements – 'Modular LCA' and
- PSM by approximation – 'Limiting system boundaries by baseline approximation' (both in bold in Table 2-3 and in hatched boxes in Figure 2-2)

are elaborated in detail.

Before going into detail, it is important to make a differentiation in regards to the nature of the system boundary problems addressed in 'PSM by approximation' and 'PSM focused on the sphere of influence' (see (9) in Table 2-3). While both deal with limiting system boundaries, they are targeting inherently different questions of system boundary selection, which are outlined in the following.

For 'PSM focused on the sphere of influence' the central question is *'What elements, i.e. parts of the product system usually consisting of a number of unit processes, can be* *excluded from the product system model without affecting the resulting recommendations for the decision to be supported?'* The focus here is on issues regarding the inclusion or exclusion of 'macro elements' of the product system model such as the use phase of a given product, specific transport activities, or the influence of infrastructure, etc. The associated process of system boundary selection and limitation, which is mainly carried out in the goal and scope definition, is thus defined as 'macro boundary selection'. This type of system boundary selection, while using a different terminology, is discussed for instance by [Trinius and Le Téno 1999]. The associated questions have triggered research with the involvement of the author (see [Rebitzer et al. 2003c; Braune et al. 2005]. This work, however, is only at the beginning.

For 'PSM by approximation' on the other hand, the essential question is *'Which single unit processes within a 'macro element' selected for inclusion in the product system model can be neglected without significantly influencing the final category indicator results?'* The focus here is on which process producing or using inputs and outputs connected to a single unit process to include based on relevance for the category indicator results of this macro element. The procedure of system boundary selection and limitation, which is mainly carried out in the life cycle inventory phase, often in a iterative way, is thus defined as 'micro boundary selection'. This type of system boundary selection is addressed e.g. by [Fleischer and Hake 2004; Lichtenvort 2004; Raynolds et al. 2000a and 2000b; Rebitzer and Fleischer 2000; Suh et al. 2004]

While these two different questions of selecting system boundaries are not clearly differentiated in the ISO standards[10] [ISO 14041: 1998] and [ISO/TR 14049: 2000] it is evident that they pose completely different methodological challenges, which have to be separately addressed.

Before focusing further on the methodological challenges and proposed solutions for system boundary selection and the resulting product system modeling procedures in Section 2.5, Section 2.4 concentrates on a new approach for making LCA more efficient for those cases where data and models are available.

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[10] On the other hand one could interpret the steps 'initial system boundaries' [ISO 14041: 1998, No. 5.3.3] and 'criteria for initial inclusion of inputs and outputs' [ISO 14041: 1991, No. 5.3.5] as corresponding to the 'macro and micro boundary selection', respectively. However, the methodological differences are not addressed and operational guidance is missing in the standards.

2.4 Modular LCA – A Decision-Oriented Approach for Product System Modeling Based on Reusable Elements

2.4.1 Introduction and Rationale for the Specific Developments

Product system modeling based on reusable elements (see Section 2.3) is an important strategy when the motivation for establishing an LCA model is not limited to producing results for one specific study, but rather when LCA is used as a management tool for continued and regular decision support. In such cases it is of immanent importance to have data and assessment models that are fast and easily accessible for decision-makers and supporting functions, also without the involvement of an LCA expert at all times. In addition, the method has to be targeted at decision-making processes. The latter means that the method should enable decision-makers responsible for different parts of the overall system to create a direct link between their influence and the occurrence of environmental impacts, related to both 'their part' and the overall system.

Such a method has been developed, as one part of this thesis, for the implementation of life cycle approaches within Alcan, one of the world's main suppliers of aluminum metals and products (semi-finished and finished) as well as composite components and packaging solutions. The main applications for Alcan's products can be found in the building, transport, engineering, electrical, and packaging sectors (for more information see www.alcan.com). While the method, coined 'Modular LCA'[11] (to be comprehensive the method should be called "Modular LCA based on foreground processes", however, in this thesis the abbreviated terminology is used), has been developed for the internal use at Alcan, the principles and methodological specifications are generic and can be transferred to uses with similar backgrounds and needs (see Section 2.4.2 independent of the industrial company or sector. Within Alcan, the methodological developments presented in this Section of the thesis were inspired and heavily supported by Kurt Buxmann, who is one of the industrial protagonists for making the LCA methodology more practical (see e.g., [Buxmann 2005]).

The setting of an LCA methodology as a management tool for decision making, from the perspective of an industrial company, is outlined in Figure 2-3 and Figure 2-4: Figure 2-3 shows a conventional approach, where the decision making framework focuses only on the value creation (economic and other benefits) and efforts (costs, risks) of the enterprise. Figure 2-4 on the other hand, shows how life cycle approaches

[11] While most life cycle inventories of LCAs are modular, the terminology "modular" refers to the fact that the extend of modularity is much higher than in a conventional LCA. In a Modular LCA the modularity does not only cover the unit processes, but also extensions of the processes and mandatory steps (see [ISO 14042: 2000]) of life cycle impact assessment). This fact and the differences to the conventional LCA approach are illustrated in Figure 2-5.

can be used to integrate the views of the external stakeholders (e.g. suppliers, customers, general society, NGOs) as well, leading to a more holistic based decision making framework and process. In the example of Figure 2-4 the Modular LCA approach is part of the step 'Detailed Assessment'.

Here, the example of product development is given, since this is one of the most important applications for LCA (compare Figure 3-1) and because product development is an area, where the efficiency of LCA is of utmost importance (see Section 2.1). However, the application of the Modular LCA methodology is not limited to product development, but is also usable for other purposes (see Section 2.4.2).

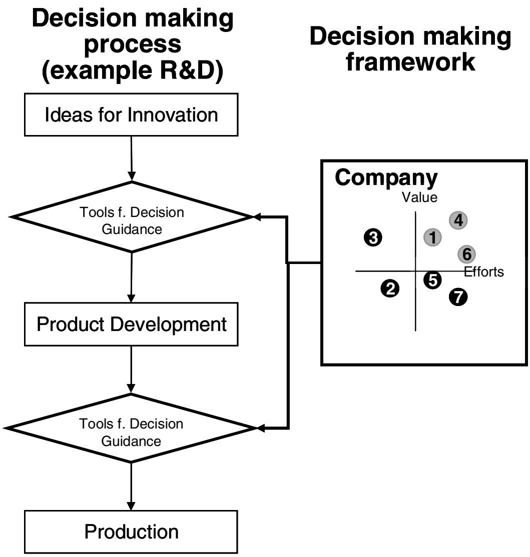

Figure 2-3: Conventional decision making process, for the example of product development (based on [Kistler 2004] and [Rebitzer et al. 2004d]). The dots and numbers represent different alternatives, whereas the origin of the plot represents a neutral position (in this example, options 1, 4, and 6 are preferable).

In the aforementioned decision-making context Modular LCAs can be used, besides other tools of life cycle management, for detailed assessments where comprehensive information is needed and which are continuously used and/or modified for further analyses of product modifications or products with equal/similar components, etc. Additionally, this approach for product system modeling (PSM) can be used for site-oriented environmental management (see Section 2.4.2.1).

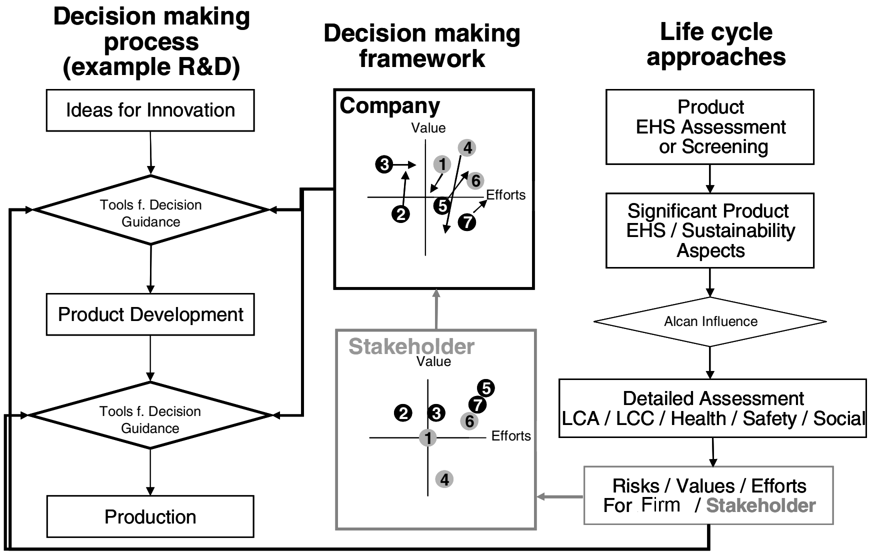

Figure 2-4: The use of life cycle approaches to include the perspective of all stakeholders into the decision making process, for the example of product development (based on [Kistler 2004] and [Rebitzer et al. 2004d])

It is evident that the support of product development and other decision making processes in companies needs quick and focused assessments. In addition, most of such decisions do not start from scratch, but from a given product, which is varied or for which new alternatives are to be explored and developed. Also, collected data and created models should be (re)usable for different applications as far as possible. These and other specific criteria and challenges of PSM in an industrial context are elaborated in the following Section.

2.4.2 Application Criteria for the New Methodology

In this Section the specific application criteria and associated challenges that formed the starting point for the development of the new methodology for product system modeling are derived and elaborated. These compliment the more general needs identified in Section 2.2.1.

For the use of LCA as a management tool as outlined in the preceding Section, two central questions arise [Rebitzer and Buxmann 2004a]:

- How to collect and update the necessary base data (raw data from company internal sites and external processes) with minimal effort?

- How to use and reuse collected data for different applications efficiently and rapidly while ensuring quality and validity?

To answer the first question for the example of Alcan, a central EHS (environment, health, and safety) database in which site-specific data of all facilities and process-specific data of the most important facilities are stored (inventory data per production/manufacturing process per year) provides the base data. The essential benefit of such a central database is that process data are stored only in one system, avoiding double inventories and inconsistencies when updating. Interfaces to several applications that can use the data compliment the system. For the implementation of LCA an interface to the Gabi 4 LCA software [IKP/PE 2004] exists that automatically transfers and converts the data to be used in the LCA software. The LCA software is subsequently employed to create 'independent modules', which are 'at the heart' of the Modular LCA methodology (see Section 2.4.3). A more detailed introduction to this data collection system is given by [Gabriel et al. 2003].

Independent from the specific solution of providing the base data, it is evident that the routine use of LCA requires a systematic data collection procedure. This can be done via sophisticated software systems or targeted data collection campaigns. The specific procedure is not relevant for the methodological developments in regards to the use and reuse of collected data and created product system models and sub-entities.

For the second aforementioned question ('How to use and reuse collected data for different applications efficiently and rapidly while ensuring quality and validity?'), the specific criteria that have to be met by a methodology in order to enhance the application efficiency of LCA in industry are listed in the following and described in Sections 2.4.2.1 to 2.4.2.4.

- Usability of models and data for both product oriented assessments (i.e. LCAs) and purposes of environmental management.

- Minimization of the effort and know-how needed for assembling system models based on reusable elements.

- Facilitation of the interpretation of the results at different levels (process, production site, supply chain, product, etc.) and for different applications.

- Suitability of the resulting indicators (LCIA indicator results and other indicators) to cover the relevant impacts and understandability to decision makers.

2.4.2.1 Usability of Models and Data for Both Product Oriented Assessments and Purposes of Environmental Management

The usability of any approach for product oriented environmental assessments, i.e. LCAs, is at the heart of LCA and therefore obvious. However, the direct use of LCA models and data for environmental management according to [ISO 14001: 2004] or EMAS [EC 2001] is an issue that needs further development. In the following the focus is on ISO 14001, though the issues are equivalent for EMAS.

Due to internal goals, supply chain pressure, strong interaction with other management systems (e.g., quality management), and demands from internal and external stakeholders, site-oriented environmental management and the subsequent certification to standards is more or less a 'must' in many sectors nowadays. Compared to this, LCM and specifically the consistent usage of LCA, on the other hand, is still in its infancy and adopted so far only by leading multinationals and some spearheading smaller companies, who have recognized the potential of this approach and who have the necessary resources available. Besides different scopes, the actors in site-oriented environmental management and LCM are different. While the former is dealt with by the management of one site, the latter involves many sites (internal or external to the company) within the value chain and addresses an overarching organization (such as a Business Unit or Business Group in the case of Alcan). [Rebitzer and Buxmann 2004b]

However, common goals and synergies of the approaches for both product LCAs and site-oriented environmental management from an application as well as a methodological point of view, and in regards to data provision, clearly exist and should be exploited, for reasons of efficiency and consistency. In addition, the new version of the ISO 14001 standard [ISO 14001: 2004] incorporates a broader, more systems oriented approach, also for the environmental management of sites. In the following, this broader approach for site-oriented environmental management and the required links to LCA and product system modeling based on reusable elements are discussed.

[ISO 14001: 2004] clarifies the requirements for environmental management systems in regards to which aspects shall be taken into account. Specifically, the standard states "The organization shall establish and maintain (a) procedure(s):

- to identify the environmental aspects of its activities, products and services within the defined scope of the environmental management system, that it can control and those which it can influence taking into account planned or new developments, or new or modified activities, products and services; and
- to determine those aspects that have or can have significant impact on the environment (i.e. significant environmental aspects)."

The identified environmental impacts to be managed by a site can be categorized as follows:

- Direct impacts: impacts generated by processes within the boundaries (fences) of the site, as usually reported to the authorities (compliance).
- Indirect impacts of operations: impacts generated by processes outside the site (up- and downstream), clearly linked to the operations of the site in the form of a cause-effect chain.
- Indirect impacts of goods and services[12]: impacts generated during the life cycle of the produced products of the site, as far as the site has control or influence over these.

With regards to the indirect environmental aspects, specifically

- environmental performance and practice of contractors, subcontractors, and suppliers,
- waste management, and
- extraction and distribution of raw materials and natural resources

are listed, among other issues.

Somewhat in contrary to this, traditionally, in most organizations throughout the world, ISO 14001, since its establishment in 1996 [ISO 14001: 1996], has been understood mainly as management of the direct environmental aspects (resource consumption, emissions, waste generated) of a site without a comprehensive assessment of the indirect aspects. Environmental emissions from the production of ancillaries or energy sources that are used within the industrial site, as well as impacts from provision of goods and services, have often been neglected. As the clarification of the ISO 14001 revision shows [ISO 14001: 2004], this practice has to be expanded in order to also address the indirect impacts of operations as well as the life cycle perspective.

Additionally, with the widespread establishment of ISO 14001 certification and the required management measures in addition to the related environmental improvements, the marginal costs of further improvements in regards to direct impacts increase. By more stringently involving up- and downstream processes in the priority setting of

[12] While [ISO 14001: 2004] uses the terminology ‘products and services” to denote material and immaterial utilities, [ISO 14040: 1997] uses ‘product’ as the general term, which also includes services besides (material) goods. In the following the terminology of [ISO 14040: 1997] is followed.

environmental goals and measures, a continued reduction of resource use, emissions, and waste can be facilitated, also inducing economic incentives.

Regarding the consideration of goods and services and indirect environmental aspects, all those should be taken into account which

- are of high relevance in relation to the overall activities of the sites and
- can be influenced by the site.

If products play a significant role and if the environmental impacts caused by them can be influenced by a site or a larger business entity encompassing several sites, may it be through research and development activities, manufacturing processes, or recycling efforts, etc. life cycle approaches can be directly applied as e.g. for product development (see Figure 2-4).

A suitable decision oriented application method of LCA should also enable to directly link the influence of a site or other actor in the life cycle of a product to the environmental impacts. With such a link it is possible to relate specific changes in the management of a site to actual environmental improvements. This is even more relevant since the points of influence and the occurrence of impacts are often not identical. Therefore, the comprehensive assessment of both direct and indirect impacts of operations as well as goods and services within environmental management requires elements of LCA in the form of specific models and data modules. In addition, such models should be compatible and/or equivalent to the corresponding unit processes used in LCA, so the same data and models can be used for site-oriented environmental management and product assessments. An approach that can be used for both product oriented assessments and for identifying and quantifying environmental aspects in site-oriented environmental management is essential for a better proliferation of life cycle approaches in industry.

The elaborations in this Section clearly point to the need for linking models and data of site oriented environmental management with LCA models. As a consequence, LCA models should be directly usable for ISO 14001 and similar management schemes, without additional modeling efforts. Additionally, in order to be decision-oriented and to enable the contribution to environmental improvements, the employed LCA models should be focused on those issues, which can be influenced by a site: the LCA elements used for environmental management should reflect those influences and the resulting changes in environmental impacts.

2.4.2.2 Minimization of the Effort and Know-How Needed for Assembling System Models Based on Reusable Elements

There is no shortage of sophisticated and powerful software tools and comprehensive databases for developing and reusing LCA data and models (see Section 2.2.1). However, these tools and databases are expert systems, which are not directly usable by non-LCA practitioners without specific training and knowledge. While such expert systems form the basis, there is a need for models that can be easily assembled, used, and modified also by non-LCA experts for specific and defined purposes (e.g. by an R&D engineer, who wants to compare different design options, or a sales manager, who wants to explore the environmental advantages and disadvantages of a given product compared to the competition). Only if such a simplified application can be achieved, will LCA be broadly used for the different applications within a company [Frankl and Rubik 2000, p. 235].

The task of the internal or external LCA experts should be to develop and provide simple tools and models as well as data in a suitable format, and to coordinate and elaborate the necessary updates and integration of new specifications, based on the needs and requirements of the different decision-makers or other LCA users. For the non-LCA expert, on the other hand, it should be possible to introduce simple modifications of the product system model without further support. Such modifications are usually 'what-if scenarios' (see [Weidema et al. 2004, p. 11]), where different options in well-known situations are compared. Examples are the variation of the amount of material needed for a given product, different assumptions in regards to the service life of a road vehicle (distance driven), or different end-of-life scenarios.

> For the 'end user' of LCA, the decision-maker or other non-LCA expert looking to use the results of LCA in a regular manner, it is crucial to have a simple system that delivers valid results. Such a system should allow for the variation of important parameters (technical specifications, basic assumptions, etc.) in order to identify their influence on the overall result. It should be possible to create a new product system model without using complex and specialized LCA modeling software.

2.4.2.3 Facilitation of the Interpretation of the Results at Different Levels and for Different Applications

LCA results are often complex and the challenge is to provide results in a form usable for different purposes (for different functions and applications within a company and for both environmental management and LCM, see Section 2.4.2.1). Different functions might focus on improvements of a single process, a process chain within a site, a complete production site, the supply chain, the downstream processes, the complete product life cycle, or only on end-of-life activities. Therefore, interpretation (according to [ISO 14043: 2000]), but also in a more general sense) should be possible not only for the assessment of the complete life cycle, but also for sub-systems such as unit processes or specific process-chains if the complete LCA is not (yet) available or when the influence of specific sub-systems of the product system is the main interest. Essentially, the interpretation of LCA results, including the identification of improvement potentials at different levels and for different applications, should be tailored to the decision to be supported. A wide proliferation of LCA applications within a company is only possible, if an easy interpretation of the results, linked to specific interest, is possible. Then LCA adds insight and value and is integrated into specific tasks (e.g. in product development, for purchasing, or for sales and marketing), in contrary to LCA being limited to reporting purposes on an expert level.

The possibility to interpret sub-systems (partial life cycle models), in the context of site-oriented environmental management (see 2.4.2.1) or for other purposes, is also an important success factor for the process of internal collection of raw data (see 2.4.2). If it is easy to interpret the results as outlined in the preceding paragraph and if the results are directly usable for a site, there is a high incentive and motivation for data collection. It is crucial to establish a link and to enable feedback between the data collection process and the use of the data. If such a link can be created and the corresponding awareness can be created, it is much easier to motivate the sites for data collection than in cases, where the collected data and the subsequent LCA result cannot serve directly for purposes of the data collector.

An LCA methodology to be applied in industrial practice should enable an easy interpretation at different levels and for different applications within the organization, without requiring additional complex analyses and expert systems. Additionally, it is desirable to have an interpretation mechanism that enables a feedback between data collection and LCA results, preferably in a way that the results are directly usable and decision-relevant for the data collector.

2.4.2.4 Suitability of the Resulting Indicators in Regards to Coverage of the Relevant Impacts and Understandability to decision makers

The use of complete and complex sets of life cycle indicator category results (as e.g. produced by advanced LCIA methods such as CML 2001 [Guinée et al. 2002] or IMPACT 2002+ [Jolliet et al. 2003]) is often not feasible when the indicators are to be used for rapid internal decision making. For most cases the interpretation of indicator results of such sets, specifically if there are trade-offs between different category indicator results, needs the involvement of an LCA expert to arrive at targeted decision support. The consideration of different levels of uncertainties of the category indicator results, which can be crucial, makes this interpretation even more complex and know-how intensive.

The solitary use of highly aggregated indicators on the other hand, such as fully aggregated Eco-indicator 99 scores [Goedkoop and Spriensma 2001], while eliminating the complexity of indicator sets, has other disadvantages. They are problematic specifically in regards to acceptance, transparency, and often also to validity (prohibitively high level of aggregation and simplification[13]), unless the validity for a given product or product group can be verified by using more disaggregated methods. Additionally, value choices that are inevitably involved when using weighting factors for the aggregation of indicators (see [ISO 14042: 2000]) can vary widely even within one organization, leading to additional acceptance und uncertainty problems [Schmidt and Sullivan 2002]. Therefore, for the efficient application of LCA in industry, it is desirable to use a relatively small set of indictors, which is easy to understand also by non-LCA experts, while limiting aggregation across impact categories and value choices within the method.

In addition, there are other potential impacts, which are not or only partially represented by current LCI data and models and the corresponding LCIA methods, but which can also be of high relevance. A prominent example are impacts caused by waste generation, where the inventory is often limited to system flows[14] (e.g. waste to landfill) and the resulting elementary flows are unknown or extremely uncertain. Therefore, the set of indicators should not be limited to category indicators in the strict sense, i.e. which are obtained on the basis of elementary flows, but should also include additional indicators, which are relevant, though difficult to grasp with current LCI models and LCIA methods.

Another need is to have a clearly defined set of indicators that can be used consistently and does not change from assessment to assessment, for reasons of comprehension, acceptance, and performance reporting. Regarding acceptance it is important that internal awareness building and training explains the indicators and the general background, so that decision-makers feel comfortable with the results.

[13] This is also the reason why methods with an extremely high level of aggregation are sometimes referred to as '42-methods' (see "The answer to life, the universe, and everything" [Adams 1979])
[14] System flows are input flows from or output flows to another process, i.e. those material, energy, or other flows that are not elementary flows.

It is a clear requirement to find an appropriate balance between validity/sophistication, transparency, necessary level of detail, consistency, and clear guidance, when selecting indicators to be used in LCA for industrial purposes. Specifically, for routine applications with pre-determined indicators it is important to limit the level of complexity, while avoiding extensive aggregation. Additionally, indicators that relate to impacts that cannot be fully or appropriately addressed by assessing elementary flows, should compliment the analysis, where necessary and appropriate. This ensures that the related impacts are not neglected.

2.4.3 Methodology of the Modular LCA Approach

2.4.3.1 Scope and Overview of the Approach

Existing base data, i.e. inventory data per unit process, forms the starting point (input) for the Modular LCA approach (see Section 2.4.2). There are no differences to a conventional LCA[15] methodology in regards to data collection, subsequent calculation of the inventory data of a unit process and all preceding steps, or the usage of existing data from databases, until the completion of the operational step "Relating data to unit process" according to [ISO 14041: 1998, No. 6.4.3] As a result, the approach does not require any specific inventory data, existing databases and models can be used without any adaptations.

There are some differences in the following steps, however, though it will be shown that the results of the Modular LCA are methodologically equivalent to those of a conventional LCA. Therefore, the Modular LCA is also in compliance with the ISO standards. Differences are limited to different techniques for modeling the product system and the specific application of the mandatory elements of the life cycle impact assessment phase according to [ISO 14042: 2000]. While most life cycle inventories of LCAs are modular, the terminology "modular" refers to the fact that the extend of modularity is much higher than in a conventional LCA. In a Modular LCA the modularity does not only cover the unit processes, but also extensions of the processes and mandatory steps of life cycle impact assessment. This fact and the differences to the conventional LCA approach are illustrated in Figure 2-5.

The differences to a conventional LCA procedure end with the completion of the mandatory elements of the life cycle impact assessment phase according to [ISO 14042: 2000, No. 4.3], thus with the calculation of the category indicator results for the

[15] The term 'conventional LCA' refers to the methodological procedure exactly as described in [ISO 14040: 1997; 14041: 1998; 14042: 2000], relating to the single steps as well as the order of steps. It is a detailed LCA that complies to the ISO standards without specific modifications or adaptations.

product system model. The Modular LCA does not address the optional elements of LCIA such as normalization, grouping, weighting, and data analysis (see [ISO 14042: 2000]). These techniques as well as any interpretation steps of [ISO 14043: 2000] can be employed for both a conventional as well as a Modular LCA approach.

An overview of the procedural steps of the Modular LCA, compared to those of the conventional LCA, is given in Figure 2-5.

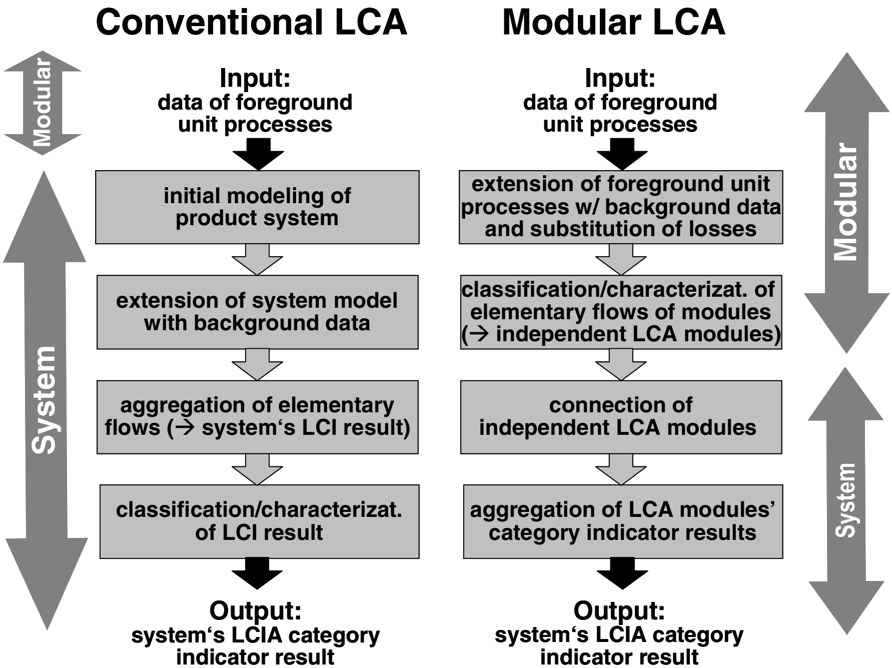

Figure 2-5: Comparison of the procedure for establishing product system models via a conventional LCA vs. the Modular LCA approach (modified from [Rebitzer and Buxmann 2004c])

In the following, the steps

- extension of foreground unit processes with background data and substitution of losses,
- classification and characterization of elementary flows of independent LCI modules,
- connection of independent LCA modules,

- aggregation of the LCA modules' category indicator results,

as well as the input (starting point) and output (final result) of the Modular LCA and the differences to the conventional LCA approach are developed and explained. Special emphasis will be given to the specific methodological aspects and product system modeling issues.

2.4.3.2 Input: Data of Foreground Unit Processes

Input to and starting point of the methodology are the collected data (data base) on the elementary flows of the 'foreground unit processes'. The foreground processes are those processes that are "related specifically to the product system at stake" while "background data are not specifically related to the product system and may consist of averages or ranges" [Udo de Haes et al. 1994, p. 11]. Others coin the foreground processes "main process chain" (e.g., [Fleischer and Hake 2002]) or "major unit processes" (e.g., [Guinée et al. 2002]) or use other similar terms. One can conclude that the foreground unit processes are those processes which are in the core of the analysis, for which data have to be analyzed, and whose interactions are of interest for the LCA. Often, many of these foreground processes are under the direct influence of the organization conducting the LCA, which is of high importance in regards to influencing environmental impacts and introducing improvements. Specifically, for industrial applications the internal production and recycling processes are usually foreground processes.

Therefore, the starting point of the Modular LCA is identical to a conventional LCA, i.e. existing unit process data can be employed for the Modular LCA approach. A modified or different data collection procedure is not necessary. Conventional LCAs also start with the foreground processes, based on which the complete system is established.

2.4.3.3 Extension of Foreground Unit Processes with Background Data and Substitution of Losses

In the first step, the data of the foreground processes are each extended to include a) the associated background data and b) the substitution of losses, the latter where relevant and appropriate. Specific methodological developments within this step of the Modular LCA are <u>underlined</u> in the following and subsequently explained in detail.

a) Generally, each foreground unit process is connected to background data such as life cycle inventories for generic energy generation, supply of commodities such as standard materials or ancillaries, or generic recycling and disposal processes. These data are usually obtained from publicly available or commercial data bases that are based on industry averages and generic models. Examples of such databases are given in Section 2.2.1. In contrary to the conventional approach in the Modular LCA this extension with background data is prepared per foreground process prior to the modeling of the initial or complete product system (1).

b) The extension to include the primary production needed to substitute losses of the foreground process (2) completes this steps. These substitutions represent the environmental impact related to the materials or energies that are not transformed into the desired output of the processes (losses [%]= 100% - yield [%]). As a consequence many processes can be modeled in a way that the value of the mass or energy flow that is linked to the preceding foreground process is identical to the mass or energy value of the succeeding process (3) (e.g., 1 kg output as reference flow and 1 kg linked input flow). Though there are exceptions such as processes where the main input materials are raw materials in the form of elementary flows (minerals or fossil resources in the ground) or the primary production that replaces losses is represented by the foreground process itself.

Results are extended unit processes, coined 'independent LCI modules', with defined input and output system flows (one of them being the reference flow, depending if it is a production or use, or an end-of-life process). These input and output system flows resemble the links to the other foreground processes, down- and upstream, respectively. After the extension all other system flows and the associated processes are part of the module. Remaining inputs and outputs of the module are elementary flows or other flows for which there are no data on receiving or producing unit process available (4).

In order to explain the specific methodological aspects related to this step of the Modular LCA the underlined issues of the preceding paragraphs are elaborated in the following.

1. Extension with background data per foreground process prior to the modeling of the initial or complete product system

The basis for the extension is the collected data per foreground unit process. This starting point prior to the extension is illustrated in Figure 2-6. The foreground process has inputs and outputs to the preceding and following processes for which data are collected as well as to background processes that deliver the ancillaries and energy needed to run the process or take up outputs that are not the process product(s) (emissions to be treated, wastes for recycling and disposal). Additionally, the elementary flows that are direct input to or output from the foreground process are recorded.

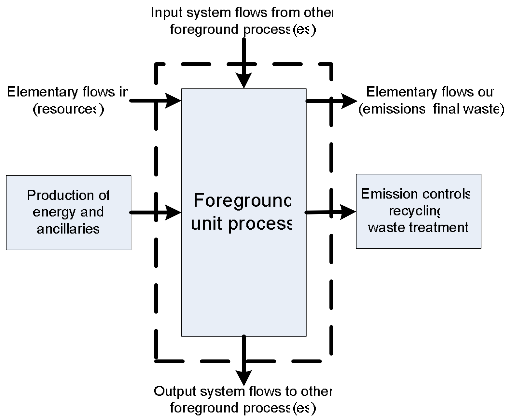

Figure 2-6: Collected data for foreground unit processes (starting point)

In a first step the foreground unit process is extended by integrating the associated background processes. Thus, the boundaries of the extended process now integrate the associated system flows and (ideally) only the elementary flows of the background processes leave the extended process. This extension, which can be interpreted as a gate-to-gate aggregation, is illustrated in Figure 2-7. It is important to note that this is performed per single unit foreground process without first connecting the processes to model a full cradle-to-gate system. At this step they are kept separate.

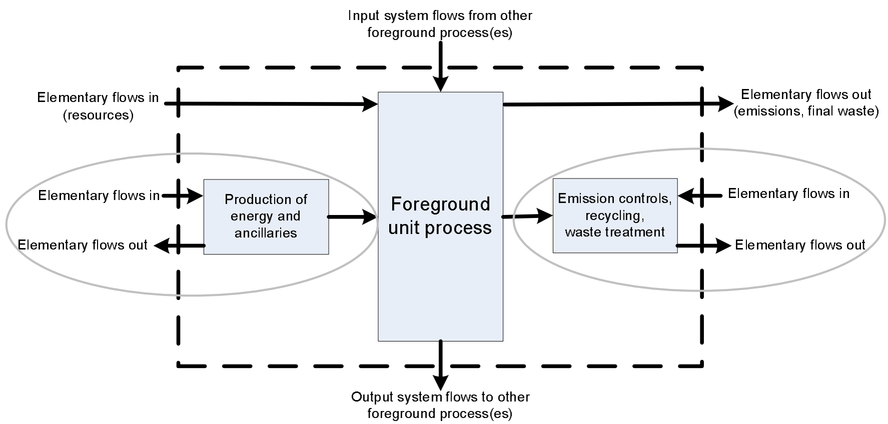

Figure 2-7: Foreground unit process extended by background processes (the ellipses mark the integrated background processes that are added as part of the extension)

2. Extension to include the primary production needed to substitute losses of the foreground process

In addition to the integration of the background processes, the substitutions of losses are also accounted for. These losses refer to losses of the input flows from the preceding foreground process(es), i.e. material flows or energy system flows that are not transferred into the output system flow(s) leading to the next foreground process. These losses, caused by the inefficiency of the foreground process (losses = 100% - yield [%]), are accounted for by a substitution model, where the losses are replaced by the equivalent primary production, which is in turn integrated into the extended process (see Figure 2-8). The result is the LCI module in idealistic form, where only the connecting flows to the other foreground processes and elementary flows cross the system boundary.

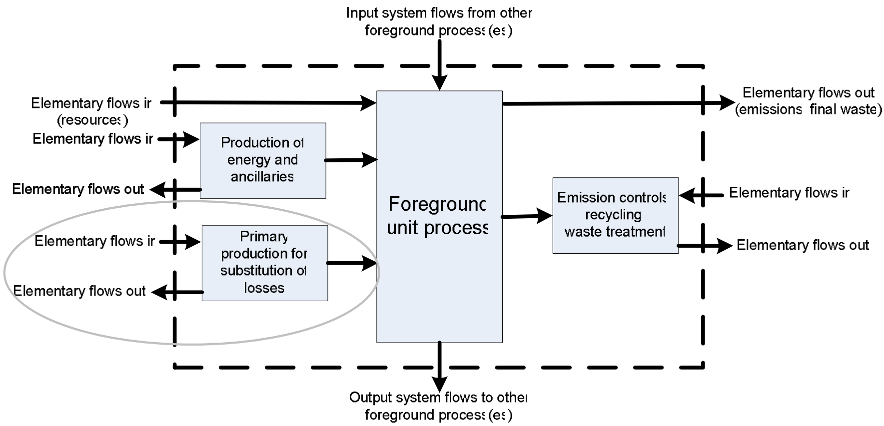

Figure 2-8: Complete LCI module in idealistic form (all flows crossing the system boundaries are either connections to other foreground processes or elementary flows; the ellipses mark the additional extension)

This substitution of losses can best be explained by using a hypothetical example: assuming that the analyzed foreground process is an aluminum extrusion process with a reference output system flow of 1 Mg extrusion profile and a preceding process 'aluminum ingot casting from primary aluminum'. It is assumed that the production of this 1 Mg extrusion profile also leads to 200 kg of scrap (unused material input). Due to the inclusion of the background processes for recycling, the recycling of this scrap is included in the LCI module. If this recycling step (remelting of the scrap to produce new ingots) would have an efficiency of 100%, then there would be no losses and the recycling can be seen as closed loop within the LCI module. However, the efficiency is never 100%; in this specific case it might be 98%. An efficiency of 98% for the remelting of the 200 kg scrap means that 4 kg of aluminum material are lost due to the operations within the LCI module. This loss is then allocated by integrating the primary production of an aluminum ingot of 4 kg.

Methodologically this integrated substitution of the losses assigns the impacts of the losses to that process that causes the impacts, which often varies from the processes where this occurs, i.e. in the upstream chain. Any losses of the input system flow that is supposed to be transformed into the output system flow is accounted for by this method of substitution. This is especially useful for process optimization, showing directly the improvement in process efficiency within one module. The goal of the independent LCI module is to integrate as many of such losses as possible, so that all relevant impacts that are caused by a foreground processes can be identified and allocated to that process.

This principle can be applied not only to material transformation processes, but also to processes where both input and output system flows are given in energy units. In cases where the primary production that replaces losses is represented by the foreground process itself or for the main flows in energy conversion processes this substitution approach is not applicable, but also not necessary. In these cases the efficiency and related losses are already represented by the resulting elementary flows of the LCI module without substitution. Special cases are also processes that do not have a preceding foreground process (resource extraction processes) or that do not have an outgoing system flow to a following foreground process (final disposal processes). Also here, the integration of the substitution of losses in not necessary. The losses are addressed by the efficiency of the process itself.

3. **Many processes can be modeled in a way that the value of the mass or energy flow that is linked to the preceding foreground process is identical to the mass or energy value of the succeeding process**

Staying with the example of producing 1 Mg extrusion profile (see above) it is logical that – after the substitution of losses has been integrated - the input system flow from the preceding foreground process step of aluminum ingot is also 1 Mg. Therefore, if all the substituted losses are integrated in the system of Figure 2-8, the different yields of the foreground processes are already accounted for and do not have to be taken into account later when the complete system is built by combining the foreground processes.

However, if there is more than one preceding foreground process (e.g. the use of two or more main materials, e.g. for the production of a composite material that consists of 30% aluminum and 70% polyethylene), then this modeling principle cannot be used, since the total input mass is split into different preceding foreground processes. In this case, the specific connection of LCI modules has to explicitly consider the contribution of the different LCI modules. This is also the case for those special situations where the substitution of losses cannot be integrated (see above).

Other special cases are processes that produce more than one product, where co-product allocation has to be employed, and situations of open loop recycling. In these cases the used allocation procedures influence also the preceding processes, which has to be considered in the contributions of each preceding process. Therfore, correction factors for all upstream processes have to be introduced.

While the modeling of the complete product system by combining the independent modules is simplified if the efficiencies of the processes do not have to be considered, the consideration of the efficiencies does not pose a major problem either. The only point to consider is that this should not be overseen, which can be ensured by marking those special cases. Similarly, processes with co-product allocation or open loop recycling also have to be marked, so that the correction factors for the upstream processes can be introduced when combining the modules (see Section 2.4.3.5).

4. Other flows for which there are no data on receiving or producing unit process available

As previously mentioned and visualized in Figure 2-8, ideally all flows apart from the input and output system flows that are connected to other foreground processes are elementary flows. This, however, is not always possible, due to a lack of the necessary process data or lack of knowledge of the preceding or following processes. While such cases should be minimized as far as possible by integrating the relevant background processes or collecting data thereof, they cannot be completely avoided. In such cases and in order to not neglect system flows to and from background processes where the receiving or delivering process is either not known or uncertain or the corresponding input and outputs are not known, additional non-elementary flows are introduced. Such, flows are on the output side mainly waste flows before or after treatment. On the input side these are flows such as water for which often no process and site-specific data in regards to production (provision of water) is available. Instead of neglecting these non-elementary flows via cut-offs, as often done in practice, they are separately accounted for in the Modular LCA approach and aggregated to some extend in the form of indicators (see below).

In this context waste is a specific issue and particularly the landfilling of waste with or without prior treatment is a disposal process where sufficient data is lacking. While there are databases attempting to also completely model such disposal processes (e.g. ecoinvent 2000 [Frischknecht et al. 2004a]) the methodological challenges, specifically for long-term emissions from landfills are far from being solved (see [Hellweg and Frischknecht 2004]). The lack of data and high uncertainties for modeling landfilling is also acknowledged by the developers of the aforementioned database (see [Doka 2003]). Therefore, in addition to the elementary flows, water as resource as well as wastes as outputs that have not been modeled 'to the end' are recorded and termed 'technical non-elementary flows' (see Figure 2-9)

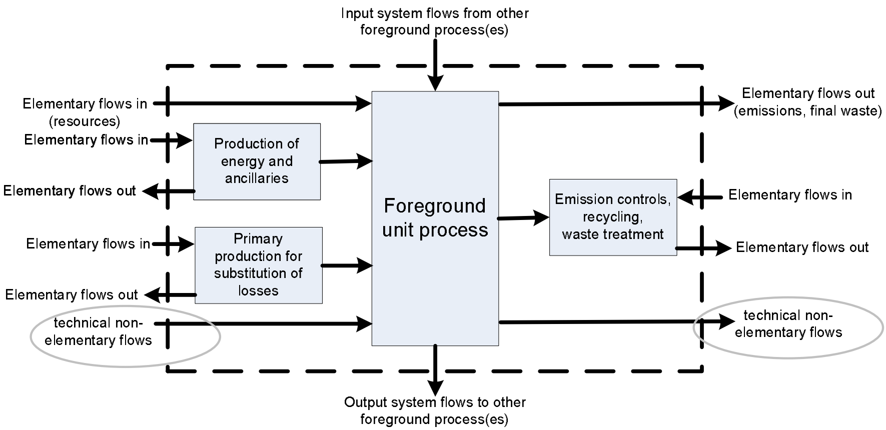

Figure 2-9: Complete LCI module with the inclusion of technical non-elementary flows crossing the system boundaries (the ellipses mark the additional inclusion of these flows, which are neither system nor elementary flows)

2.4.3.4 Classification and Characterization of Elementary Flows of Independent LCI Modules

After the completion of the system expansion of the foreground unit processes, the impact assessment is the next step. The specific methodological developments within this step of the Modular LCA are <u>underlined</u> in the following and subsequently explained in detail.

In the phase of life cycle impact assessment the Modular LCA utilizes a set of <u>pre-determined indicators (1)</u>, consisting of LCIA category indicator results and indicators based on technical non-elementary flows that have shown to be valid for the considered applications. This pre-determined set reflects the needs in regards to decision-support, while also acknowledging the limitations of existing impact assessment methods. Additionally and in contrary to the procedures described in ISO [14041: 1998] and [ISO 14042: 2000], the classification and characterization steps in the Modular LCA approach are carried out not for the LCI result of the complete product system model, but for the LCI result of the independent LCI modules. This results in the provision of single LCIA category indicator results per independent LCI module (2). The assessed modules are termed 'independent LCA modules'.

1. Pre-determined indicators

In any LCA the goal and scope definition pre-determines the life cycle impact assessment method(s) to be used in the study [ISO 14040: 1997, 5.1.2]. Thus, also the impact categories, category indicators, and specific characterization methods are defined early on. The Modular LCA approach also adheres to this procedure, though the choice of impact assessment methodologies is not done on a case by case (for a specific LCA), but on a general basis. This is necessary to allow the rapid company-internal reuse of LCA modules from different studies and to combine them for new LCAs. In addition, non-elementary flows, which have not been modeled 'to the end' (see 4. in Section 2.4.3.3), due to missing data and/or lack of knowledge, are also aggregated to indicators. The latter indicators are positioned between the life cycle inventory results and mid-point indicators of LCIA. While one could argue that these indicators are insufficient, it is clear that it is better to include them in the impact assessment rather than to completely neglect the associated impacts.

In order to address both types of the aforementioned indicators this thesis distinguishes between life cycle impact assessment category indicators according to ISO 14042: 2000 and 'technical life cycle indicators', the latter building on the technical non-elementary flows (see 4. in Section 2.4.3.3). Both types are summarized under the term life cycle indicators.

Which set of pre-determined (i.e. generically selected) life cycle indicators to use for the purposes of an industrial company cannot be answered generally, this depends on the sector and specific requirements. The following elaborations focus on the set of indicators that have been selected and developed as part of this thesis for the internal use at Alcan.

At Alcan, for internal studies, five life cycle indicators have been selected as the major indicators for the Modular LCA approach:

- Greenhouse gas potential (100 years) of the emissions caused by the processes involved; in kg CO_2 equivalents

- Non-renewable primary energy demand (also coined non-renewable cumulative energy demand [VDI 4600: 1997], i.e. the energy content (lower heating value) of fossil and nuclear energy resources which are extracted from the earth; in MJ

- 'Eco-indicator 99 minus' (previously 'Eco-indicator 95 minus'), an indicator based on the Eco-indicator methodologies ([Goedkoop and Spriensma 2001] and [Goedkoop 1995], respectively), where the contributions of the other single indicators greenhouse gas potential and primary energy demand covered in the specific set are subtracted; in Eco-indicator points

- Waste generation, which is a life cycle indicator based on technical non-elementary flows and expresses all remaining (i.e. not modeled 'to the end') waste flows; in kg municipal solid waste equivalents

- Water consumption, i.e. the total quantity of water which enters the system; in kg

Greenhouse gas potential (sometimes expressed as 'climate change') has been chosen, because it can be seen as a central indicator, with a high global acceptance and representing one of the most relevant environmental pressures (see e.g., [IPCC 2001]). This category impact assessment indicator is a default midpoint indicator used in literally all studies, methods, and recommendations (see e.g. [Goedkoop and Spriensma 2001; Guinée et al. 2002; Jolliet et al. 2003; Wenzel et al. 1997]).

Non-renewable primary energy is not an life cycle impact assessment indicator in the strict sense (see also the discussion of [Jungbluth and Frischknecht 2004]), however it summarizes impacts related to the consumption of non-renewable energy resources required for virtually all product systems and thus covers a broad range and magnitude of impacts (compare Section 2.2.2.2). This is also the reason, why this indicator has found widespread application as a screening indictor for LCA (see Section 2.2.2.2) and has a high acceptance at all stakeholder groups.

An advantage of using greenhouse gas potential and non-renewable primary energy is also the relatively low uncertainty related to these indicators, in relation to the inventory data required and the characterization or aggregation factors, respectively. Additionally, data related to greenhouse gas emissions and energy consumption are relatively easily available or can be modeled for a process without high efforts. Therefore, related background data are often also the most complete in regards to coverage of processes and elementary flows. These characteristics are clear advantages compared to other, less established indicators, which depend on many and often uncertain assumptions and where results of different studies are often problematic to compare.

In order to generally cover additional impacts that are not properly addressed or represented by greenhouse gas potential or non-renewable primary energy consumption (as e.g. impacts of toxic substances not originating from energy generation), the Eco-indicator 99 [Goedkoop and Spriensma 2001] is used as an aggregated end-point indicator. However, in order to avoid double-counting of identical impacts, the contributions of global warming potential and non-renewable energy resources are subtracted, since these are separately covered. The result is the 'Eco-indicator 99 minus' (due to technical reasons, the older Eco-indicator 95 [Goedkoop 1995] was used as a basis previously, as e.g. in some examples shown in Section 2.4.4).

The inclusion of this indicator ensures that non-energy and non-greenhouse gas related impacts are not neglected, while placing a lower emphasis on category indicator results with high uncertainties (for elaborations on uncertainties e.g. of toxicity indicator results see [Huijbregts et al. 2000] and [UNEP/SETAC 2004]). However, one has to note that this is clearly a matter of choice. If other users of the Modular LCA want to put more emphasis on other indicators that are part of Eco-indicator 99 or other advanced methods such as e.g. IMPACT 2002+ [Jolliet et al. 2003] the method can be tailored to those needs by selecting a different set of pre-determined category indicators.

As already mentioned, the remaining technical life cycle indicators 'waste generation' and 'water consumption' are based on technical non-elementary flows, with some water input flows directly from nature being the exception. These indicators have a special position, since they are - from their concept - positioned between LCI results and the conventional midpoint category indicators (LCI/LCIA indicators). They are, in principle, simplified a-priori aggregations of midpoint indicators and their corresponding damage categories (for a discussion on midpoint indicators, damage categories and their interrelations see [Jolliet et al. 2004]).

For the case of waste one can say that the waste indicator aims at representing aggregated LCIA results for specific end-of-life treatment process chains via simplified midpoint LCI/LCIA indicators. Since there are no established and internationally recognized indicators for waste available (yet), a first system for the Modular LCA approach was developed for the internal use at Alcan. This is based on 'kg municipal solid waste' as category indicator and relates the potential impacts related to other waste categories by using the characterization factors listed in Table 2-4.

Table 2-4: Characterization factors for different waste categories

Waste category	**Characterization factor**
Mining residues	0.05
Residues from ore refining	0.5
Hazardous waste	5.0
Non-hazardous waste (municipal solid waste)	1.0

The characterization factors are based on internal expert judgment, since reliable data for the associated impacts do not (yet) exist. Since waste indicators comprise a very new and not yet established area, further research should start with a comprehensive review and structuring of existing LCAs comparing waste management strategies and technologies. Comparing and analyzing existing studies (as e.g. done by [Björklund and Finnveden 2005] and [ADEME 2002]) is a first step in the direction of default characterization factors for waste.

While this could and should be refined into further categories that also take into account the route of disposal (direct landfill, chemical-physical treatment, incineration, etc. for hazardous and non-hazardous waste) in the future, for practical reasons the very simple set of Table 2-4 is chosen as a starting point. While this is a very simplified and not yet validated assessment scheme, it is an improvement over the often existing situation, where such waste flows are not considered in the impact assessment phase at all, essentially completely neglecting the associated impacts.

For water consumption the case is similar to that of waste. Here, all water used in the product system is aggregated, while only the water directly extracted from nature (from a well, river, lake, ocean, etc.) is an elementary-flow. Other water input from public or commercial systems of water provision, including purification, different treatment steps, transport, etc. can often only be accounted for by the water flow itself, for practical reasons. The involved background processes are often not available, in particular for water as used for industrial applications. Additionally, the consumption of water as a resource is not accounted for in the established LCIA methods, there are no characterization factors for water available. Therefore, the computation and reporting of this indicator adds additional information. Specifically, if it is related to a certain foreground process, where, and if appropriate, the geographical influence can be estimated (e.g. different impacts of fresh water consumption in Iceland compared to those of the same consumption in Saudi Arabia).

Since there are no LCIA methods for covering water as a resource available, in the scope of the Modular LCA approach at Alcan water consumption is only aggregated on a per kg basis in the impact assessment phase, which means that there is no differentiation between the types of water listed in the inventory (ground water, sea water, fresh water, etc). The contribution of water consumption as an impact on the input side clearly also needs additional research activities. It would be desirable to have a method that not only takes into account the type of water, but also the geographical region (as e.g. discussed by [Bauer 2003]).

One can conclude that the chosen set of pre-determined indicators for the Modular LCA covers the key impacts, while avoiding both a very complex set of indicators and an overemphasis on impacts with high uncertainties. This selection of indicators for environmental aspects can be seen similar to the process employed in the balanced scorecard concept, though the latter tries to cover even a broader range of impacts and values (see e.g. the sustainability related discussions of [Figge et al. 2002]).

2. Provision of Single LCIA Category Indicator Results per Independent LCI Module

After the extension of the processes (see Section 2.4.3.3), the elementary flows of the resulting LCI module are classified and characterized according to the mandatory steps of [ISO 14042: 2000, No. 4.3] and using the set of pre-determined indicators explained above. In a later step (see 2.4.3.5) these category indicator results of the module, relating to the input and output system flows, are combined in order to compute the category indicator results of the complete product system model.

In the Modular LCA methodology the impact assessment is conducted prior to the modeling of the complete system. While this step reverses the procedure described in [ISO 14041: 1998] and [ISO 14042: 2000], where the impact assessment is conducted for the complete life cycle inventory of the whole system, it can be easily shown that both procedures are methodologically fully equivalent:

Equation 2-1 shows the conventional calculation procedure for an impact category A, where first the LCI result of the complete product system model is calculated and afterwards the impact assessment result is obtained via multiplication of the aggregated elementary flows with the respective characterization factors. Equation 2-2 shows the calculation procedure of the Modular LCA, where the indicator results of the impact category A of the (extended foreground[16]) unit processes are summed up in order to receive the category indicator results of the complete system.

$$CIR_{AC} = \sum_{k} \left(\sum_{i} ef_{k,i} \bullet CF_{k} \right) \qquad \text{Equation 2-1}$$

$$CIR_{AM} = \sum_{i} \left(\sum_{k} ef_{k,i} \bullet CF_{k} \right) \qquad \text{Equation 2-2}$$

CIR_{AC} : Category indicator result of impact category A (e.g. global warming), Conventional calculation as described in [ISO 14041: 1998] and [ISO 14042: 2000]

[16] In this specific Modular LCA methodology the impact assessment is performed based on the LCI modules, which resemble the extended foreground unit processes. The reversal of the steps aggregation of the life cycle inventory and impact assessment, however, is also possible for other process entities (e.g. without extension). Therefore, the equations represent the generic case.

CIR_{AM} : Category indicator result of impact category A, calculated with the Modular LCA approach

CF_k : Characterization factor for elementary flow k:

ef_k Elementary flow k

i: Foreground unit process

Equation 2-1 can be easily transformed into Equation 2-2, via the basic mathematical distributive law (see Equation 3). This leads to the conclusion expressed in Equation 2-4.

$$a \bullet b + a \bullet c = a \bullet (b + c) \qquad \text{Equation 2-3}$$

$$CIR_{AM} = CIR_{AC} \qquad \text{Equation 2-4}$$

This is proof that the aggregation of life cycle category indicator results as carried out in the Modular LCA approach complies to the ISO standards for life cycle assessments, ensuring validity, acceptance, compatibility and comparability with other methods.

As an additional benefit for the practical application, this step, together with the extension of the foreground unit processes, greatly facilitates the interpretation phase (see [ISO 14043: 2000]) since the impacts of the examined foreground processes and the related background and substitution processes (together the "processes related to the product system at stake", see Section 2.4.3.1, I.) are directly available for interpretation. A specific des-aggregation of the results in order to find the most important processes is not necessary any more.

It should be noted, however, that it is also important to retain the original information from the life cycle inventory 'below' the aggregated result level, for purposes of error identification, updating of data, and for applications or analyses where a higher level of detail is necessary. This requirement can be easily fulfilled with today's LCA software packages.

2.4.3.5 Connection of Independent LCA Modules

The model of the complete product system is obtained by connecting and combining the independent LCA modules to include all relevant processes and life cycle phases related to the functional unit analyzed. This modeling step is similar to the normal modeling of the product system in the life cycle inventory analysis phase. However, in this approach it is not the elementary flows that are aggregated, but the life cycle indicator results (see Section 2.4.3.4, 2.). Since this has to consider only the pre-determined set of indicators (see Section 2.4.3.4, 1.) and not a huge array of material and energy flows and other interventions, this combination is simpler than in a conventional LCA, though leading to identical results. In addition, for many processes the combination via the reference flows of the independent LCA modules (defined output or input system flows, see Section 2.4.3.3) does not need to consider process yields, if the substitution of the losses is fully integrated into the independent LCI module (see Section 2.4.3.3, 3.).

In order to create models that are easily usable also by non-LCA experts and without specialized software, the combination is implemented in the form of spreadsheet models, which contain the category indicator results of the required LCA modules and the interconnections. Parameters that should be variable due to their relevance in regards to assumptions, 'what-if' scenarios (see [Weidema et al. 2004, p. 11]), decision-relevance, or that relate to different options (e.g. comparisons to competing products) are introduced in a flexible way. This ensures that the influence of these parameters on the overall result of assessing one single product or for internal comparisons between products and alternative life cycles can be easily analyzed. Simple variations of the product can be assessed without complex changes to the model.

Of specific importance for industrial applications, e.g. for the product development process, are the variable parameters in regards to decision-relevance. Decision-relevant are those parameters which can be influenced by the industrial actor (compare Figure 2-4). For product development, for instance, these might be technical parameters of the product design that can be varied. This means that the LCA expert setting up the system has to identify the specific requirements of the user(s) of the model, who will later work with the system. The latter is essential, so that the model is specifically targeted at the user and can give direct decision support, without the need to involve the LCA expert at all times. The LCA expert is only required if more severe changes to the model, such as the establishment of new parameters, the adding of new independent LCA-modules, or other general modifications are necessary. This reduces the effort for the internal LCA expert to providing the tools, associated training, and support for very specific or complex questions and interpretation needs.

The aforementioned procedure for modeling the complete product system model also enables straightforward dominance and sensitivity analyses in regards to the variable parameters, which is important for finding key environmental aspects and assumptions that in turn can be used again as guiding factors for product design, process improvements, etc.

2.4.3.6 Aggregation of LCA Modules' Category Indicator Results

Once the complete product system model is set up (see Section 2.4.3.6), based on the life cycle indicator results of the single LCA modules, (see Section 2.4.3.4) the indicators are aggregated for the complete product system model. If the computation is done via a spreadsheet software (see Section 2.4.3.5) default graphical representations of the aggregated results can be easily introduced in the model, which greatly facilitates the direct use of the results for the decision-maker/user of the model. This is particularly useful, if there is a standard reporting format of such results, because an additional visualization of the results is then not necessary and a common basis for the understanding of the results can be created within the organization, similar to standardized and well known charts used to report for example financial or safety performances.

2.4.3.7 Output: System's LCIA Category Indicator Results

The final output of the Modular LCA, the category indicator results (system's LCIA result) is equivalent to the one of the conventional approach, where the system's LCIA result is calculated from the classification and characterization of the LCI results of the complete product system model. The subsequent steps of the LCA methodology are consequently outside the scope of the Modular LCA approach, i.e. identical to the conventional approach.

There is an additional useful feature of the Modular approach in regards to the following interpretation phase according to [ISO 14043: 2000]: The extensive modular nature highly facilitates the identification and analysis of the "essential contributions" to the LCIA results from different foreground unit processes, as required by [ISO 14043: 2000, No. 5.3]. The category indicator results of single parts of the system are already available and do not have to be extracted from the final result via a separate analysis. The use of model parameters and the ability to efficiently conduct sensitivity and dominance analyses (see Section 2.4.3.5) further supports the interpretation and use of the results for decision-making as illustrated in Figure 2-4.

2.4.4 Conclusions

One can conclude that the main difference of the procedure of the Modular LCA compared to the conventional one is characterized by the fact that the impact assessment is performed per extended foreground unit process, before the modeling of the complete product system. As a result the modular characteristics are not limited to the inventory data of the single unit processes as in a conventional approach, but carried further. This further reach of the modular characteristics and thus the creation of reusable elements that integrate more elements of the complete LCA methodology can be seen as a clear advantage in regards to an efficient application of LCA. While the application efficiency is enhanced, the required granularity for decision making that aims at the foreground processes, is retained. This is a clear distinction to the use of fully aggregated cradle-to-gate data, where a later distinction between the different foreground processes in not possible anymore.

When checking the methodological features elaborated in Section 2.4.3 against the criteria derived in Section 2.4.2.3 one finds that all the postulated criteria can be met by the Modular LCA methodology. With this approach, synergies of using independent LCA modules for both product-oriented as well as site-oriented environmental management can be exploited, which is particularly interesting due to the widespread implementation of site oriented systems (see Section 1.2). Additionally, the effort is minimized and simple models also to be used by the non-LCA expert can be easily assembled, facilitating the regular use of LCA calculations and results. In addition, the compact set of pre-determined indicators can be easily communicated to different functions and decision-makers within a company. Other possible uses not mentioned in this Section before concern the employment of independent LCI modules for Environmental Product Declarations (EPDs), since these resemble the so-called information modules needed for an EPD (see [ISO/DIS 14025: 2005]). This means that also customer inquiries in regards to the environmental performance of products or components can be answered based on the approach, without additional efforts.

2.5 Limiting System Boundaries by Baseline Approximation

2.5.1 Goal and Scope of the Specific Methodological Developments

2.5.1.1 Goals

The collection and compilation of unit process data is generally recognized as having the greatest influence on the effort involved in conducting LCAs (see the elaborations in Section 2.1). Based on this finding, the methodological developments presented in this Section start with the assumption that there are neither data and models for the product system under examination from previous studies available nor that there are reliable screening methods that can give specific guidance for the product system modeling (PSM). Therefore, it deals with PSM by approximation as classified in Section 2.3 (see Figure 2-2 and Table 2-3), where the LCA starts from scratch. The general goal is then to establish the product system model efficiently, i.e. with as few unit processes as possible, while ensuring a high ‘level of confidence’.

In this thesis the 'level of confidence' of a simplified LCA is defined in relation to its ability to deliver a similar result as a detailed LCA, which can be seen in regards to the ranking of alternatives in comparisons (see Table 2-2) and/or in absolute values of category indicator results.

More specifically, the goal of the method that is elaborated in the following is to give guidance whether to include a unit process in the product system model or not in regards to the 'micro boundary selection' as defined in Section 2.3. The fewer unit processes are necessary to be included in the product system model the less data have to be compiled and fewer processes have to be modeled, resulting in a reduced effort for the LCA. This is extremely desirable and necessary for enhancing the application efficiency of LCA as long as the LCA still provides results with a high level of confidence usable for the specific decision at hand.

The goal of this Section can be summarized as developing guidance for the application of cut-off criteria, i.e. to guide the selection of the appropriate cut-offs leading to a system boundary and thus to a product system model that is sufficiently large to deliver the necessary level of confidence while being as small as possible in order to minimize the involved effort.

2.5.1.2 Scope and Specific Problem Setting

The developed methodology deals with the problem of 'micro boundary selection' as defined in Section 2.3. This is that part of the system boundary selection that deals with the questions which production of inputs or further processing of outputs should be included in the product system model (see [ISO 14041: 1998, No. 5.3.5 and 6.4.5] and [ISO/TR 14049: 2000, No. 5]).

In [ISO 14041] possibilities for cut-off criteria, i.e. criteria based on which a unit process can be excluded from the product system model, are defined. The criteria suggested are total mass, cumulated energy, and environmental relevance, all in relation to the respective aggregated value of the complete product system model, i.e. essentially to the value of the complete product system (not model!) in reality. If a cut-off criterion of e.g. 5% of environmental relevance is chosen, this means that every input flow, whose cradle-to gate production has an impact of less than 5% compared to the functional unit studied, can be omitted.

While the problem of accumulated cut-offs (e.g. if there are 25 inputs each contributing 4%) can be easily solved by applying an additional criteria relating to the aggregated cut-offs that overrides the single cut-off, there is one inherent problem with this procedure: the procedure can only be applied, if the complete product system with all unit processes and associated elementary flows is known and can be calculated (in this case the product system model would be a one-to-one mapping of reality, resulting in an isomorphic representation, where the model has the same complexity as the 'real world' system [Krallmann 1996]). The goal of models in general, however, is to simplify the system in reality. Therefore this isomeric representation is not suitable for the model of a product system, essentially for reasons of the effort involved, because this would virtually lead to a modeling of the complete industrial network due to the complex and quantitative interrelations (compare also [Suh et al. 2004]). The fact that the application of the cut-off rules as defined by ISO 14041 is not operational has – amazingly – been largely overlooked in the literature (exceptions being e.g. [Suh et al. 2004; Raynolds et al. 2000a and 2000b]). Other elaborations that deal in detail with the analysis and evaluation of different approaches for the application of cut-off criteria completely overlook this central issue (see e.g. [Fleischer and Hake 2002]).

As a result of this short discussion one can conclude that the definition of cut-off criteria and the resulting selection of inputs and outputs can only be interpreted as a vision, but not as a requirement[17], because one can only try to get as close as possible to the recommendations, but in reality they are impossible to follow fully. Additionally, the question on how to operationalize the application of cut-offs to produce product system models with as few unit processes as possible while delivering results with a high level of confidence is clearly not solved.

Therefore, Section 2.5 tries to propose a practical approach that is at least as good as existing procedures as outlined above, but much more practical and thus contributing to enhancing the application efficiency of LCA.

The scope of the methodology focuses on the production (cradle-to-gate) phase, because this is, for most products, the phase where most effort for data collection is needed and where the central problems of PSM as discussed in Section 2.1 are most obvious. While the focus is on the production phase, the findings will also be transferable to the end of life phase and to the modeling of any supply or disposal chain, also for the use phase.

Additionally, the developments are based on a product system model that is structured according to the sequential method as defined by [Heijungs and Suh 2002, pp. 100], the most commonly applied model for formulating and calculating the life cycle inventory of an LCA. The findings, however, will also be transferable to other approaches, where cut-off criteria also have to be addressed as e.g. in the matrix calculation with sparse matrices [Frischknecht et al. 2004c].

[17] One should note that the cut-off criteria as defined in [ISO 14041: 1998] are no requirements that must be fulfilled ('shall' requirement), but more general recommendations ('should' requirement).

The concepts and basic ideas behind this approach focus on

- the specific definition and application of cut-off criteria as well as on
- the employment of input-output LCA for establishing a baseline (baseline concept).

The baseline concept, which has not been proposed before, relies on the idea that one can establish a baseline that represents an approximation or at least an indication of the overall result of the LCA. Essentially, the baseline is an educated guess on the impact of the overall system, which then gives guidance when iteratively establishing the product system model. Using the baseline it should be possible to give a clue, using an indicator, if a simplified product system model, i.e. a partial model, is complete 'enough' or additional processes should be included in order to achieve the necessary level of confidence.

In the following the two principal concepts underlying the methodological developments are shortly described and then further elaborated and discussed in subsequent Sections.

2.5.2 Iterative Development of the Product System Model Based on Unit Process Specific Cut-offs

The general concept of cut-off criteria and their role in selecting inputs and outputs and the associated unit processes is generally known (see also Section 2.2.2) and described in the standard literature (see e.g. [Baumann and Tillmann 2004; Guinée et al. 2002; Lindfors et al. 1995; Wenzel et al. 1997]) and well as in normative and guiding documents of ISO (see [ISO 14040: 1997; ISO 14041: 1998; ISO/TR 14049: 2000] and the discussion in Section 2.5.1). However, the aforementioned and other publications list different options for defining cut-off criteria, both in regards to the inputs and outputs (criteria referring to mass, energy content, or environmental impact of a flow) and in regards to the reference (mass, energy content, or environmental impact of the reference flow; mass, energy content, or environmental impact of the overall flows of the product system, etc.). An overview and classification of the existing approaches defining system boundaries is given by [Lichtenvort 2004, pp. 12].

General scientific agreement, however, exists only in regards to the iterative nature of applying cut-offs, which should be step by step refined based on intermediate results (see e.g. [Guinée at al. 2002; Wenzel et al. 1997]). In Section 2.5.2.1 an operational and application oriented, i.e. practical definition of cut-off criteria is proposed, which is later further developed in conjunction with the baseline concept.

2.5.2.1 Process-Specific Cut-off Criteria as Conceptual Feature

Due to the inherent problems with defining cut-off criteria in relation to the overall system, a much simpler operational definition is proposed here, focusing on the single unit process only.

An underlying concept of the developed method is the procedure of recursive modeling by path analysis [Rebitzer and Fleischer 2000], based on the sequential method for establishing the product system model [Heijungs and Suh 2002, pp. 100]. With such a procedure one starts at the unit process that delivers the final product (the process at the 'gate') and then follows the input flows of this process to identify the processes that have these input flows as process products (outputs), and so on. For outputs that are going to waste management, recycling or emission control processes the same logic applies (following the output in order to identify the processes which take them up as inputs) This procedure, which allows to trace back all involved unit processes until the extracted raw materials and other elementary flows, is a systematic method to model the product system. It makes sense to apply this method in an iterative way, i.e. to judge the results in relation to the extend of the (intermediate) model as shortly described in Section 2.2.2.4).

When following the processes, often against the material and energy flow (for inputs), the question is then which of these inputs or outputs to cut-off, i.e. which of the unit processes producing the input or taking up the output to exclude from the PSM and which to include?

The concept proposed here is to use a simple process specific cut-off criterion relating to the mass of the input and output flows in question. This concept is illustrated in Figure 2-10.

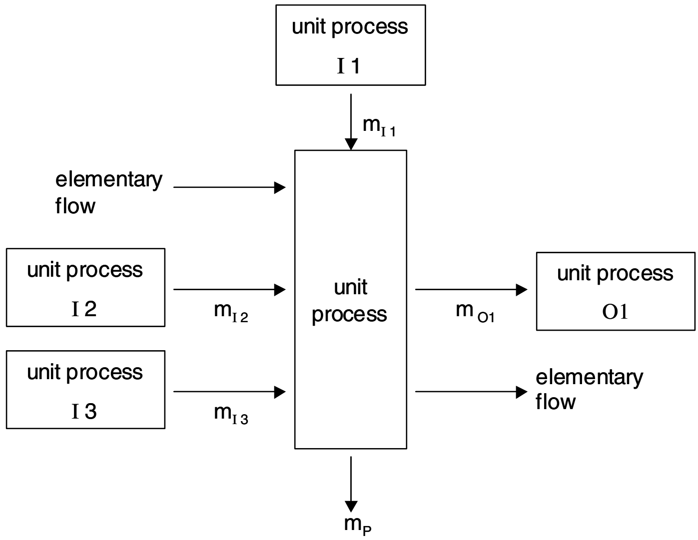

Figure 2-10: Concept of process-specific cut-off criteria

m_P: Mass of process product

m_{I1}: Mass of input from other unit processes (preceding foreground process, energy, ancillaries, etc)

m_{O1}: Mass of outputs to other unit process (by-product, flows to waste treatment, recycling, emission control)

Relating to the mass of the process product m_P a cut-off criterion of x% is defined. The selection if the connected unit process should be included or not is then done as shown in the following two Equations:

Cut-off x %: $m_{Ii}, m_{Oi} \leq (x\,\% \bullet m_P)$ ➔ outside boundaries Equation 2-5

$m_{Ii}, m_{Oi} > (x\,\% \bullet m_P)$ ➔ inside boundaries Equation 2-6

This only applies to mass flows, assuming that fuels and other energy carries than electricity and heat etc. are recorded as mass flows in the process data. Flows that can only be represented in energy units as well as elementary flows are not submitted to any cut-offs. The reasoning behind this different treatment is that energy flows, electricity being by far the most important, and the corresponding unit processes, are usually well documented and easy to obtain. For elementary flows the simple line of thought applies that in order to quantify their magnitude one obtains the exact data already, making a cut-off obsolete. Cutting off data one has available should not be the goal, unless it is required for reasons of consistency. Here it is assumed that process data are consistent to each other.

While the verification of the approach is not possible, in the following the approach is empirically tested using detailed cradle-to-gate LCAs of a number reference flows. For this the result of the detailed LCA is assumed to be 100% and it is tested how results with different process specific cut-offs affect the result. Therefore, it is outside the scope here to discuss how complete the results of the detailed LCA really are, important is the difference of simplified product system models compared to the very complex and large model of the detailed LCAs.

2.5.2.2 Testing of the Process-Specific Cut-off Approach

For the testing of the methodology, selected product system models of cradle-to-gate product system where chosen. The selection was mainly done based on the estimated completeness of the system models as well as the level of granularity, i.e. it was avoided to use highly aggregated data, where the cut-off would not yield meaningful results. The goal was to simulate PSM from scratch, where data are collected for each technical process and where one cannot rely on previously collected and aggregated data and resulting inventory models.

A description of the employed modeling software and database system can be found in Section 4.1.1.4 of this thesis. This system has been used to calculate category indicator results and CO_2 as inventory flow as a function of the cut-off. For this calculations for 1 kg of the following materials were carried out:

- The plastics PE (HD) and ABS
- The metals primary aluminum, cast iron, and primary steel
- Glass fibers and paper

These examples were selected, because they represent different types of materials (plastics, metals, fibers) that are frequently used in LCA studies.

For the life cycle impact assessment the categories of IMPACT 2002+ [Jolliet et al. 2003] also used for the case study on the automotive front subframe system were employed (for a description of the method and the category indicators see Section 4.1.1.5).

Figure 2-11 to Figure 2-17 show the analyses of the process specific cut-offs and the coverage of the impacts compared to detailed LCA (0% cut-off, which leads to a coverage of 100%). The Figures on the left show the range from 60% to 0% cut-off while the Figures on the right give a more detailed insight into the behavior of the system when the cut-off is varied from 10% to 0%.

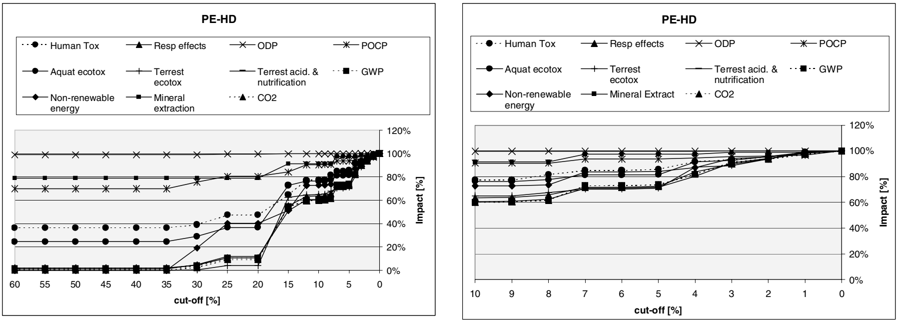

Figure 2-11: Analysis of cut-offs for PE-HD granulate

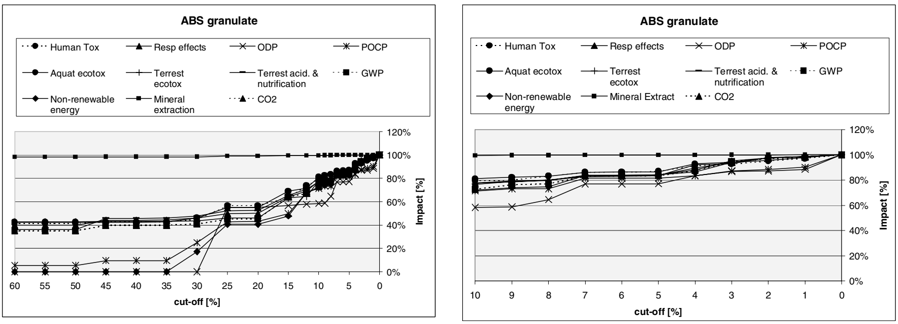

Figure 2-12: Analysis of cut-offs for ABS granulate

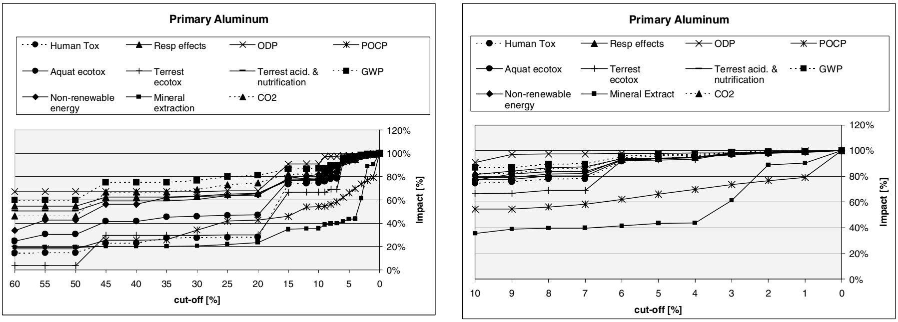

Figure 2-13: Analysis of cut-offs for primary aluminum

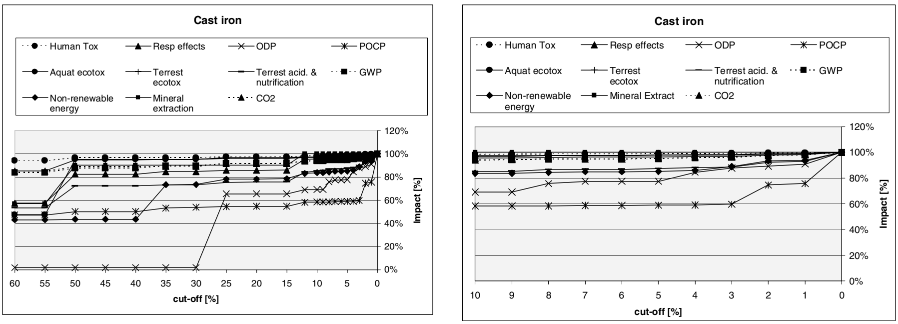

Figure 2-14: Analysis of cut-offs for cast iron

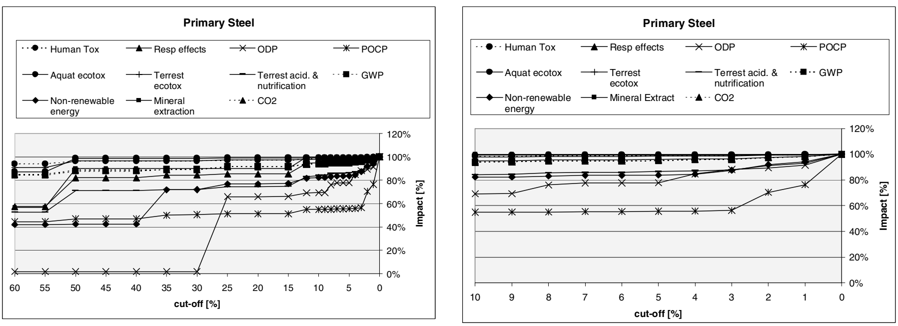

Figure 2-15: Analysis of cut-offs for primary steel

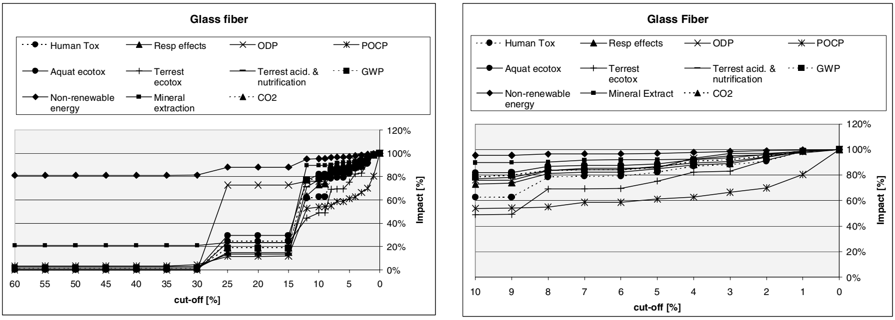

Figure 2-16: Analysis of cut-offs for glass fibers

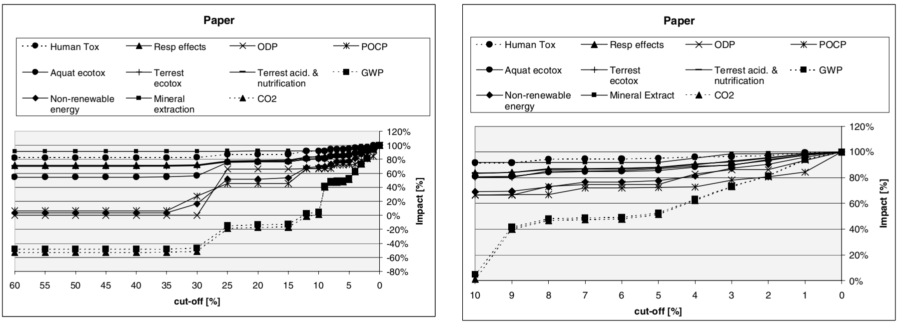

Figure 2-17: Analysis of cut-offs for paper

When analyzing the data of Figure 2-11 to Figure 2-17 the question arises, how much coverage of the total impact is 'good enough' for a simplified LCA, in relation to a detailed LCA (100%). While a general answer cannot be given - this will largely depend on the goal and scope of the study - an attempt is made to start with the a figure based on the so-called Pareto principle [Juran 1937], also known as the 80-20 rule and to postulate 80% coverage as an acceptable value. This also corresponds to the 20% rule of thumb regarding identification of significant differences as mentioned in [de Beaufort-Langeveld et al. 1997] (see also Section 2.2.2.4). On first view one can conclude that definitely a cut-off smaller than 10% is needed to reach the 80% mark for all impact categories. On the other hand, it is also clear that very small cut-offs in the range of 0-2% do not seem necessary as well.

In order to analyze this in more detail, Figure 2-18 shows the relative frequency of the largest cut-off with which all impact categories across all materials reach a coverage of

80%. This also includes the two alternatives for the frontend subframe system analyzed in detail in Section 4.1.2. Here one can see that a cut-off of 4% seems to have a very high probability of reaching 80%. Based on this sample, in 92% of all cases, the 4% cut-off is sufficient to cover 80% of the impacts of all impact categories.

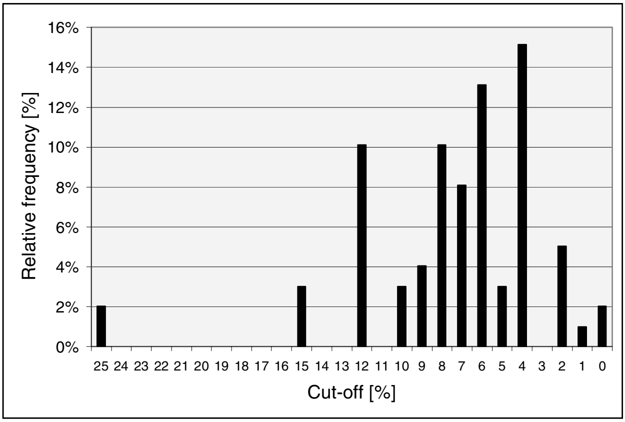

Figure 2-18: Relative frequency distribution of cut-off criteria of 9 materials and 11 impact category indicators (incl. CO2)

A second question is then if the allowable cut-offs vary significantly between impact categories. Therefore, the coverage of the recorded impacts as a function of the cut-off were recorded by impact category across all materials and components. For seven out of the eleven indicators, even 100% coverage is reached at 4% cut off. For also seven out of the eleven categories 80% is (almost) reached with a 6% cut off. Apparent differences between the indicators can not be identified from this sample. In Table 2-5 a value of 100 at e.g. a cut-off of 4% means that in 100% of the cases the required 80% coverage of the detailed LCA is reached.

Table 2-5: Frequency analysis of the data points by midpoint impact category (in %; some cells are empty because there is no change of the impact category result when going from a larger to this cut-off)

Cut-off [%]	Human Toxicity	Respirator. effects	ODP	POCP	Aquat. ecotox.	Terrest. ecotox.	Terrest. acid. & nutrific.	GWP	Non-renew. energy	Mineral extract.	CO_2
30-60	33	22	11	0	22	22	0	22	11	44	22
25				11				33			
20											
15			22							67	33
12		33			33		33		44	78	
10					56	33					
9	44				67			44			44
8	56	67	44			44	67				
7				22	89			56	78		56
6	78	78	56	44	100	56	78	78	89	89	67
5	100	89									
4		100	100			100	100	89	100		89
3											
2				56				100		100	100
1				67							
0				100							

While this analysis, based on 99 samples (9 materials/components with each 11 midpoint impact categories), is still limited, it points to a conclusion that a process-specific cut-off of about 5% might be a good general recommendation. It seem that if this is applied and then checked by an sensitivity analysis, based on which the decision should be made to further refine or not, reliable results with a high level of confidence can be achieved.

In addition to the analysis based on midpoints, an analysis of the same data and with the same methods, but based on unnormalized endpoint categories of IMPACAT 2002+ was carried out. The corresponding results are shown in Figure 2-19 and Table 2-6.

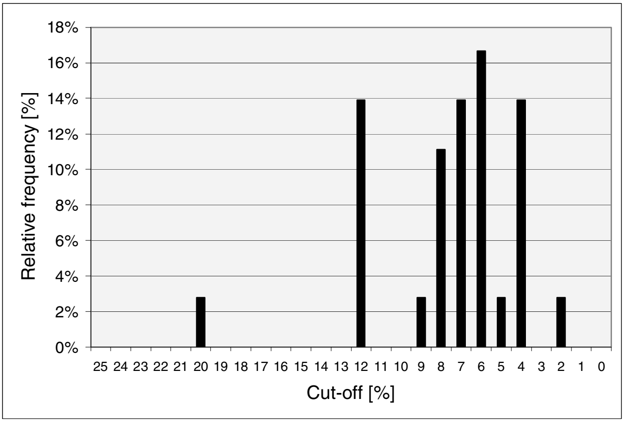

Figure 2-19: Relative frequency distribution of cut-off criteria of 9 materials and 4 endpoint categories

The analysis of the endpoint categories confirm the previous result. In order to reach 80% coverage of the impacts of a detailed LCA without any cut-offs, the recommendation to use 4-5% seems reasonable.

Such a procedure also has the advantage, in contrary to other cut-off rules, that it can also be implemented by computer algorithms, as done for the specific test cases. The ability of methodology approaches to be implemented in (semi) automatic algorithms is a clear requirement of any methodological approach for conducting LCA in an efficient was. Specific computerized systems for conducting LCAs have become more commonplace [Frankl and Rubik 2000, p. 232; Vigon 1996], because they facilitate efficient decision support [Bullinger and Jürgens 1999; Schmidt 1999; Vigon 1996]. .

Software based assessments are particularly attractive if LCA is applied in the context of Design for Environment [Ryan 1999].

Table 2-6: Frequency analysis of the data points by midpoint impact category (in %; some cells are empty because there is no change of the impact category result when going from a larger to this cut-off)

Cut-off [%]	Human Health	Ecosystems Quality	Climate Change	Resource Depletion
30-60	22	22	22	11
25				
20			33	
15				
12	33	33		44
10				
9			44	
8	67	44		
7		56	56	78
6	78	78	78	89
5	89			
4	100	100	89	100
3				
2			100	
1				
0				

If a cut-off procedure as described and implemented above is implemented in software systems then they can support the practitioner of an LCA in the task of establishing the product system – in addition of computation and evaluation models that are stored based on existing data.

2.5.3 Establishing a Baseline Based on Input-Output Analysis[18]

The central idea followed in this Section is to create a baseline based on input-output LCA. Because it is often claimed that input-output LCA has a much higher coverage of the overall impacts due to the consideration also of non mass- and energy-related environmental impacts of services (for instance services such as banking), first the contributions of such services to the overall result of input-output LCA are determined (contribution analysis, see below). These adapted figures from input-output LCA should then serve as a baseline for estimating the 100% coverage as explained in the previous Section.

The methods described and applied in this Section are founded on research carried out by Wassilij Leontief [Leontief 1936]. Leontief's macro economic input-output model forms the basis for the so-called input-output life cycle assessment methodology (input-output LCA), a macro economic based model of environmental impacts (see also the brief discussion of this concept in regards to screening and simplifying LCA in Section 2.2). This method makes use of national level economic input-output tables, linked with databases on environmental releases per sector. The U.S. input-output accounts differentiate the production of roughly 500 different commodities by roughly 500 different industries.

2.5.3.1 Contribution Analysis

The way to fully capture the total importance of all uses of commodities in a product's supply chain, including both the direct impacts plus the supply chain impacts of the commodities, is to compute the difference in total contribution comparing a complete model with one that has all usage, and thus all direct burdens plus upstream burdens, of the commodities suppressed.

Background and goal of the analysis

For the present analyses input-output tables of the U.S. economy from 1997[19] were used. Input-output tables show, amongst other information, the financial flows between industries of an entire economy. The model used for this study is based on an industry-by-industry input-output direct requirement coefficient matrix, denoted 'A'. It includes data for 490 interlinked industry sectors, each sector producing one principal commodity (simplified view of input-output analysis). Each element of 'A' (a_{jk}) denotes the amount

[18] This Section is largely based on a co-authored paper (see [Kaenzig et al. 2005])

[19] Such input-output data are available only for some countries, with the one of the United States of America being the most detailed one.

of one directly required commodity (commodity j) to produce one unit of an industry output (industry k).

In the following a computational procedure is developed, which allows for the elimination of the influence of financial flows of and caused by defined commodities. With this computation, both the direct as well as the indirect influence (influence in the supply chain) of these commodities are suppressed. The resulting input-output model resembles a theoretical model of an economy where those commodities do not exist. In the following the excluded commodities are denoted as 'eliminated'. In order to fully capture the total importance of all uses of the eliminated commodities, the difference in financial flows to a complete model is calculated. For the complete model the input/output tables of the total U.S. economy (see above) is employed.

Computation

First, the modified direct requirements coefficient matrix, A_r, is created in which all inputs of the eliminated commodities to other sectors have been zeroed out. Considering that the columns of A are the using industries and the rows are the supplying industries, the coefficient rows of the eliminated commodities are set to zero by an array multiplication[20].

$$A_r = A \ .\!* \ R \qquad \text{(Equation 2-7)}$$

A_r: Modified direct requirements coefficients matrix (j x k)
A: Direct requirements coefficients matrix (j x k)
R: Matrix that consists only of 1s except in rows and columns corresponding to the commodities and industry sectors to eliminate, which are set to 0 (j x k)
.*: Element by element multiplication

The principle can be visualized as follows: on the one hand there is the 'A' matrix with direct coefficients for each industry/commodity used for the modeling of the total economy. On the other hand a model of the economy is created where one or several commodities do not exist and do not induce any inter-industrial financial flows and thus environmental impacts in the input-output LCA model. This is illustrated in Figure 2-20.

[20] Alternatively, this could also be done by removing the rows from the Direct requirement matrix A. The other method has been chosen in order enable a standardized handling for all the 490 industry sectors with an unchanged size of the matrices.

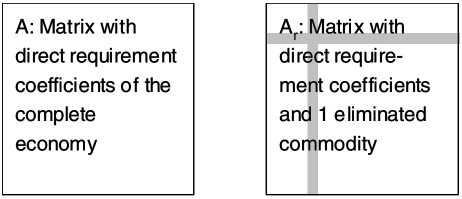

Figure 2-20: Illustration of the input-output matrix, where commodities are eliminated

Then, the complete input-output LCA model (U.S. economy) to compute the total environmental flows (emission flows, resource use) due to a final demand 'y' is set up. Further information on such calculations can be found in [Leontief 1936].

$$E_t = B \cdot (I-A)^{-1} \cdot y \qquad \text{(Equation 2-8)}$$

E_t: Total environmental burdens in the complete economy (i x j) [physical units]
B: Environmental intervention matrix (i x j) [physical units / output ($)]
i: Environmental intervention (environmental factors: emissions and resources use per industry output ($))
j: (=k) supplying industry/commodity (490 in the case of the database used [Norris 2002])
I: Identity matrix (j x k)
k: (=j) demanding industry (490 in the case of the database used [Norris 2002])
A: Direct requirements coefficients matrix (j x k)
$(I-A)^{-1}$: Total (direct and indirect) requirements matrix (j x k) [$ / $]
y: Diagonalized final demand vector (j x k)

Analogously, the modified input-output LCA model (U.S. economy without eliminated commodities) to compute the environmental burden due to a final demand 'y' is set up.

$$E_r = B \cdot ((I-A_r)^{-1} \; .\!* \; R) \cdot y \qquad \text{(Equation 2-9)}$$

E_r: Environmental burdens in a modified economy (i x j) [physical units]
B: Environmental intervention matrix (i x j) [physical units / $ output]
i: Environmental intervention (environmental factors: emissions and resources use per industry output ($))
j: (=k) commodity (490 in the case of the database used [Norris 2002])
I: Identity matrix (j x k)
k: (=j) using industry (490 in the case of the database used [Norris 2002])
A: Direct requirement coefficients matrix (j x k)
$(I-A)^{-1}$: Total (direct and indirect) requirements matrix (j x k) [$ / $]

R: Matrix that consists only of 1s except in rows and columns corresponding to the commodities and industry sectors to eliminate, which are set to 0 (j x k)

y: Diagonalized final demand vector (j x k)

.*: Element by element multiplication

Subsequently, by subtracting the result obtained with the modified model from the result obtained with the complete model provides the environmental flows due to the eliminated commodities.

$$E_c = E_t - E_r \qquad \text{(Equation 2-10)}$$

E_c: Environmental burdens of the eliminated commodities in the complete economy (i x j) [physical units]

E_t: Total environmental burdens in the complete economy (i x j) physical units]

E_r: Environmental burdens in a modified economy (i x j) [physical units]

It is possible to integrate Equations 7-10 in one equation:

$$E_c = \overbrace{B}^{\text{Emission, resource use factors, etc.}} \cdot \overbrace{\{[(I - A)^{-1}]}^{\text{A: Complete U.S. economy}} \cdot y)] - \overbrace{[(I - A .* R)^{-1} .* R)]}^{A_r\text{: U.S. economy without the commodities analysed}} \cdot \overbrace{y\}}^{\text{Final demand}}$$

Ec: Environmental burdens of the eliminated commodities in the complete economy (i x j) [physical units]

B: Environmental intervention matrix (i x j) [physical units / $ output]

i: Environmental intervention (environmental factors: emissions and resources use per industry output ($))

j: (j=k) commodity (490 in the case of the database used [Norris 2002])

I: Identity matrix (j x k)

k: (j=k) using industry (490 in the case of the database used [Norris 2002])

A: Direct requirements coefficients matrix (j x k)

(I-A)-1: Total (direct and indirect) requirements matrix (j x k) [$ / $]

R: Matrix that consists only of 1s except in rows and columns corresponding to the commodities and industry sectors to eliminate, which are set to 0 (j x k)

y: Diagonalized final demand vector (j x k)

.*: Element by element multiplication

2.5.3.2 General Results

To obtain general results all financial flows related to service sectors have been removed in the reduced model.

The analyses has been restricted to the comparison of CO_2 emissions. Figure 2-1 shows those sectors of the 490 where the eliminated services have the highest contributions, ranging up to 70 to 80%, depending on the specific data base used. The highest contributions are found with complex products such as motor vehicles and with goods and services that use mainly commodities that have been considered out of scope for process LCA.

However, it is also evident that the overall contributions of the service sectors are not that large. For instance already that sector which is still in the top 10% of sectors in regards to contribution of services (40 sectors out of 490) only shows an absolute contribution of services of around 20%

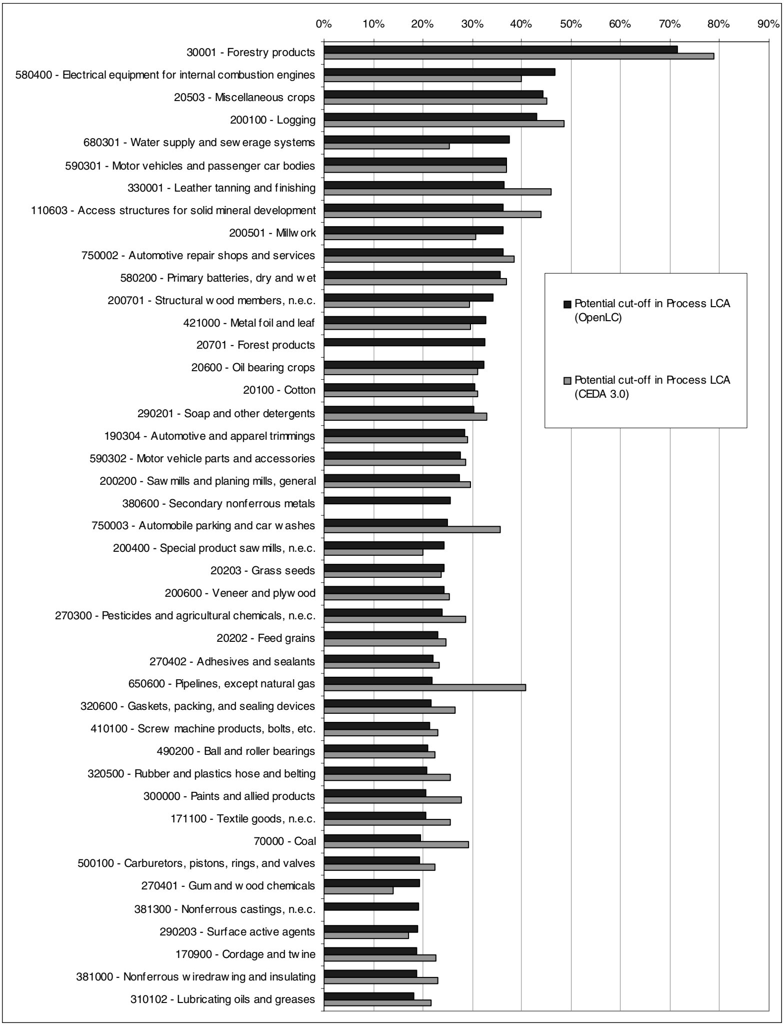

Figure 2-21: Contribution of service sectors to total CO_2 emissions

2.5.3.3 Specific Results of Selected Materials

For some materials the data of input-output LCA without service sectors ('reduced input-output LCA') was analyzed and compare to the results of process LCA. Such a comparison is shown in Figure 2-22

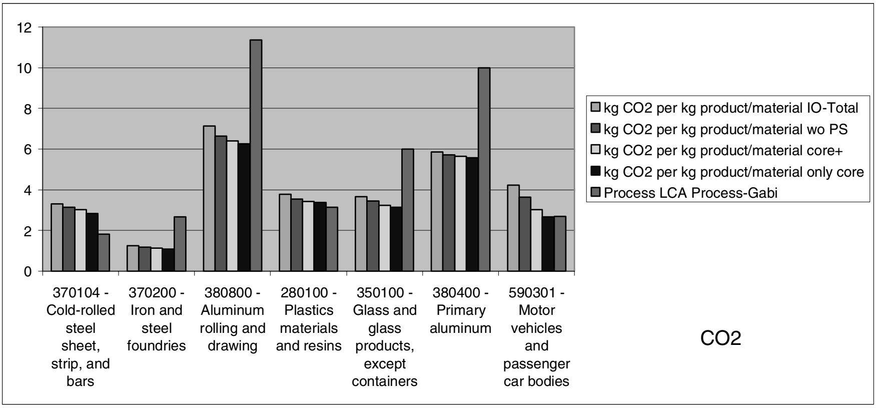

Figure 2-22: Comparison of results of reduced input-output LCA with process LCA (the various input-output results ("kg CO2 per .." differ due to different levels of exclusion of service related commodities, since their selection is not unambiguous)

From the data series the four left columns present different levels of a reduced input-output LCA, while the bar on the right always corresponds to the process LCA result. The picture is not clear, sometimes the input-output result is higher, sometimes that of the process LCA.

Additionally, if one looks at price variations within one product group (due to qualities, requirements), the picture can change extremely (for instance the data in Figure 2-22 were calculate with using prices of 5 USD for 1 kg aluminum profile or sheet and 3 USD for 1 kg of steel product for the input-output LCA). These, however, are rather low figures, which are typical for standard products used e.g. in the building sector. If more sophisticated profiles or sheets ranging from 15 to 30 USD, for instance for high performance vehicles or aircrafts, respectively, per kg material are chosen as a base, Figure 2-23 is the result.

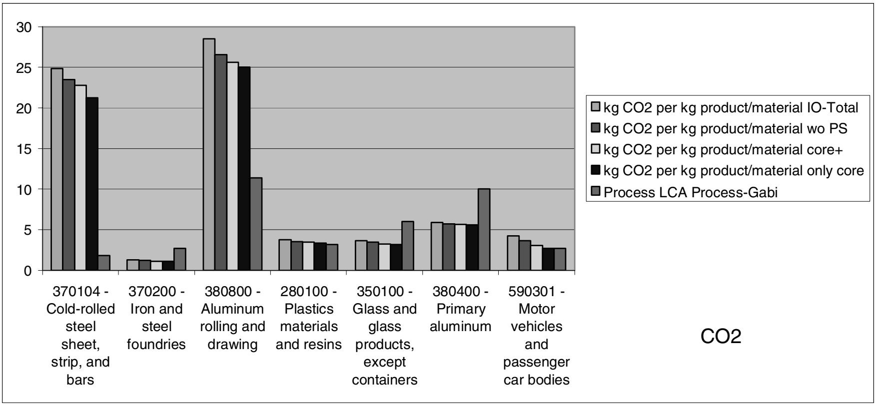

Figure 2-23: Same calculations of input-output LCAs as in Figure 2-21, though using higher product prices for steel and aluminum products

Figure 2-23 demonstrates that the aforementioned product and therefore price choices have an extremely high influence on the results and that the results of the input-output LCA, even without the influence of the service sector are difficult to use for decision-support.

2.5.3.4 Recommendations on Selecting the Baseline

From the short elaborations above one can clearly see that the price variations within one product group influence the results of input-output LCA tremendously. Since most industrial applications of LCA concern comparisons of similar products or improvements of one product, it seem that results as delivered by input-output LCA cannot be used. The different products within one sector vary too much in order to allow a reasonable assessment. Another path for further research along the same lines, but based on different data, could be to apply the same concept of baseline selection to proxy data from process LCA, i.e. by selecting the “closest” available data set. It is recommended to explore this in future research work.

Uses of input-output LCA, also including the contribution of services, can probably better be found in areas, where process data do not exist at all or for sectors with a very high share of services that are difficult to account for in process LCA.

3 Life Cycle Costing in LCM

3.1 Background and Rationale

In the context of life cycle management (LCM) (see Section 1.2) 'cost management'[21] plays a crucial role. Therefore, profitability has to be taken into account as a central element of LCM [Hunkeler and Rebitzer 2003]. Environmental considerations, on the other hand, are often viewed as obstacles to business development, particularly in the very short term. This is where, within the sustainability framework, the concept of 'life cycle costing' (LCC) emerges. LCC is an essential link for connecting environmental concerns with core business strategies [Hunkeler et al. 2004]. Synergies between the environmental and economic considerations have to be utilized in order to move towards sustainable development [Dyllick and Hockerts 2002; Hunkeler and Rebitzer 2003].It is of utmost importance to assess the (potential) future consequences of decisions if more sustainable products and processes and thus more sustainable business practices are the goal. It is well known that costs and revenues [Ehrlenspiel 1985] as well as the environmental impacts of products [Keoleian 1996] are determined to a high percentage in the design phase of products and processes (see Figure 3-1) and that already during this phase a long term view incorporating the full life cycle should be taken.

However, in many instances business practices have sufficiently short perspectives which limit the time, resources, or experience to consider costs outside of the company's gate, for example regarding usage and disposal of a product. Therefore, LCM challenges business management to incorporate environmental management along the life cycle into research and development, purchasing, manufacturing and sales planning and to interact with academia, governmental organizations, NGOs and local interest groups to achieve business, environmental, and social advantages (as described e.g., in [Fussler and James 1996]).

When addressing the economic pillar in LCM conventional cost accounting and cost management are not or only partly suited to assess costs and revenues over the life cycle of a product, since they do not have the specific systems perspective of LCM. Therefore, methods are necessary that can integrate and link existing financial data and specifically cost information with metrics in life cycle approaches (in the following such methods are summarized under the term life cycle costing – LCC).

[21] Cost management encompasses all (control) measures that aim to influence cost structures and cost behavior precociously. Among these tasks the cost within the value chain have to be assessed, planned, controlled, and evaluated. [Hilton et al. 2000; Kaplan and Cooper 1997]

In short, LCC, with its systems approach, is a means to integrate the life cycle perspective into the costing view, e.g. by considering use and end-of-life costs in addition to the production costs/product price. Also, LCC can be used to move the environment from an indirect cost in the environment, health, and safety (EHS) units of the actors in the value chain to considerations as a direct, manufacturing, and liability issue, and, under appropriate conditions, an asset [Hunkeler and Rebitzer 2003].

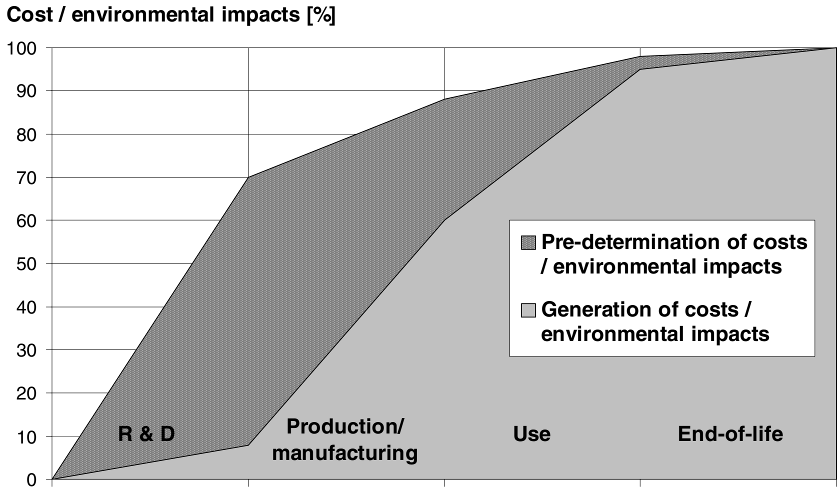

Figure 3-1: Generation and pre-determination of costs and environmental impacts in the life cycle of a product (principal figures, varying from product to product) [Rebitzer 2002]. The notation 'costs / environmental impacts' refers to the fact that this principal figure applies to costs or environmental impacts over the life cycle.

The systems approach for both the environmental as well as the economic aspects, is reflected in Figure 3-1, based on a physical (see below) product life cycle. The principal generation (where/when they occur) as well as the pre-determination (when/where they are influenced) of costs and environmental impacts are illustrated. It shows the relevance of addressing environmental issues and life cycle costs from the very beginning of product planning/design/development (research and development – R&D). The research and development phase does, itself, not cause a great share of the overall costs and environmental impacts. In general between 3-25% of the direct cost of the product manufacturer, which itself is only one segment of the total life cycle cost, can be allocated to R&D [Ehrlenspiel et al. 1998]. However, it is extremely significant for the determination of the costs and impacts in the other phases of the life cycle. The research and development phase is the key to a cost efficient product with a good environmental performance (minimized environmental impacts such as resource consumption and emissions).

It is important to note that the systems approach in LCC and the addressed life cycle resembles the physical life cycle as in LCA (see the definition of the life cycle in [ISO 14040: 1997]), which refers to the actual life cycle of a unit of product and thus to the functional unit. This life cycle must not be confused with the marketing and sales life cycle of products (see [Levitt 1965] and [Wöhe 1993, p. 679 ff.]) which looks at the establishment of a product on the market (introduction, growth phase, stabilization, phasing out) and focuses on the revenues and benefits of the number of units sold.

In the following Sections first the state of the art in LCC is reviewed (Section 3.2). Subsequently a conceptual framework for LCC within life cycle management is developed (Section 3.3) and specific methodological procedures for efficiently analyzing life cycle costs based on product system models of LCA are described (Section 3.4). In later Sections case studies of the developed approach from the automotive sector (Section 4.1.3) and for the service of waste water treatment (Section 4.2) are presented.

3.2 State of the Art in Life Cycle Costing

3.2.1 Historic Roots and Developments

LCC in the pure economic sense ('purely economic LCC')– long before the emergence and development of sustainable development and (environmental) life cycle thinking - has first been used in the 1960s by the US Department of Defense for the acquisition of high cost military equipment such as planes, or tanks [Sherif and Kolarik 1981]. The rationale was that the purchasing decisions should not solely be based on the initial acquisition cost, but also on the costs for operation and maintenance, and, to a lesser degree, disposal.

Building on this tradition, LCC has so far been applied mainly for decisions involving the acquisition of capital equipment or long lasting products with high investment costs per unit. Early areas of application have been [Sherif and Kolarik 1981]:

- buildings, mainly for commercial or public purposes,
- energy generation and use,
- transport vehicles with high investment costs (mainly from the aerospace sector), and
- major military equipment and weapon systems.

In the US Department of Defense several directives for the calculation of life cycle costs and design to costs were developed in the beginning of the 1970s . Also in the United States several regulations have been issued that require the calculation of life cycle costs for the acquisition of public buildings [Zehbold 1996, p. 77 ff.]. For the latter a standard exists [ASTM 1999]. However, life cycle costing has usually been limited to very sector or product specific applications as listed here.

[Sherif and Kolarik 1981] give a comprehensive overview of the aforementioned applications, the used costing models, and the corresponding literature. They conclude that "… LCC has developed more as a result of specific applications rather than hypothetical models." This conclusion is still more or less valid to date, even for the purely economic LCC analysis. A generally usable methodological framework or model did not evolve, even though there are have been tendencies in this direction. The concept that most closely resembles such a general method has been developed initially by [Blanchard 1978], later refined by [Blanchard and Fabrycky 1998]. This has its roots in systems engineering and focuses on the assessment and comparison of technological alternatives. It structures the life cycle of a product or system into R&D, production and construction, operation and support, and retirement and disposal. The structure, in addition to the cost elements considered and the systems view, is very similar to the life cycle approach in LCM and, thus, a good basis for the development of an LCC method for LCM. However, also [Blanchard and Fabrycky 1998] do not elaborate a methodology that gives guidance on how to quantitatively calculate and compare costs, they present LCC rather in the sense of qualitative 'life cycle thinking' and stress the importance of the systems view.

Concluding this short discussion on the historic roots and developments of LCC one can say that purely economic LCC has never developed into a broad and generally applicable methodology, because this was not aimed for. Instead it has been used – based on the principal life cycle view – in the form of very application specific procedures and limited to few sectors with specific approaches. In this context, one can raise the question why this intriguing and simple concept was never broadly established in industry and the public sector, as for instance quality management approaches were. Reasons can be, for example, organizational barriers or budget restrictions. For instance, in many cases the purchaser and the operator of a product are different stakeholders with separate budgets within one organization or are even affiliated to different organizations, making a consistent cost assessment for the life cycle difficult. On the other hand, the practice of annual budgets or budgets for specific investments often limits the long term view, since the decision frequently has to be based solely on the financial resources available today, without the option of considering future costs and revenues.

Existing purely economic LCC approaches or conventional cost management practices, while addressing very relevant issues and containing elements of importance to LCM, are usually not suitable for a consistent assessment of the economic implications of a product life cycle in a sustainability framework. In addition, they are not connected to the established framework (LCA) for the assessment of the environmental dimension and thus – even in cases were suitable in principal – would cause unnecessary high efforts.

3.2.2 LCM related Life Cycle Costing

One can trace the roots of LCC in LCM back long before the term LCM was created and even before LCA has been established. The 'Produktlinienanalyse' [Öko-Institut 1987] was probably the first method, which assessed environmental, economic, and social aspects of product life cycles in parallel in order to arrive at a three dimensional holistic evaluation. However, in contrary to LCA and LCC as presented in this thesis, those assessments were of a mainly qualitative nature, lacking quantitative indicators and calculations.

After the establishment of the life cycle assessment methodology (see [Hunt and Franklin 1996]) and the sustainability framework (strongly influenced by [WCED 1987] and [Royston 1979]) in the early 1990s, the concept of LCC was (re)discovered as part of the sustainability debate. Research attempting to create an economic parallel of LCA started in the mid 1990s in the US by [Hunkeler et al. 1998] and Germany (since 1996), the latter partially pioneered by the author of this thesis. In 1997, he co-authored the first paper where life cycle costing, based on the life cycle inventory analysis (for a further elaboration of this method, see Section 3.4), was proposed for evaluating the economic dimension of sustainability: "The … Life Cycle Costing (LCC) method is based upon the inventory analysis (LCI) of the LCA module. Included are all costs and benefits from cradle-to-grave." [Fleischer et al. 1997]. A first method based on this proposal was presented in the year 2000 [Rebitzer 2000]. A related, though inherently different, method for assessing both environmental and economic affects of products from a life cycle perspective has been proposed by [Norris 2001]. The latter research focuses on the integration of LCA with independent LCC analyses (see the 'purely economic LCC' in Section 3.2.1), managerial cost accounting, and scenario-based economic risk modeling in order to use results from both environmental and economic assessments for decision-making in product design and development.

The relevance of the economic pillar in LCM has been growing ever since the conceptual work on LCM has started at the end of the 1990s. For instance, specific LCC applications complimenting results of LCA for the automotive sector [Bubeck 2002] and for train components [Trebst et al. 2001] have been developed. While still in its infancy, a common understanding of LCM and the role of business relevant economic information and decision support, has been created through the SETAC Working Group on Life Cycle Management, which started in 1998 and was recently concluded with the publication of the Working Group Report [Hunkeler et al. 2004].

However, while the importance of LCC for LCM is unchallenged, at present, there is not yet fully scientific and procedural agreement between the various stakeholders regarding terminology, methodology, data formats, reporting, etc [James 2002]. Therefore, there is a need to develop the methodological background of and application procedures for LCC.

This need has also been recognized by SETAC with the consequence that a Working Group (WG) on LCC was founded in 2002 [Rebitzer and Seuring 2003]. Within the elaborations of the SETAC LCC WG a survey on existing LCC approaches was conducted [Ciroth and James 2004]. This survey revealed that three types of LCC exist (based on [Rebitzer et al. 2004b]):

- ‘Conventional LCC’: Pure LCC studies, concentrating on assessing (conventional) costs of a product that are directly covered by life cycle stakeholders, without any connections to environmental assessments (purely economic LCC that has been practiced since the 1960s, see Section 3.2.1). This is a one dimensional approach, without connections to the other axes of sustainability.

- ‘Societal LCC’: LCC studies including monetized environmental effects of the investigated product (leading to an environmental life cycle impact assessment result expressed in monetary units). These are highly aggregated assessments that do not differentiate the three elements of sustainability, therefore risking intransparency and extremely high uncertainties[22] (see also the discussion below).

- ‘Environmental LCC’: LCC studies performed in conjunction with a non-monetizing assessment of the environmental impacts of the product, typically via an LCA, where the results of the LCA and the LCC are kept separate.

One has to note that the terminology (‘conventional, ‘societal’, and ‘environmental’ LCC) is still in development, probably needing further refinement and clarification. Within the toolbox of LCM, ‘conventional’ and ‘environmental’ LCC studies, respectively the underlying methodologies, can be used, though the ‘environmental LCC’ is probably the

[22] The argument of a ‘42-method’ referring to [Adams 1979] (see the related footnote in Section 2.4.2.4), applies here even more due to the significantly higher level of aggregation and related to completely different values and systems.

most relevant. The ‘societal LCC’ (as e.g. proposed by [White et al. 1996]) proposed by is either the least relevant or not at all suitable in the context of LCM, since environmental effects are covered by other life cycle approaches outside the scope of the economic assessment. Furthermore, an ‘all-in-one’ method (potentially also including monetized social effects) cannot be aligned with the overall goal of LCM (to ‘put sustainable development into (business) practice’ [Hunkeler et al. 2004]) due to the involved uncertainties and the lack of acceptance. In addition, such methods which include the monetization of externalities have been shown to be very problematic and arbitrary (see [Ackerman and Heinzerling 2004]). Therefore, ‘environmental LCC’ studies and methodologies are most suitable for the integration of economic aspects in LCM. They stand the chance to consistently and efficiently address economic and environmental issues from a life cycle perspective [Rebitzer et al. 2004b]. ‘Conventional LCC’ approaches should be used if no information on environmental impacts is needed, while ‘societal LCC’ targets other applications than LCM such as cost-benefit analyses. The following elaborations on LCC in LCM and the LCI-based LCC are of the ‘environmental LCC’ type.

Drafts of chapters of this thesis as well publications of the author created during the research on LCC since 1996 have contributed to the SETAC WG, which is well on its way to establish a standard LCC methodology for sustainability assessments, similar to the now commonly accepted established LCA methodology.

3.3 Conceptual Framework of Life Cycle Costing in LCM[23]

In this Section the framework for LCC in LCM is defined. This framework will then serve as a basis for a specific LCC methodology, life cycle inventory based LCC (see Section 3.4). However, the developed framework is universal and can also be used for life cycle costing methodologies that are not based on LCA data.

3.3.1 Boundaries of LCC

Figure 3-2 summarizes the conceptual framework of LCC, based on the physical product life cycle, with which also the relationship of LCC to LCA and social assessments (e.g., including employment conditions and unemployment rates) in LCM can be explained.

It has to be stressed that – if the LCC is part of a sustainability assessment in conjunction with LCA – the functional unit of the life cycle costing analysis has to be identical to that of the LCA.

[23] Section 3.3 is largely based on a previous publication by [Rebitzer and Hunkeler 2003]

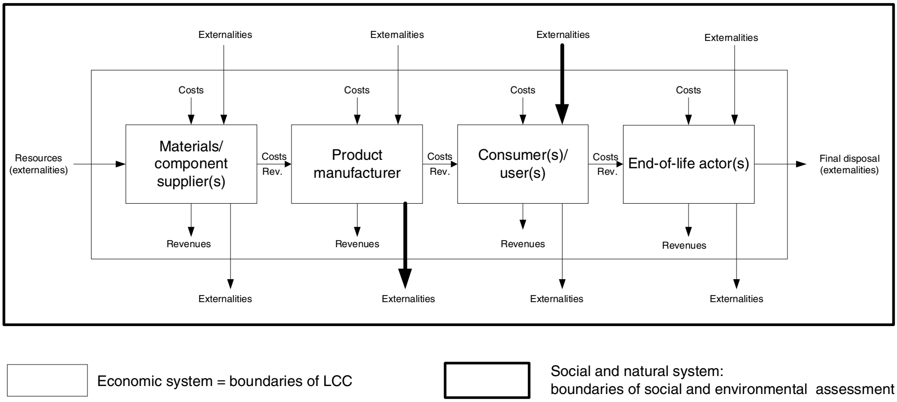

Figure 3-2: The Conceptual Framework of LCC [Rebitzer and Hunkeler 2003]

One can differentiate, in Figure 3-2, between:

1) Internal Costs along the life cycle of a product, with 'internal' implying that someone (a producer, transporter, consumer or other directly involved stakeholder) is paying for the production, use, or end-of-life expenses and, thereby, it can be connected to a business cost, and, indeed, liability. This concerns all the costs and revenues within the economic system (inside the fine lines as represented in Figure 3-2).

2) External costs that are envisioned to include the monetized effects of environmental and social impacts not directly billed to the firm, consumer, or government, etc. that is producing, using, or handling the product. These are the so-named 'externalities' so popular in LCC and LCA debates, which are outside the economic system, though inside the natural and social system as illustrated in Figure 3-2.

In this context it is important to note that the terms, and boundaries for, economic, as well as social and natural systems, are not synonymous to the product system in LCA. For a common assessment of two or three of the LCM elements, the product system has to have the same system boundaries, as stressed by [Klöpffer 2003] and [Schmidt 2003].

If one examines a perfectly free market, without any taxes or subsidies to account for externalities, LCC can focus only on the economic system if the following condition is satisfied: LCC is applied in conjunction with environmental and/or social assessments for the same product system with the same system boundaries. Under such an, albeit simplified, scenario, all externalities are covered by the other assessments within the LCM toolbox. On the other hand, if taxes and subsidies exist and they are fair[24], or justifiable based on the collection of a social overhead based on a product's burden, then the economic system can be used as a simplification for the complete social and natural system. Therefore, if all externalities would be completely and perfectly covered by tax and subsidy mechanisms, nationally and supra-nationally, LCC would provide all the necessary information for LCM, rendering systematic environmental and other assessments unnecessary for all but new products.

Clearly, the aforementioned macroeconomic assumptions are oversimplified, and, in particular, the latter (complete coverage of externalities by tax and subsidy mechanisms) does not approach reality. If one assumes the tax system is valid for certain products, and not so for others, from socio-environmental perspectives, then integrating externalities by monetization as suggested e.g., by [White et al. 1996] and [Shapiro 2001], could, theoretically, provide the complimentary information needed to consider the social and environmental consequences of a decision. This would lead to a full aggregation of the three pillars of LCM[25] in monetary units. Though such an aggregation might be desirable from a scientific point of view, it cannot be aligned to the goals of LCM, i.e. making life cycle approaches transparent, understandable, operational, and readily applicable in routine decision-making (see also the related discussion in Section 3.2.2. This is relevant not only in SMEs and in emerging regions, but also for multinational companies [Schmidt and Sullivan 2002], since it would drastically increase the complexity of the analyses and introduces additional value choices and major methodological problems of other disciplines as e.g. macroeconomic cost-benefit-analysis (for a an extensive elaboration on the shortfalls of using monetization methods for external effects see [Ackerman and Heinzerling 2004])

Concluding the discussion of the question 'what to include in LCC within LCM?', it seems appropriate to base LCC, as long as it framed by independent other assessments such as LCA, on the assumption of a primarily unregulated market (see

[24] A simple, though relevant, example is the cost, to the user, of cigarettes. Clearly, the high taxes contribute to the social and environmental overhead of smoking. However, the price of a box of cigarettes, which typically is 4 € in Europe, is a lucrative tax means which may over- or underestimate the actual externalities. If these taxes are fair, from a public health and environmental perspectives then the externalities are built in. If they are unfair, then externalities can be either unaccounted for, or double counted.

[25] Environmental, economic, and social issues form the three pillars.

above), even if this includes some double counting for the external affects actually internalized via taxes or subsidies and introduces additional uncertainties.

3.3.2 The Scope and Definition of LCC

The elaborations in the previous Section lead to the conclusion that generally only internal and internalized costs, though along the life cycle, should be accounted for in LCC within LCM (the ‘environmental LCC’ introduced in Section 3.2.2). Exceptions are cases where externalities occur that are shown, based on preliminary or prior analyses, to introduce significant (potential) costs in the future due to internalization via regulatory measures (e.g., anticipated CO_2 taxes, renewable energy subsidies). The bold arrows in Figure 3-2 schematize such an example wherein a selected number of externalities have to be considered in the LCC in order to include these risks.

Building on this approximation of a rather free (unregulated) market economy, one can **define LCC as an assessment of all costs associated with the life cycle of a product that are directly covered by the any one or more of the actors in the product life cycle (supplier, producer, user/consumer, EOL-actor), with complimentary inclusion of externalities that are anticipated to be internalized in the decision-relevant future** ([Rebitzer and Hunkeler 2003], modified on the basis of the definition of [Blanchard and Fabrycky 1998]. In this sense, LCC evaluates the feasibility of an option with good environmental and social performance. In other words, if several options for managing the life cycle of a product are compared and one option is preferable due to environmental and social benefits, this option cannot be unsustainable in the economic sense, as long as someone in the economy produces and markets the product with success. This also implies that life cycle costing, without additional assessments, cannot serve as a sole indicator for good (sustainable) LCM practice, unless there is a validated correlation of low life cycle costs to low environmental and social impacts for specific products or product groups (for related arguments on sustainability see [Dyllick and Hockerts 2002]).

The preceding definition, therefore, defines LCC within LCM as a method which accounts for only those externalities, above a threshold (i.e., they are significant to the decision), that are anticipated to become internal costs. In simple words, LCC in LCM covers ‘real-world’ money flows that are associated with the life cycle of a product. These flows can occur in the past, present, or future, depending of the goal and scope of the LCC analysis.

The economic pillar of LCM often has a comparative nature, which means that only those costs which differ between alternatives are taken into account [Rebitzer 2004]. Therefore, comparisons and cost differences are frequently the main focus, rather than absolute and detailed costs figures[26]. LCC does not resemble a financial accounting method, but a tool for cost management or management accounting along the life cycle. On the other hand, LCC can also be one component in environmental management accounting, when the product perspective is addressed [Bennett and James 1998]. There is also the concept of Total Cost of Ownership (TCO), which can be seen as a specific case of life cycle costing, where the assessment takes the perspective of the product user/consumer (for a discussion of the perspectives in LCC see Section 3.3.4).

3.3.3 Limitations of LCC

An important qualification to the prior definition (see Section 3.3.2) is that LCC is not a method for financial accounting. Rather, it is a cost management method with the goal of estimating the costs associated with the existence of a product, as LCA is not a method for environmental accounting of the environmental impacts of a specific industrial site or operation. In addition, LCC, just like LCA is a method for relative comparisons in the light of a certain decision. Therefore, decision-relevant differences are of importance and the absolute cost or environmental impacts, respectively, are of less concern. The goal is to improve a good, process, or service in the sense of a continuous improvement and/or to benchmark it against alternative options, where only the differences are important. In consequence, LCC is not concerned with the allocation of overhead costs as long as they are not influenced by the decision or they do not differ from one option to the other. As a result, LCC is also not a substitute for cost management methods that aim to achieve a better cost allocation such as activity based costing (ABC). However, if one wants to better analyze the life cycle costs of a product in detail in order to identify cost-drivers and trade-offs for decisions within the life cycle, then existing approaches such as activity based costing can be utilized as one component of LCC.

Returning to the issue of taxes and subsidies, partial accounting (i.e. excluding identical impacts or costs for alternatives) will not create an invalid result, in LCA or LCC, nor in their complimentary application in LCM, if the inventory or financial data included contain the majority of burdens, whether environmental or economic. Furthermore, the assessed product system must have the same boundaries for both metrics (see Section 3.3.1). Last, but not least, it has to be stressed LCC is not suitable as a sole indicator for sustainability, since it only represents the economic dimension, which is not sufficient as a stand-alone assessment (see Section 3.3.2 for the underlying reasoning).

[26] However, this does not mean that overall costs should be neglected, since they are the basis for the overall profitability of an activity. However, this is rather an issue of management accounting, but of life cycle costing as presented here. On the other hand, if desired and relevant for the specific application, LCC also allows to include all costs related to a product.

3.3.4 Perspectives in LCC

One heavily debated issue in LCC is the question whose costs one is accounting for. Are the costs of the user/consumer, of the producer, or the waste management operator, to give some examples, the relevant ones? This is caused by existence of value-added and margins, which have no counterparts in environmental or social assessments. Therefore, the cost of one actor (e.g., the consumer, who buys a product) is the revenue for another one (here for the product manufacturer, if one neglects the trade sector for reasons of simplicity). This also has consequences for the necessary level of detail. If the perspective of the assessment is that of the user/consumer (see c) in Figure 3-3), the costs within the boundaries of the other organizations/actors can be viewed as a black box, without requiring any differentiation. Of specific interest, however, are the specific costs and revenues associated with the use of the product (e.g., introductory and energy costs, maintenance, re-selling of product). On the other hand, if a manufacturer seeks to optimize the life cycle costs, the detailed process costs that can be allocated to a product within the company are the major focus (e.g., aided by activity-based-costing as outlined in Section 3.3.3). In this case the other cost elements in the life cycle require less detail (see a) in Figure 3-3). Part b) in Figure 3-3 represents a case where the level of detail within different actors/organizations is important. This is the case, if e.g. the supply chain is integrated by acquisition of the supplier or by supply chain coordination efforts [Seuring 2002]. Detailed elaborations on such approaches for supply chain coordination can be found e.g. in [Seuring et al. 2003]. The notion of value added requires one to consider both costs and revenues in each stage in LCC (see also Figure 3-2).

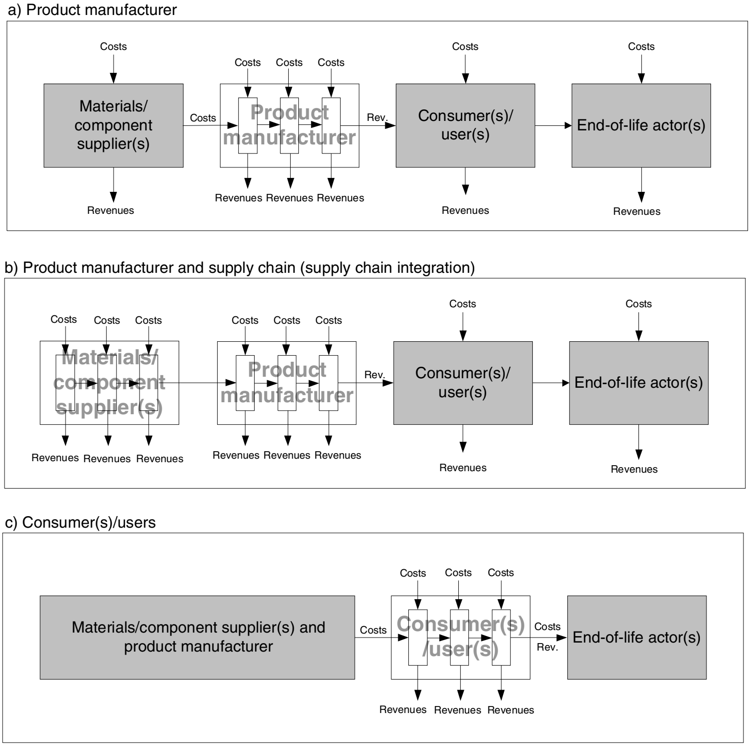

Figure 3-3: Different Perspectives in LCC (non-exhaustive examples)

3.3.5 Relation of LCC to LCA

If one relates or compares LCC to/with results of LCA, and this is a strong recommendation of many (see e.g., [Huppes 2003, Klöpffer 2003, Norris 2001; Rebitzer 2002]) and necessary for the application of LCC in LCM, then it must be based on the physical life cycle of the good or service. This implies the inclusion of, specifically, a product's material, energy and service flows from acquisition through production, transport, use, disposal, and for very durable installations such as nuclear reactors, dismantling and long term disposal. In addition to the costs caused by physical processes and their associated material and energy flows, expenses such as labor costs or costs for utilizing knowledge (e.g., patents), research and development, transaction costs (e.g., information flows), as well as marketing expenses have to be considered [Rebitzer et al. 2003a]. For example, all costs for research and development, which are not directly linked to material flows, have to be integrated.

3.3.6 Concluding the Framework Discussion

Life cycle costing, to be used in LCM, must be based on a systematic analysis that is complimentary to and consistent with parallel environmental and social assessments. In this way it serves as an efficient measuring instrument for estimating the economic feasibility of changes required to move towards sustainable development. It may be based on life cycle inventory methods (see Section 3.4), though not without inclusion of some aspects traditionally not dealt with in LCA (or assumed to be of zero impact) and addressing issues of data compatibility in regards to level of aggregation and time dependency. Full aggregation of all internal costs and externalities, within LCC, though perhaps desirable for some, seems to be outside the goals and scope of LCM, due to the practical problems involved in doing such analyses and the lacking acceptance[27]. As noted, those externalities clearly above a given threshold, which itself is a controversial issue, and which are anticipated to be internalized, should be included.

In the following a specific methodological approach, where life cycle costing is based on the life cycle inventory of LCA and the corresponding product system, is introduced. This method is the first method which consequently exploits LCI data and the systems analysis approach of LCA in order to establish the economic pillar of LCM.

3.4 Methodology of Life Cycle Inventory Based LCC

In the following a methodology is introduced which bases life cycle costing on the life cycle inventory phase of LCA and models life cycle costs for a product system that is equivalent to the product system in LCA. The central idea is to utilize the system modeling and the data of a life cycle inventory for cost calculations. Since the application is within LCM the view is predominantly an industrial/corporate one aimed at internal decision making.

3.4.1 Goal of Methodological Developments

Figure 3-4 illustrates the principal goal of life cycle costing within LCM. The goal is to reduce costs along with environmental impacts and adverse social implications or to improve at least one of the dimensions of sustainable development without causing trade-offs in any of the other ones. One can argue about trade-offs where one dimension is improved and another one is worsened, though this is usually not acceptable in the corporate context due to external and internal regulations and

[27] The aggregation of LCC and LCA to one indicator via monetization methods, with its potential benefits and drawbacks, is another issue and not discussed in this thesis.

standards (such as e.g., [ISO 14001: 2004]) and the strive for continuous improvement, which is relevant for all three dimensions.

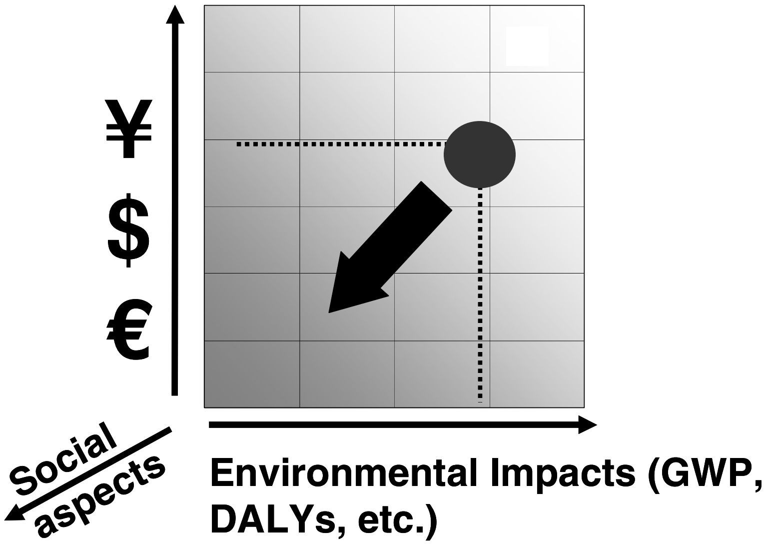

Figure 3-4: Goal of LCC activities in LCM

Therefore, a basic assumption is that a baseline situation exists that should be improved through changes within one product system or through switching to another option which fulfills the same functions as the baseline product. In such situations, a need to analyze trade-offs between the three dimensions is not necessary. In addition, methods and procedures to aggregate the dimensions are not needed in such cases, avoiding complex analyses with high uncertainties and value-laden choices.

However, in cases where there is no baseline scenario, as e.g. in the selection process of a supplier for a given material for a completely new product, trade-offs between the dimensions can become relevant. While the analysis of such trade-offs can be of high importance, it is not within the scope of the LCI-based LCC methodology. Instead other methods such a multi-criteria analysis or panel methods can be used to improve decision support on top of and in addition to the separate environmental, social, and economic assessments. An example on how these three assessments can be viewed within an holistic assessment of LCM is given in Figure 3-4.

LCC as defined here deals with an approach to estimate the economic dimension within LCM. As for the environmental assessment it is of utmost importance to provide an assessment that can be quantified and thus be used for measuring progress. Without such measuring instruments, aspects of sustainability can not be managed and thus improved ('only things that can be measured can be managed'). It is assumed that the environmental dimension is covered by LCA methods and the social aspects by other approaches, the latter being in very early stages of development [Klöpffer 2003].

It has to be noted that the LCI-based LCC methodology is meant to be used for rough cost estimations in, for example, product development or marketing studies. Due to its comparative and systemic nature it does not resemble a method to replace traditional detailed financial cost accounting or cost management practices. It is rather a method to estimate decision relevant differences between alternatives or improvement potentials within one life cycle. One can also say in LCA terminology that the LCC method presented herein follows in principal the consequential approach and is not an attributional method (for an introduction and detailed discussion of these approaches see [Rebitzer et al. 2004a] and [Ekvall and Weidema 2004], respectively). The general limitations of LCC, which are discussed in Section 3.3.3, apply.

In general, LCI based LCC aims at:

- comparing life cycle costs of alternatives,
- detecting direct and indirect (hidden) cost drivers,
- recording the improvements made by a firm in regards to a given product (sustainability reporting), and
- estimating improvements of planned product changes, including process changes within a life cycle, or innovations.
- identifying win-win situations and trade-offs in the life cycle of a product, once it is combined with LCA

For all these goals, the definition of the functional unit is of essential importance. The functional unit of the LCC analysis has to be identical to that of LCA, if both address the same study object.

3.4.2 System Boundaries and Scope

3.4.2.1 System Boundaries

As explained in Section 3.1, in LCC the term life cycle has to be seen analogously to the physical life cycle for a functional unit as in LCA. However, while the latter commonly includes the phases production (from raw materials extraction to manufacturing), use/consumption, end-of-life (from ‘cradle to grave’), the life cycle in LCC starts even earlier since it also has to include the phase of research and development (R&D). This is no fundamental difference to the physical life cycle of LCA since it also includes this additional stage in principal. However, in many cases it is only relevant from the economic perspective, but not from an environmental one and thus usually neglected in LCA (exceptions could be e.g. cases from the pharmaceutical sector).

It is plausible to assume that resources consumed and substances emitted during the R&D phase usually do not have any significant impact on the environmental performance of a industrial (mass) product, because they can be allocated to a high quantity of products. In addition, the absolute material- and energy flows originating in R&D are rather small, since this mainly involves thought and calculation processes as well as laboratory and testing work, but no large production volumes. Therefore, one could argue that R&D is also part of LCA, but usually (implicitly) not included, because its direct impact can be neglected (contrary to the influence of the R&D phase for the environmental performance of the other life cycle phases, see Figure 3-1).

Other elements that are usually not included in LCA, such as for instance marketing activities, can also be consistently included in the physical life cycle with the same rational as the R&D phase. They can be viewed as part of the production phase that is neglected in LCA due to the normally irrelevant influence. However, if for instance a marketing campaign could cause relevant environmental impacts, this should also be within the system boundaries of LCA.

To conclude the discussion of additional elements in LCC compared to the life cycle in LCA, one can say that additional elements which are of interest from the economic, but not the environmental perspective shall be included without violating the framework condition that the boundaries of LCA and LCC should be equivalent (see Section 3.3.1). The same is true for a specific assessment of the environmental and economic implications of a decision. If selected parts of the system are not taken into account in the economic system (for the differentiation between the environmental and economic systems and their relations to the system boundaries see Section 3.3.2), because they are known to be insignificant, they can still be included in the environmental assessment and vice versa.

One can say that the assessment system (environmental or economic) and the addressed scope (what environmental or economic impacts to include, see below) can be different, but that the system boundaries referring to the product system model (compare the ‘macro boundaries’ defined in Section 2.2.3) have to be equivalent.

The resulting concept of LCC, in the simplified form with one product manufacturer and one product user, is illustrated in Figure 3-5. This figure shows the product manufacturer and the product user (consumer) as the central actors in the life cycle. These actors are the driving force that a product exists at all, the consumer being the one who seeks to fulfill a need or desire and the manufacturer, who offers a suitable product and who sometimes also creates a need or desire of a product via marketing. Therefore, these two actors are both interesting in the life cycle performance, other additional actors such as those dealing with end-of-life activities only have a secondary function, delivering a service that either the manufacturer or the consumer is asking for. In addition, in LCA terminology the functional unit in LCA and LCC is always seen from the view of the consumer, while the manufacturer delivers the reference flow. This also illustrates that this 'LCA terminology' (see [ISO 14040: 1997; ISO 14041: 1998]) can be directly transferred to LCI-based LCC. If the utility provided by the functional unit is owned by the product user, the LCC approach also resembles the total cost of ownership approach (TCO) (see Section 3.3.2).

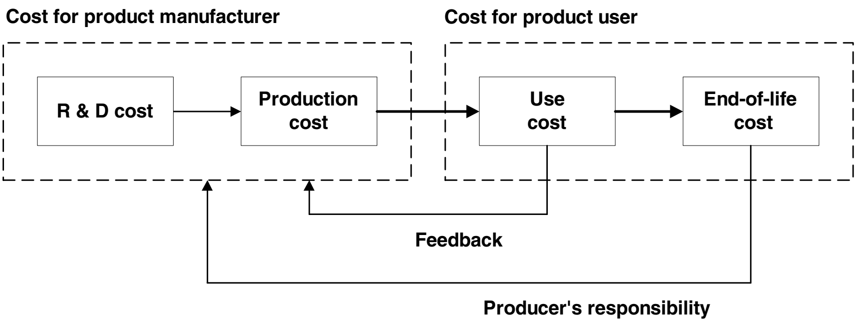

Figure 3-5: Life cycle costing concept [Rebitzer 2002]

3.4.2.2 Scope

Obviously, the scope of LCC has to differ from LCA, since the costs rather than environmental impacts are of interest. However, also here connections and overlaps exists.

Table 3-1 shows the most relevant direct cost elements and how they are connected to elements of LCA. Those costing aspects that can be directly derived by from an LCI by attributing material and energy flows to their economic value are written in ***bold italics***. The life cycle inventory of an LCA provides the quantities of these flows and the costs can be obtained by multiplying these flows with the respective company costs or market prices (e.g., materials purchasing). Those costing aspects that are written in *italics (not bold)* can be derived in part or indirectly from the information contained in an LCI. For these aspects additional information as for instance the labor requirements for the operation of a certain process or the machine costs per hour have to be gathered. If this is carried out concurrently to the establishment of the LCI model, minimal additional effort is required, since all processes are studied and analyzed in depth for the LCI. Only the costs associated with research and development (R&D) of the product cannot generally be derived from the LCA model, if the R&D phase is excluded in LCA (see Section 3.4.2.1). These then have to be determined separately.

The LCI and the product system model are a good basis for the direct cost, as the amounts of material or energy used can be multiplied with a cost or price figure to yield a cost number. Indirect costs (as e.g., overhead costs) are much more difficult to capture. Since the developed methodology has a comparative nature, such indirect costs are only within the scope of the LCC method, if they differ between the alternatives studied. If relevant differences can be expected either in overhead or in total output of functional units, the product system should be expanded in order to also include the relevant processes (e.g. marketing activities). For this existing methods, such as activity based costing (ABC, see below) can be utilized in a synergistic manner, since the product perspective also helps to break down some often only indirectly accounted costs (as e.g. waste management costs) and converts them to direct costs.

One can conclude that all those processes within the product system that are covered by the LCA are a good basis for deriving the associated costs directly (for material and energy flows) or indirectly (e.g. for labor costs and costs for capital equipment). In addition only those costs that occur in physical or immaterial processes and that are expected to be different between the compared alternatives (as some overhead costs, see above), but are not deemed relevant for the assessment of the environmental impacts have to be added.

Table 3-1: Connection of LCA elements with LCC elements (modified from [Rebitzer 2002])[28]

	Cost for Product Manufacturer	Cost for Product User
R&D	Market research Development costs	
Production	***Materials*** ***Energy*** *Machines, plants* *Labor* ***Waste management*** ***Emission controls*** ***Transports*** Marketing activities	
Use	*Maintenance/ repair (warranty)* *Liability* *Infrastructure*	***Transports*** *Storage* ***Materials*** ***Energy*** *Maintenance/repair* *Infrastructure*
End-of-Life	***Waste collection, disassembly/recycl ing/disposal if take back schemes etc. exist***	***Waste collection*** ***Disassembly/re-cycling/disposal***

The aforementioned links between the product system of LCA with processes and the corresponding material- and energy flows as well as other exchanges (e.g. land use) are the fundamental basis for the LCI based LCC methodology.

For the calculation of the life cycle costs the same concepts apply whether the product resembles a material good or a service, there is no principal methodological difference.

[28] Those costing aspects that can be directly derived by from an LCI by attributing material and energy flows to their economic value are written in ***bold italics***. Those costing aspects that are written in *italics (not bold)* can be derived in part or indirectly from the information contained in an LCI

3.4.3 Calculating Life Cycle Costs

3.4.3.1 General Procedure

As in LCA, in LCI-based LCC the calculations are based on data that are collected per unit process. As a unit process is defined as the single process for which data are collected [ISO 14040: 1997], the level of aggregation can vary highly depending on data availability and the goal and scope of the specific assessment.

Similar to the discussion on the differences between the environmental and economic system and the boundaries of the product system under study (see Section 3.3.1), different levels of aggregation can occur in LCA and LCC, also if both assessments are carried out concurrently for the same product. The desired or necessary level of aggregation in LCC depends, aside from the situation of data availability, on the perspective from which the study is carried out (for a discussion of possible perspectives see Section 3.3.4). This implies that different unit processes can be used as long as they are compatible to each other (e.g. the material price reflecting the complete upstream processes, which consist of many unit processes in the LCI, but only one sub-system, the cradle-to-gate costs, in LCC). Here, a sub-system denotes a part of the product system model that comprises several unit processes.

Once the costs for materials and energy and the operation of the processes (materials/chemicals production, component/product manufacturing, transports, use, reuse/recycling, waste management, etc.) and additional costs with no equivalents in LCA (as e.g. labor costs) have been determined, they are aggregated for the quantity of product (reference flow, derived from the functional unit [ISO 14040: 1997] of the LCA) to be assessed. An example is the aggregation of costs for the treatment of the average amount of municipal wastewater per person and year in a given region (see the case study on waste water treatment in Section 4.2). For costs or revenues that occur in the middle or long term future (e.g. from recycling of an automobile after its useful life 12 years into the future, see the automotive case study in Section 4.1.3) discounting applies.

In addition to defining the reference flow according to the functional unit, which has to be the same as in the underlying LCA model, a cost perspective corresponding to the actor and decision to be supported has to be chosen. This is necessary, because the prices are different depending if they are producer prices (e.g., the cost of raw materials for the manufacturing of an automobile) or consumer prices (e.g., the cost for purchasing a manufactured product as an automobile) due to the value added throughout the supply chain.

If there are high uncertainties in respect to expected costs, specifically in the future, or in regards to the discounting rate to be chosen, it is advisable to focus on those costs and assumptions that are different in the alternatives studied and to employ sensitivity analysis on a comparative basis. With such procedures the uncertainty within a comparison of alternatives can be minimized effectively, without causing significant additional efforts for the data compilation process. Only if such an analysis yields high sensitivity of the results to certain data points, specific efforts need to be undertaken to validate or improve their quality.

3.4.3.2 Specific Methodological Issues: Similarities and Differences between LCA and LCC

Definition of Functional Unit and Reference Flows

For LCC based on LCI the functional unit has to be the same as in the underlying LCA, since it builds on the same product system providing the same function. While the magnitude of the functional unit might be chosen arbitrarily both in LCA and LCC, it is important to use the same unit (e.g. packaging for the provision of one liter of beverage vs. packaging for the provision of the total quantity of beverages consumed by a given population). Therefore one common reference is necessary in order to allow for an appropriate interpretation of the results. In consequence, also the reference flows have to be identical, may they resemble physical material- or energy flows or immaterial services.

Definition of Unit processes, Data Aggregation, and Data Availability

Unit processes and thus the level of data aggregation can in principal be regarded in LCC as in LCA, i.e. that the data can be collected for the same units. However, in many cases – at least when a detailed assessment of all single technological processes is not necessary – the price for a given process input (e.g. material, component, service) can serve as a measure for the aggregated upstream costs. In such a case the detailed costs and added values of the upstream costs need not to be known. This is a fundamental difference to LCA, where data on the complete upstream processes are necessary for the calculation of the aggregated value. Therefore, the unit processes do not have to be the same for LCC as for the underlying LCI, compatible aggregates are often sufficient. On the other hand, if there are cost data available for different unit processes within a product system, they cannot be simply added up like the material- and energy flows and/or corresponding impacts in LCA. The existence of value added has to be taken into account in addition to the costs. A recommendation is to use prices for those inputs purchased or outputs for further treatment that are out of the influence of the perspective (see Section 3.3.4) of interest. Internally, if the aim is to identify costs drivers within one organization, costs are usually the better choice. Such choices also reflect the data availabilities: costs can often only be obtained from the processes internal to an organization or cost unit, but prices are relatively easily available also for external processes and flows.

In the context of data availability it is important to realize that costs and prices can vary highly over time and from case to case, depending on the market elasticities, new developments, market powers, transaction costs, etc. The variance of costs and prices is often much higher than variations in technologies reflected in different LCI data. Therefore, much care has to be taken when collecting and using generic cost or price data. Using specific data for the specific object under study, considering the relevant market situations, is highly preferable.

When collecting data for a combined LCA and LCC analysis care should also be taken that they can be related from a temporal point of view, i.e. the costing data should reflect costs and prices of those processes, materials, energies, technologies, etc. that are represented in the LCA product system model.

Discounting

Discounting for costs or revenues that occur in the future is a heavily debated subject in LCC (see [Huppes 2004]). Discounting is relevant of long lasting product life cycles with relevant costs during the use phase (e.g. energy consumption, maintenance) or at the end of life (costs for waste treatment/recycling and/or revenues for secondary materials or reusable products).

There is consensus that a purely scientific solution for selecting the most suitable discount rate does not exist, since it involves individual time preferences and value choices. Results from analyzing existing LCC case studies of different types [Ciroth and James 2004] as well as policy recommendations [EC 2003a] lead to the conclusion that generally accepted discount rates for LCM are in the range of 0 – 10%. Such a range of discount rates is also consistent with nationally or supra-nationally controlled discount rates as for instance the US Federal Reserve System (varying rates of 0.50 – 7.00% in the period from 1983 until 2004), the Deutsche Bundesbank (varying rates of 2.50 – 8.50% in the period from 1948 to 1996), or the European Central Bank (varying rates of 2.00 – 4.75% from 1999 – 2003) [Leitzinsen 2004].

With this range as a background an LCC practitioner should ask the question if the time dimension can play a relevant role in the life cycle. If this is not the case, for instance if there are no significant use or end-of-life costs or, or if the use phase is very short, then 0% discounting can be applied. However, as soon as there are opportunity costs (as e.g. involved in the question if costs are covered in the present vs. the future) the renouncement of lost benefits (such as interest) should be considered. If the product life cycle spans a longer period such as one year or more and if relevant costs or revenues occur during use or end-of-life, it generally does not seem appropriate to apply a rate of 0% discounting. As a rule of thumb, it can be recommended to use a 5% discount rate as a default, and to employ a sensitivity analysis (see Section 3.4.5), varying the rate from 0 – 10% in order to test the robustness of the results. As an example, the proposal for a Directive of the European Parliament on establishing a framework for the setting of Eco-design requirements for Energy-Using Products (EuP) prescribes the above mentioned rate of 5% [EC 2003a].

Another option to tackle the issue of discounting has been proposed by [Hunkeler and Swarr 2004]. For industrial applications, they propose to use an internal corporate discount rate in the range between 5% and 15% for the first 5 to 10 years of a product life cycle, but then switch to a rate in the area of 0.001% per annum (exact rates and time spans may vary, depending on the product). This concept reflects both industrial reality, while also integrating the long term view of sustainability as proposed by [WCED 1987]. However, in any case where the overall results and recommendations highly depend on the discount rate, extreme care has to be taken in the interpretation phase (see Section 3.4.5).

While discounting in LCC is generally recommended where relevant and in relation to the goal of the study, this has to be seen completely disconnected from the discussion if environmental impacts occurring in the future should be discounted (for the discussion on discounting environmental impacts see [Hellweg et al. 2003]). There is no need to align discounting rates of economic costs and revenues with those for environmental impacts even for the same study, since these are different issues, one dealing with the evaluation of future environmental impacts and the other with the future costs and revenues from the perspective of the present Generally, discounting impacts in LCA is neither widely applied, resulting in an implicit discount rate of 0%, nor generally recommended [Hellweg et al. 2003], at least for damages to objects with an intrinsic value.

Allocation

Allocation is a heavily debated subject in LCA. In LCC the challenge has a different nature, since co-product and recycling allocation can be directly correlated based on market prices. It is obvious to use, in LCA terminology, economic allocation [ISO 14041: 1998] due to the economic nature of the assessment. However, allocation of indirect cost such as relevant overheads (see the short discussion in Section 3.3.3) or allocation of costs caused by different components within one product are important methodological challenges. The issue of overhead allocation is subject to a complete discipline in management accounting and can be summarized under methods such as activity based costing (ABC). In this context LCA based LCC can provide improvements since more costs can directly be allocated to the single processes than usually done in corporate cost management, which is often organized around costs centers without the product perspective and the related process focus in mind. Therefore, LCA-based LCC can minimize overheads that cannot directly be assigned to single processes and the related material or energy flows.

The systems view with the focus on processes and products allocates more direct costs by better identifying and eventually transferring indirect costs. An example for this transfer is the cost for the management of production waste, which is often part of the overhead costs of a company, but can be converted into direct costs by the developed LCC approach. In addition, only those overhead costs that are different from one product to another are of interest, costs that are not product specific can be neglected since this method follows a consequential approach and has a comparative nature such as LCA (see Section 3.4.1).

The question of allocating different parts of components (or chemicals etc.) of a product to costs that can only be directly associated with the complete product has to be solved on a case by case basis. An example is the allocation of the weights of different components to the cost of using an automobile. In such cases, where economic allocation can not be applied, the allocation mechanisms used for LCA should be used, where systems expansion is the preferred method of choice [ISO 14041: 1998]. An example for such an allocation procedure is presented in the automotive case study in Section 4.1.3).

3.4.4 Data Compilation and Aggregation

There is no generic data format for LCI-based LCC (yet) and it is questionable if the data requirements for LCC can be standardized in detail at all. Data requirements highly depend on the goal and scope of the study and cost differences are the main concern rather than absolute figures (see Section 3.4.2.2). This also means that different studies of the same object, with different goals and scopes, are usually not directly comparable to each other, as is the case for LCA studies. In addition, cost information is much more variable over time than life cycle inventory data, therefore static databases would not be very useful for LCC, while the contrary is the case for flow data of LCI unit processes (for arguments related to the latter issue see [Frischknecht and Rebitzer 2004]). However, in cases where specific data is lacking or where only a coarse generic life cycle costing analysis is the goal, prices from databases such as those from [Granta Design 2004], which provides default price ranges for material as well as manufacturing process costs, or relative cost catalogues (see e.g. [VDI 2225: 1996]) can be employed. If LCI-based LCC is applied regularly within an organization, it is advisable to build and maintain an internal database for the most relevant cost elements of the processes, materials, and energy carriers under study. For the latter case, an internal data format should be established.

The general procedure for identifying and quantifying the relevant cost data per unit process or sub-system of the product system model (see Section 3.4.3.1) and the aggregation to life cycle costs for the production, use, and end-of-life phase can be summarized as follows:

Step 1: Identification of the sub-systems or unit processes that could result in different costs or revenues (in the following steps only the term 'costs' is used, denoting both costs and revenues)

Step 2: Assignment of costs (for internal processes) or prices (for external processes) to the respective material and energy flows of the unit processes or sub-systems identified in Step 1, with the process output as reference unit (e.g. 1 kg intermediate product)

Step 3: Identification of additional cost or price effects of the unit processes or sub-systems identified in Step 1) that differ between the studied alternatives (other operating costs of the process taken into account investments, tooling, labor, etc.)

Step 4: Assignment of costs or prices to the additional process operating costs identified in Step 3, with the process output as reference unit.

Step 5: Calculation of the costs per unit process or sub-system by multiplying the costs per reference unit from Steps 2 and 4 with the absolute quantities of the process outputs for providing the reference flow(s) of the complete product system.

Step 6: Aggregation of the costs of all unit process or sub-system (from step 5) over the complete life cycle, including the value added.

Theoretically, the data needed in Steps 2 and 4 could be retrieved to a large extent from internal enterprise resource planning (ERP) software systems by coupling these systems with LCI modeling tools (such as e.g. SimaPro [PRé 2004] or GaBi [IKP/PE 2004]), as suggested for environmental flow data already several years ago (see e.g. [Krcmar 1999; Möller 2000]) and since implemented by some corporations (see e.g. [Gabriel et al. 2003]). In the future, if LCI-based LCC methods such as presented in this thesis, become more standard applications, such a coupling of the IT systems should be aimed at.

3.4.5 Interpretation of LCC Results

In LCA the interpretation phase is defined as a “systematic procedure to identify, qualify, check and evaluate information from the results of the LCI and/or LCIA of a product system, and to present them in order to meet the requirements of the application as described in the goal and scope of the study” [ISO 14043: 2000]. This definition can directly transferred to LCI-based LCC, by replacing “of the LCI and/or LCIA” by “of the LCC analysis”.

The interpretation phase is very individual to a study, involving checks of completeness, consistency, and sensitivity [ISO 14043: 2000] in order to arrive at findings or recommendations relative to the goal(s). Methods of uncertainty analyses, apart from sensitivity analysis, might also be one element of interpretation.

As in LCA, the aim of the interpretation in LCI-based LCC is to evaluate the results obtained in the LCC, taking into account all previous steps. Uncertainty and sensitivity analysis, where appropriate and necessary, should focus on those data which might contain the highest uncertainties due to the involvement of coarse assumptions, expected variations (e.g. of very elastic market prices or because of time-dependency of the data for a life cycle that spans several years), or value choices. The latter are a factor when discounting of future costs and revenues is applied. As a rule of thumb, for the case of discounting, sensitivity analysis should be applied that varies the discount rate in the range between 0 and 10% (see Section 0). If necessary and desired, more sophisticated techniques for assessing uncertainty of cost and revenue input data can also be applied, as demonstrated by [Norris and Laurin 2004], who use Monte Carlo analysis for calculating cost originating from risks and liabilities.

When interpreting results of LCC, care has to be taken not to underestimate uncertainties, specifically when comparing them to potential variations in LCA. Even though LCC only works with one unit (monetary unit such as € or $) uncertainties of some costing data might be higher than for technological inventory or impact assessment data. Examples of influences that increase the uncertainties of future costs, but not necessarily those of future environmental costs include [Schmidt 2003]:

- Changes in taxation, wages, fringe benefits, etc.
- Chosen discount rates (see above)
- Changes in the market (access, competitors)
- Temporal or regional price variations due to market politics, image, trends, money exchange rates, etc.

In order to identify environmental-economic win-win situations or trade-offs, the final results of an LCC study should be analyzed together with the results of the parallel LCA study. One possibility is to plot selected LCA results (e.g. one representative or the – by the LCA interpretation identified - most important impact category) versus the LCC results (portfolio representation). If the LCA results show significant trade-offs between impact categories, then it is also possible to create several portfolios. The use of one-score, i.e. weighted, LCA results is not recommended due to the resulting loss of transparency, acceptance problems, and the requirements of [ISO 14040: 1997], which directly only focus on comparative assertions disclosed to the public, but are often also followed for other applications.

An example of such a portfolio representation is given in Section 3.4.1. It is important to note that such a portfolio only shows relative differences between the alternative products studied in the combined LCA and LCC since both assessments have a comparative nature. This is in contrast to portfolios with similar appearance, but which claim to include the economic and environmental impacts of the average good or service fulfilling the functional unit as the center of the portfolio as proposed by [Saling et al. 2002]. Therefore, the resulting portfolio herein is termed 'relative life cycle portfolio' so that it is not confused with the concept of [Saling et al. 2002]. In the future such relative life cycle portfolios should be extended to also include the third dimension of sustainability, social aspects, from a life cycle perspective. Figure 3-4 shows the proposal of such a three-dimensional portfolio.

Finally, the combined results of LCA and LCC can be used for further analyses in the context of LCM and sustainability. For instance, the normalization to comparable base-lines and the subsequent calculation of ratios or metrics can add additional insights to the questions of eco-efficiency and sustainability . Examples of such metrics, where both LCA results and results from LCI-based LCC can be employed, are the 'Return on Environment' (ROE) [Hunkeler and Biswas 2000; Hunkeler 2001] and the 'Econo-Environmental Return' (EER) [Bage and Samson 2003]. ROE calculations applications from the automotive and the aerospace sector involving the LCI-based LCC method can be found in [Rebitzer and Hunkeler 2001].

3.4.6 Conclusions

The developed LCI-based LCC method is a suitable and efficient approach for assessing the economic pillar in life cycle management (LCM) and meets the requirements originating from the elaborated LCC framework (see Section 3.3). While other approaches, that are decoupled from the product system model of LCI (purely economic LCC), seem also feasible to be applied for LCC within LCM in principal, the involved effort would be much higher than with the method presented herein and therefore contradict the general goal of LCM ('to put sustainable development into (business) practice'). This finding is also confirmed by the fact that purely economic LCC ('conventional LCC', see Section 3.2.2), although already proposed in the 1960s has only found limited applications (see Section 3.2.1). In addition, it would be very difficult to ensure consistency and compatibility between the models used for the environmental and for the economic assessment. Basing the LCC on the LCI of an LCA offers the opportunity for a wide-spread application of LCC in LCM and thus within the sustainability framework by using the data and models of LCI not only for environmental, but also for economic assessments.

As in LCA, the goal and scope definition of an LCC should very carefully consider questions such as the specific decision to be supported, decision-context (for whom), necessary level of detail, internal use within a company versus external publication, different perspectives (e.g. manufacturer's point of view, focus on supply chain or user's view), time horizons to include, the handling of uncertainties, etc. For instance, apart from the specific issues relevant for an LCC study, there are different requirements for LCC within SMEs and similar firms [Rebitzer et al. 2004a] or for multinationals [Schmidt 2003]. All these issues govern the methodological and data requirements for a study as well as the involved effort [Schaltegger 1997; Seuring 2001], which should be appropriate to the goal of the study (see above).

Using the guiding principles and the systemic nature of LCA for LCC facilitates holistic assessments and is a step towards a better integration of sustainability aspects in decision-making in corporations and other organizations. As a next step, also a similar social assessment methodology, being compatible to LCA and LCC, should be developed. Research in this area has started, though no consistent method does yet exist [Klöpffer 2003; Klöpffer 2005].

4 Analysis of Detailed Case Studies

4.1 Materials Selection for Passenger Car Front Subframe System[29]

Materials selection is a crucial step in the process of developing new components and products for various sectors, perhaps the most visual, emotionally and historically important of which involves automobiles and other means of transportation. The material selected for a specific application sets performance, manufacturing technologies, production costs, and life-span of the component, as well as affecting the environmental impacts and costs of the complete life-cycle, including recycling options. The latter issues are, increasingly, relevant, specifically for the automotive industry due to European regulations on recycling and integrated product policy as well as changing, global, customer demands. These issues can also be seen in a much broader sense, in the context of sustainable development, which encompasses economic, environmental, and social considerations as integral elements of industrial activities. These three goals can also be referred to as the “triple-bottom-line”. The following Sections elaborate on the environmental and economic assessment of two material options for a front subframe system of the current Ford Mondeo middle class passenger car. This research was carried out in cooperation with Ford Motor Company.

Section 4.1.1 summarizes a detailed life cycle assessment study of the automotive component. This study is based on work by [Rebitzer et al. 2001], and is updated in regards to detail (concerning the disaggregation of processes in the life cycle inventory) and expanded to include a more comprehensive impact assessment phase. Section 4.1.2 then shows an application of the simplification methodology elaborated in Section 2.5 to this case study. Finally, the developed LCI-based LCC methodology (see Section 3.4) is employed to estimate the various costs along the life cycle of the component, for the two material alternatives studied. The case study analysis is concluded with Section 4.1.3.4, which combines the environmental and economic assessments of the car component.

[29] Note: Section 5.1 is partly based on previous publications of the author: [Rebitzer 2002], [Rebitzer and Fleischer 2000], [Rebitzer and Schmidt 2003], [Rebitzer et al. 2001]

4.1.1 Detailed LCA

4.1.1.1 Goal and Scope Definition

Goal of the Study

The goal of this detailed LCA was to support the materials selection process for the front subframe system of the Ford Mondeo. Therefore, the LCA is meant for internal decision-support and not for comparative assertions intended to be disclosed to the public, while respecting the requirements of [ISO 14040: 1997]. More specifically, it was of interest to assess the existing material solution (glass fiber reinforced polyamide) in comparison to a potential subframe system made from hemp fiber reinforced polypropylene, which could be a material for the future given that technological problems in manufacturing can be solved and the overall performance is satisfactory (technical, economically, environmentally). The focus on these two materials is the result of a previous selection process, which looked in detail into the feasibility of different material options in regards to technical performance in use, manufacturing and recycling properties (see [Rebitzer et al. 2001]). The performance and environmental impacts of other alternatives materials for this component was either already well known (as e.g. aluminum) or judged not to meet the requirements (as e.g. magnesium or steel, see Table 4-2); therefore they are not included in this study.

The intended audience are environmental experts as well as product development engineers at Ford, or, in more general terms, persons involved in materials selection within the automotive industry. The goal was to identify relevant decision-support in the context of Design-for-Environment. Knowledge gathering related to specific resource consumptions or emissions or the identification of those production or post-consumption processes or related elementary flows with the highest contributions to the overall impact was of less interest. This is due to the fact that this audience is not involved in the production of the materials or the treatment and recycling processes at the end-of-life of the components. Specific details in regards to environmental impacts were only studied if they were identified as highly relevant to the results of the study and the subsequent decision-support.

Before a further definition of the methodological scope of the study is presented, the examined product and the material options are described in more detail in the subsequent section.

4.1.1.2 Description of the Front Subframe System and the Considered Material Solutions

Due to its integrating function (e.g., for the assembly of headlights and the engine cooling system), the design (construction/shape) of the component was predefined by the development engineers at Ford. Figure 4-1 shows the component.

Figure 4-1: Front subframe system of Ford Mondeo

The product performance in regards to mechanical and other requirements was also predefined by Ford. From these product requirements, the related material specifications and properties that have to be met were derived (for a detailed elaboration on this materials selection process see [Rebitzer et al. 2001]). Table 4-1 presents an excerpt of the product performance and related material requirements.

Table 4-1: Profile of requirements of the front subframe system (excerpt)

Product performance	**Representing material specifications**	**Requirements**
Mechanical strength (> 700 N) and stiffness (> 45 N)	Modulus in flexure	> 1.1 GPa
Resistance against hood slam impact	Impact strength	> 16.0 kJ/m^2
Mass of subframe system < 4.9 kg	Density	< 10.2 kg/dm^3

The technical requirements are interrelated and have to be seen in relationship to constructional constraints. In this case the wall thickness of the subframe system was variable in the range of 0.5 to 5.0 mm. This means for instance that the required weight of less than 4.9 kg can be achieved by a material with a density of 10 kg/dm^3 only if the material is strong enough (much more than the minimum requirement of 1.1 GPa modules in flexure) so that a wall thickness of not much more than 0.5 mm can be realized. Only those materials are suitable that meet all the requirements listed in Table 4-1 (and others) and with which the constructional constrains regarding wall thickness can be realized.

Some of the resulting materials and the corresponding constructional data are listed in Table 4-2.

Table 4-2: Identified materials for the front subframe system (excerpt)

Material for subframe system	Wall thickness	Mass
(Steel)	0.9 mm	ca. 6.8 kg
Magnesium	1.6 mm	2.55 kg
PA, glass fiber-reinforced (30%)	3.0 mm	4.37 kg
PP, hemp fiber-reinforced (70%)	3.0mm	3.54 kg

In Table 4-2 steel is listed though it does not allow the design of the component within the required maximum mass of 4.9 kg (therefore the brackets). It is nevertheless included in the table to show the performance of a standard material for this application. However, one can see that from the materials listed in Table 4-2 only the fiber reinforced thermoplastics and magnesium offer the performance needed. Therefore, steel was excluded from the environmental assessment. From the remaining three materials under consideration in this study glass fiber reinforced PA (the currently used material for the component) and magnesium are feasible materials, i.e., product performance, manufacturing, and recycling requirements can be met. A previous assessment of the magnesium component, however, revealed that the impact on global warming would be prohibitively high in this specific case [Rebitzer and Schmidt 2003], which was not acceptable to Ford. Therefore, this material was also knocked-out from the present study of the component as well.

Hemp fiber reinforced PP is potentially feasible, though there is research and development (R&D) necessary to solve the existing manufacturing problems (manufacturing by injection molding, the desired technology for this part, is currently not (yet) possible due to the high fiber content). However, to explore the potential of this material it is included in the detailed assessment, which could help to answer the question if development, business, and engineering resources should be allocated to developing/adapting manufacturing processes for this material.

In the following the results for the detailed LCA of the remaining two materials solutions for the front subframe system, glass fiber reinforced polyamide with 30 mass-% fiber content (PA-GF 30) and hemp fiber reinforced polypropylene with 70 mass-% fiber content (PP-HF 70), are presented.

4.1.1.3 Methodological Scope of the Study

Functional Unit, Basic Assumptions, and Limitations

The functional unit for the comparison is defined as one front subframe system, meeting the requirements outlined in Section 4.1.1.2, for the full service life of a passenger car. Due to the nature of the component, a non-accident related replacement cannot be accepted, therefore the component has to have a life time at least as long as the car. On the other hand, the component is usually not re-used. Therefore, the service life can be assumed to be identical to that of a passenger car. The service life of a car is defined by the distance driven during its useful life. In Europe, for LCA applications of compact class vehicles, the automotive industry has agreed on a generic service life of 150 000 km [Ridge 1998], which can be seen as conservative (low service life) for the Ford Mondeo. However, since this is a consensus value and it minimizes uncertainties (it can be seen as the minimum service life), it is used in this study. The component under study does not require any maintenance, therefore the use phase is only characterized by the fuel consumption that can be allocated to the mass of the component.

In regards to replacement from collisions, one can state that the technically comparable components from different materials fulfill equivalent specifications. Therefore, it is save to assume that the accident behavior is identical in regards to other damages as well as in regards to the need for replacement. The component has no accident related function as e.g. bumpers.

The preceding analysis and evaluation of manufacturing processes identified injection molding as the technology of choice (see [Rebitzer et al. 2001]). Therefore, this technology was used in the LCA for both alternatives, even if this process is not feasible (yet) with a natural fiber reinforced materials having 70 mass-% fibers (see Section 4.1.1.2). Not included in the analysis is the assembly of the component in the car, since this step does not differ from one material to the other and since it is reasonable to assume that the related impacts are negligible.

Transport processes between the life cycle stages, i.e. between production, use, and end-of-life are excluded from the system, since it is safe to assume that the related differences in the life cycle of the two options are negligible compared to the overall differences.

For the use phase, it was assumed that from the tailpipe emissions only CO_2 and SO_2 are affected by the weight of the components, since only these emissions can be correlated to the fuel consumption (the other regulated emissions (e.g. Euro 4 emissions are functions of the engine control, catalytic converter, etc. and not largely influenced by weight differences in one component). Therefore, the impacts from tailpipe emissions in the use phase only relate to the emission of CO_2 and SO_2, the total absolute impact of the use phase is not reflected, since it would not contribute to the goal of the study. An important assumption for the use phase is also the general affect of weight reduction on the fuel consumption of a vehicle. For this study a value of 0.35 l per 100 kg weight saving and 100 km driven distance is used. This value can be seen as a generally accepted default factor for allocating the total weight-relating fuel consumption of a gasoline passenger car (see the discussion of this factor in [Ifeu 2003]). Another recent study of the automotive industry [Schmidt et al. 2004] used 0.38 l per 100 kg weight saving and 100 km driven distance as default.

For the end-of-life phase it is assumed that both alternatives are, as part of the non-hazardous shredder-light-fraction, incinerated in a municipal (or similar) waste incinerator with energy recovery, since there is neither demand for re-use nor a realistic option for materials recycling (current situation; however, there are technologies in development, which could enable materials recycling of post-shredder waste in the future). In most European countries the costs for dismantling this part would be prohibitively higher then the materials' value. Even if some materials recycling (via recovery from the shredder-light-fraction) would be possible for the composites, it has been shown that this would not alter the results significantly (for an explanation of the low influence of plastics in the EOL phase to the life cycle impacts of vehicles see [Schmidt et al. 2004]). All processes related to the decommissioning of the car and recycling and treatment operations previous to the incineration are excluded from the assessment, since they are assumed to be identical for both material options. For reasons of simplicity, the combined collection and recovery rate (percentage of total component mass that is incinerated) was set to 100%. This is justified, since sensitivity test showed that other relevant scenarios (e.g., 80% incineration, 20% direct landfilling) do not significantly influence the end-of-life phase; further detail in this regard would not be appropriate for the goal of the study.

Reference Flows

Derived from the functional unit and based on the previous material selection process, the following reference flows for the two material alternatives result (see Section 4.1.1.2):

- Front subframe system made from PA-GF 30 with a mass of 4.37 kg
- Front subframe system made from a theoretical PP-HF 70 with a mass of 3.54 kg

System Boundaries

For this LCA, the aim was to establish a product system model with system boundaries as complete as possible as far as material and energy flows and related processes and inventory flows are concerned. For these flows, systematic cut-offs were only applied to the manufacturing of production means (machinery, infrastructure, etc.) since these are not of significance for a mass product such as a component for a Ford Mondeo and since the complete inclusion would have resulted in tremendous additional effort for the life cycle inventory analysis, which could not be justified. In addition, immaterial inputs, i.e. non-material or energy flows (i.e. services), were also cut-off and do not constitute parts of the product system. Not included in the study was also the infrastructure for the use phase (roads, etc.) since the related impacts are not influenced by the material of the front subframe system.

Concluding the discussion on system boundary delimitation, in theory, cut-offs were applied only to production means and immaterial flows, while the aim was to model the processes and flows within the system as complete and detailed (level of aggregation) as possible. For these reasons, the inventory uses to a large extend process data from [Frischknecht et al. 1996], extended by specific and additional data, which were not previously available. In practice, however, one always has to be aware of potential data gaps in every inventory; a 100% coverage of all inputs and outputs is not possible (see the discussions in Section 2.5), also since not all flows are precisely recorded and variations exist (e.g. technology, operation procedure of production processes, influence of driving style for fuel consumption in use phase, etc.). To ensure that no significant data gaps sensitivity tests were applied to processes, where data gaps might be missing. In an iterative process, the data were refined were necessary in relation to the influence on the life cycle impact assessment results (see Section 4.1.1.5).

Data Requirements

As outlined above, the goal was to have an inventory as detailed as possible, therefore high standards for data requirements were established, in accordance with the data standards that have been applied by [Frischknecht et al. 1996], the database delivering the data on energy generation and other background data. Wherever feasible, data on single elementary flows (rather than groups) were obtained. If no documented LCI data were available, physical and chemical relationships (as stoichiometric balances) were used to establish the inventory for new processes. The geographical coverage of the data is Western Europe, data stem from the mid to end 1990s, though can still be considered as valid today.

Allocation Procedures

Different allocation procedures were applied throughout the inventory analysis. Regarding the data from [Frischknecht et al. 1996], the applied allocation procedures were not changed (in consequence the allocated data were used). For the additional data collected, allocation was avoided wherever possible by systems expansion or splitting up the unit processes. In cases, where systems expansion was not possible due to the involved data needs (where it would have been necessary to compile completely new inventories for the substituted materials), economic allocation was employed.

Life Cycle Impact Assessment

For the **life cycle impact assessment (LCIA) phase** (see 4.1.1.5), the method IMPACT 2002+ [Jolliet et al. 2003] was selected, since this represents one of the scientifically most advanced and up-to-date methods, combing the advantages of both midpoint and damage LCIA approaches (for a discussion of the underlying midpoint-damage assessment framework see [Jolliet et al. 2004]). In order to crosscheck the findings with a conventional and already well established method, CML 2001 [Guinée et al. 2002], a classical midpoint approach, was also applied.

Interpretation

The **interpretation phase** (see Section 4.1.1.6) is conducted based on a comparison of the alternatives impact category by impact category. On the midpoint level this is done without normalization, while with IMPACT 2002+ also the normalized and grouped damage factors are employed. Weighting procedures across damage categories are not used.

Review Process

The goal of the study is to support internal decision support in the context of design for environment. Therefore, an external critical review was not conducted. However, all basic assumptions as well as the inventory and the life cycle impact assessment results were reviewed internally by the author and partially by experts at Ford Motor Company. These internal review refined both the scope as well as the inventory analysis of the LCA study.

4.1.1.4 System Modeling and Life Cycle Inventory Analysis

Modeling Tools and Procedures

For the modeling of the product system, an internally developed tool, based on Microsoft (MS) Access, was applied. All unit process data from [Frischknecht et al. 1996] as well as the additional, specifically collected, data (see below) are included in the MS Access database. Each unit process is defined by its product (for production processes) or input (for waste treatment and recycling processes). Queries were programmed, so that the life cycle inventory is calculated by combining the processes according to the 'sequential method', which is sometimes also referred to as the 'flow chart method' (for a description of this commonly applied calculation and representation methodology see [Heijungs and Suh 2002, pp. 100]).

For reasons of practicality, one life cycle was not modeled as a whole, but in an additive manner. This means that the following life cycle (sub-) stages were modeled separately with the MS Access tool and then aggregated in a spreadsheet model (with MS Excel) after the impact assessment step for each life cycle stage or sub-stage (for the use phase) had been carried out:

- Production of the component (cradle-to-gate)
- Precombustion of the gasoline for the use phase
- Tailpipe emissions in the use phase
- Incinceration at the end-of-life of the components

For the impact assessment also a MS Excel tool was employed, which uses MS Access tables with the life cycle inventories of the (sub-) stages as input.

Though the MS Access modeling tool allows to introduce variable cut-offs in the product system model, for this detailed assessment neither processes nor flows where explicitly cut-off, with the exception of clearly identified production means and services that are not connected by material or energy flows and other processes outside the scope of the the study (see Section 4.1.1.3).

The methodological problems related to recursive processes (e.g. electricity generation processes that have electricity as inputs) are solved by approximation. After modeling up to 30 process levels, where one level corresponds to one tier of processes, the system has been stable in all cases analyzed. This means that the LCI result of modeling up to 30 compared to 31 levels only shows an insignificant mathematical difference. However, in order to reduce the uncertainty even further, for the detailed

LCA the model included 100 levels. For the purposes of an LCA study the results of such an approximation within a flow-chart model do not differ from a mathematically exact solution via a matrix approach as described e.g. in [Heijungs and Suh 2002] and applied in more sophisticated modeling tools (one example being [Frischknecht et al. 2004a]).

The internally developed tools were used in order to have full control and transparency in the modeling of the inventory analysis and impact assessment and because specific features relevant for the analysis of simplification procedures could be easily introduced (e.g., varying the process specific cut-off as elaborated in Section 4.1.2).

Data Used for the Life Cycle Inventory Analysis

As mentioned in Section 4.1.1.3, the life cycle inventories for this LCA was built using the detailed database of [Frischknecht et al. 1996], extended by new disaggregated unit process data that were not available in the database. Specifically, detailed data were collected, based on the following sources (literature, patents, oral information, etc.) for the respective process chains:

Production of glass fibers: [Bartholomé et al. 1972; Blankenburg et al. 1978; Büchner et al. 1986; Hagen 1961; Prinz 1993; Riedel 1994; Vauck, Müller 1990; VDI 2578: 1988]

Production of hemp fibers: [Böcker 1997; Christen 1997; Hartmann and Karsten 1995; Herer 1993; Hagen 1961; Hesch et al. 1996; Mühlmeyer 1999; Nowottny 1999; [Tubach 1997; Vauck and Müller 1990]

Production of polyamide: [Allinger et at. 1980; Aylward and Findlay 1999; BASF 2001; Bayer 1999; Bayer AG 1998; Beyer and Walter 1991; Boustead 1997; De 1238000: 1980; De 1244170: 1967; De 1593767: 1971; De 2417003: 1975; Gerhartz et al. 1985; Kunststoff-Handbuch 1998; Peter and Vollhardt 1988; Streitwieser and Heathcock 1982; Vauck, Müller 1990; Weissermel and Arpe 1988]

Injection moulding of fiber reinforced plastics: [Menges 1984; Metten 1995; Michaeli 1995; Tubach 1997]

Tailpipe emissions in the use phase: stoichiometric calculations based on the carbon and sulfur content of gasoline.

Within an Microsoft (MS) Access database application (see below), the processes of all these process chains were embedded in the database of [Frischknecht et al. 1996], which already contained process data for the production of polypropylene, the production of fuels and electricity, and municipal incineration. The flow nomenclature of [Frischknecht et al. 1996] was used also for the newly compiled data. This resulted in a consistent database, which is suitable for modeling detailed life cycle inventories as required by the study.

Life Cycle Inventory Analysis Results

For this LCA no interpretation on the basis of life cycle inventory results is carried out; the interpretation focuses on the life cycle impact assessment results alone. Therefore, a complete overview of the inventory results is not included, because it would be too detailed and extensive to represent all the elementary flows considered. However, to get an impression on the level of detail and the comprehensiveness of the resulting life cycle inventories, one example is given in Appendix 12.1). There, the inventory results for the production (cradle to gate) of the front subframe system from glass fiber reinforced polyamide are listed. Radiation was excluded from the inventory, since the related values are highly uncertain and would not add additional input for this study.

4.1.1.5 Life Cycle Impact Assessment Results

As stated in the scope of the study (see 4.1.1.3) , the methods IMPACT 2002+ [Jolliet et al. 2003] and CML 2001 [Guinée et al. 2002] were used for the impact assessment. In the following the results of both methods are presented and briefly discussed. The interpretation in regards to the goal of the study follows (see Section 4.1.1.6).

IMPACT 2002+

For the present study, the midpoint categories listed in Table 4-3, grouped by damage categories, were analyzed (for a description of the categories and the grouping of midpoints into damage categories see [Jolliet et al. 2003]). Land use was not part of the impact assessment phase, though available in IMPACT 2002+ due to a lack of the corresponding and compatible exchanges in the life cycle inventory.

Table 4-3: Impact categories of IMPACT 2002+ used for this LCA study

Damage categories	**Midpoint categories (potentials)**	**Units of midpoints (category indicators)**
Human health	Human toxicity (carcinogenic and non-carcinogenic)	[kgeq Chloroethylene into air]
	Respiratory effects	[kgeq PM2.5 into air]
	Ozone layer depletion	[kgeq CFC-11 into air]
	Photochemical oxidation	[kgeq Ethylene into air]
Ecosystems quality	Aquatic ecotoxicity	[kgeq Triethylene glycol into water]
	Terrestrial ecotoxicity	[kgeq Triethylene glycol into soil]
	Terrestrical acidification and nutrification	[kgeq SO_2 into air]
Climate change	Global warming	[kgeq CO_2 into air]
Resources	Non-renewable energy	[MJ prim non-renewable energy]
	Mineral extraction	[MJ surplus energy]

Midpoint Category Indicator Results

The comparison of the two material alternatives for the automotive component per midpoint category are shown in Figure 4-2, which differentiates the results by the main life cycle stages (production, use, end-of-life).

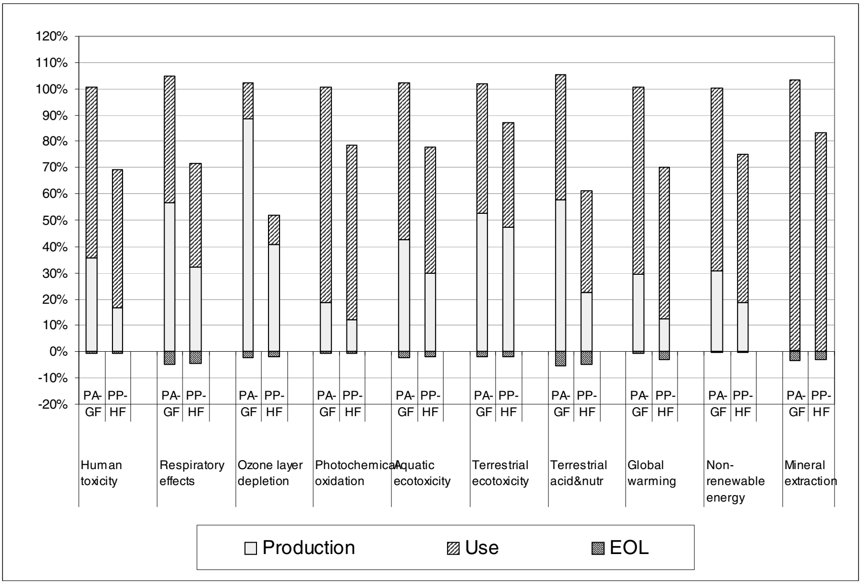

Figure 4-2: Comparison of the front subframe system from glass fiber reinforced polyamide (PA-GF) with that from theoretical hemp fiber reinforced polypropylene (PP-HF) per midpoint category (IMPACT 2002+, v2.0). The potential impacts of the complete life cycle (using the default assumptions of the goal and scope definition: 150 000 km service life, 0.35 l gasoline savings per 100 km per 100 kg weight savings) of PA-GF are set to 100%.

The results show clearly that the use phase has the highest influence for most impact categories, though in some cases the production phase is more or equally important in the chosen methodology (respiratory effects, ozone depletion, terrestrial ecotoxicity, and terrestrial acidification and nutrification). The incineration in the end-of-life phase leads to environmental benefits in all categories, though these are small compared to the impacts from other phases. The benefit is caused by the assumed substitution of energy generation by natural gas by the incineration of the plastics and combustible fibers (in the case of hemp). Another conclusion that can be drawn from Figure 4-2 is that the impacts from the PP-HF component are generally lower both in the production and the use phase, though the production does not differ very much for some impact categories, as terrestrial ecotoxicity. Therefore, the variation of the assumptions for the use phase related to the effect of weight savings on fuel consumption and service life would not change the overall results, only the absolute difference between the options would vary (e.g. lower difference if 150 000 km and 0.1 l per 100 km and 100 kg weight savings is used (as assumed for the minimal influence of the use phase by [Schmidt et al. 2004]) and greater difference if the values 250 000 km and 0.5 l per 100 km and 100 kg are used[30]).

Damage Category Indicator Results

In order to get an indication on the relevance of the different impact categories within each damage category, normalized damage category indicator results were also analyzed. The normalization factors use Western Europe as reference area and represent the average potential damages occurring in Europe per person and year (for a detailed elaboration of the underlying normalization procedure see [Humbert et al. 2004]). The resulting unit are points, where one point represents the average damage in Western Europe per person and year.

Figure 4-3 shows the results on a log-scale. If comparing the two material alternatives for the component with each other, the results are fully equivalent with those of Figure 4-2. Additional information that can be gained concerns the relevance of the different damages within each damage category. A comparison of the damages across damage categories would only be possible by applying implicit or explicit weighting procedures and is therefore not conducted for this study (see Section 4.1.1.3).

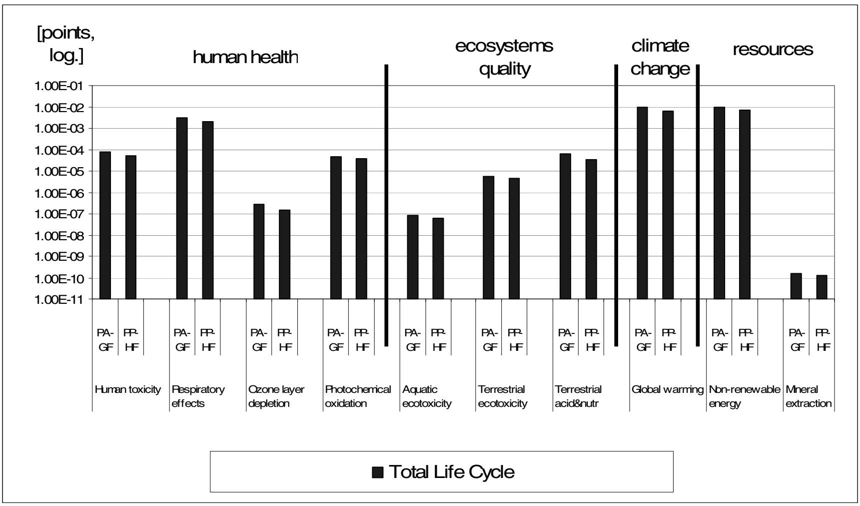

Figure 4-3: Comparison of the front subframe system from glass fiber reinforced polyamide (PA-GF) with that from theoretical hemp fiber reinforced polypropylene (PP-

[30] The value of 0.5 l is seen as the maximum value for weight related fuel savings, as used by [Schmidt et al. 2004].

HF) expressed in normalized damages per functional unit and grouped into damage categories (IMPACT 2002+, v2.0). The default assumptions of the goal and scope definition: 150 000 km service life, 0.35 l gasoline savings per 100 km per 100 kg weight savings are used.

For the chosen human health methodology, it is evident that the respiratory effects are dominating. The impacts of respiratory effects are greater by a factor of about 40 and 60 than human toxicity and photochemical oxidation, respectively. If compared to ozone layer depletion, the factor is about 10'000. Within ecosystems quality, the impacts from terrestrial acidification and nutrification are most important, being greater than terrestrial ecotoxicity and aquatic ecotoxicity by factors of about 10 and 600, respectively. For the damage category resources, only the non-renewable energy resources are of importance in this case, differing by a factor of around $5*10^7$.

CML 2001

For the assessment of the two material alternatives for the automotive component, the CML 2001 midpoint categories listed in Table 4-4 were analyzed (for a description of the categories see [Guinée et al. 2002]).

Table 4-4: Impact categories of CML 2001 used for this LCA study

Midpoint categories (potentials)	**Units of midpoints (category indicators)**
Human toxicity	[kgeq 1,4-dichlorobenzene]
Photochemical oxidation	[kgeq Ethylene]
Ozone layer depletion	[kgeq CFC-11]
Freshwater aquatic ecotoxicity	[kgeq 1,4-dichlorobenzene]
Terrestrial ecotoxicity	[kgeq 1,4-dichlorobenzene]
Eutrophication	[kgeq PO_4^{3-}]
Acidification	[kgeq SO_2]
Global warming	[kgeq CO_2]
Abiotic depletion	[kgeq antimony]

The comparison of the two material alternatives for the automotive component per midpoint category are shown in Figure 4-4, which differentiates the results by the main life cycle stages (production, use, end-of-life).

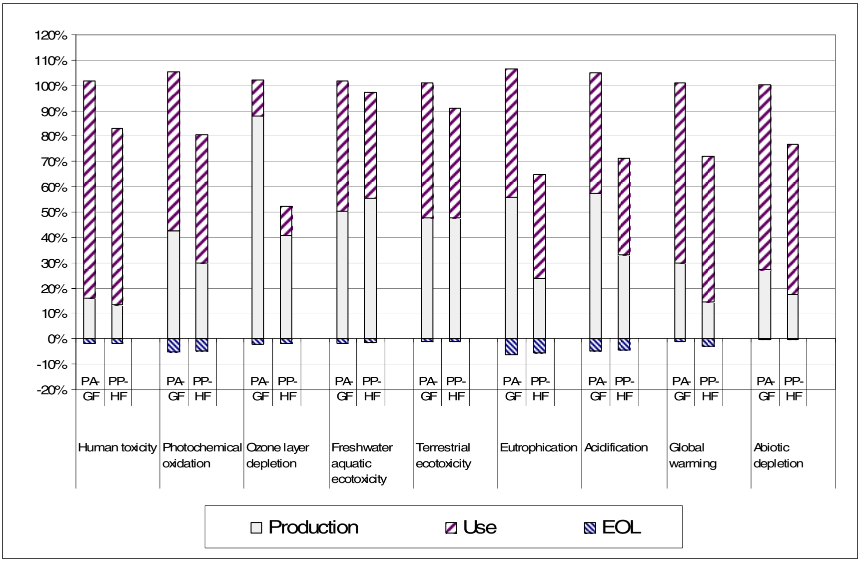

Figure 4-4: Comparison of the front subframe system from glass fiber reinforced polyamide (PA-GF) with that from a theoretical hemp fiber reinforced polypropylene (PP-HF) per midpoint category (CML 2001). The potential impacts of the complete life cycle (using the default assumptions of the goal and scope definition: 150 000 km service life, 0.35 l gasoline savings per 100 km per 100 kg weight savings) of PA-GF are set to 100%.

The category indicator results obtained via the method CML 2001, comparing the two alternatives studied, confirms the results of IMPACT 2002+ (see Figure 4-2) in relation to the goal of the study (material selection, see 0). Since the second LCIA method was only applied to check and confirm the results of IMPACT 2002+, a further analysis was not carried out.

However, it is interesting to note that with this method, the impacts from the production phase of PP-HF are not lower in all cases. For the midpoints freshwater aquatic ecotoxicity and terrestrial ecotoxicity the impacts from the production of PP-HF are higher and very similar, respectively, than those from PA-GF. Considering the range of uncertainty in ecotoxicity assessments, this difference in the results of the two methods can not be seen as a principal difference, especially since the overall result for the life cycle does not change significantly. Therefore, the results of the life cycle impact assessment with CML 2001 are consistent to those of IMPACT 2002+.

4.1.1.6 Interpretation

The goal of the LCA was to assess the existing material solution (glass fiber reinforced polyamide) in comparison to a potential subframe system made from hemp fiber reinforced polypropylene, which could be a material for the future given that technological problems in manufacturing can be solved and the overall performance is satisfactory (technical, economically, environmentally). The impact assessment clearly shows that hemp fiber-reinforced PP promises significant environmental improvement potentials.

The reduced mass of the theoretical PP-HF component, which is 19% lighter than the PA-GF component, could lead to environmental improvements in the use phase. In addition, the production of the component from this alternative material generally causes less environmental impacts, though there are – depending on the life cycle impact assessment methodology used - some exceptions (freshwater aquatic and terrestrial ecotoxicity, see Figure 4-4). While attention has to be paid to diverging impacts, specifically to the agricultural processes for the production of hemp fibers, the overall result is not affected for several reasons:

- Due to the inherent relatively high uncertainties of ecotoxicological impacts is existing LCIA methods [UNEP/SETAC 2004; Huijbregts et al. 2000], small differences should not be interpreted as significant to the overall result
- Potential differences in the production phase are offset by differences in the use phase, where the uncertainties are lower because the difference is directly related to different weights of the components ('less is better' view)
- If damage categories are analyzed, ecotoxicological impacts appear to be of of much less relevance than those on terrestrial acidification and nutrification (see Figure 4-3)

Therefore the recommendation is to invest in R&D for the manufacturing of the component from natural fiber reinforced thermoplastics. Significant environmental improvements can be expected from using hemp fiber reinforced polypropylene for such an component. Briefly, one alternative was identified for the current design and a second for further development.

4.1.2 Simplified LCA Application

4.1.2.1 Limiting System Boundaries by Baseline Approximation

For the two components of the automotive subframe system, the iterative cut-off analysis described in Section 2.5.2 has been applied for the cradle-to-gate phase (see Figure 4-5 and Figure 4-6).

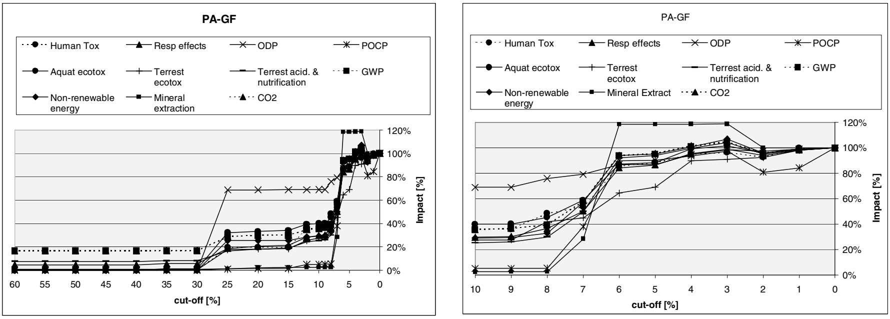

Figure 4-5: Analysis of the cut-offs for the production of the PA-GF component (intermediate impacts larger than 100% can be explained by the fact that at these cut-offs credits for co- or recycling products are not considered yet)

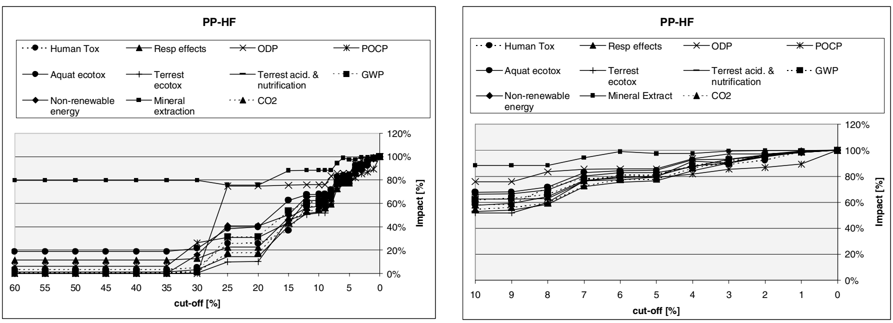

Figure 4-6: Analysis of the cut-offs for the production of the PP-HF component

These results confirm the recommendation to use a cut-off in the range of 4-5%.

4.1.3 LCI based Life Cycle Costing

For the case of the automotive subframe system, comparing two material alternatives (see Section 4.1.1.2), the LCI-based LCC method (see Section 3.4) was applied. The basis was the LCI and the underlying product system model of the LCA documented in Section 4.1.1. Since the same product system model as for the LCA was used, in the following the elaborations are limited to specific aspects of the LCC study and the results of the economic analysis. Modeling assumptions and procedures that are taken directly from the LCA are not repeated here.

4.1.3.1 Goal and Scope

The goal and scope of the LCC study for this automotive component can be seen parallel and consistent to the goal and scope of the LCA (see Section 4.1.1.1). For the application of materials selection in product design and development, the goal was to analyze the life cycle costs of the alternatives in order to compliment the results of the LCA. Therefore the system boundaries and other assumptions relevant for the physical life cycle (the life cycle according to ISO 14040: 1997) were the same as in the LCA.

Additional life cycle phases or elements such as the phase of research and development (R&D), marketing activities, etc., which might be part of LCC studies (see Section 3.4.2.1) but not of the LCA (if deemed to be of insignificant importance), were not included in the study due to the following reasons:

- It was of specific interest to determine if a new material for the application, a theoretical hemp fiber reinforced polypropylene (PP-HF), could lead to environmental and/or economic benefits compared to the currently employed material (glass fiber reinforced polyamide, PA-GF). For the current material no additional R&D effort is necessary, and for the new material the necessary effort in order to enable the manufacturing with the desired technology is unknown. Basically, the assessment should lead to answers to the question if it is worth to invest in R&D for the new material or not.

- Other cost influences along the life cycle that are not related to the product system model of the LCA can be neglected in this case, since they would be the same for both alternatives or irrelevant. For instance, marketing activities are not relevant for this part, since the selection of a material for such a component is purely based on technical performance. No secondary benefits (e.g. image for the customer) can be observed, since the end customer (car purchaser) can not identify any differences between the alternatives (the component is more or less ‘invisible’ for the customer).

Consequently, the product system model of the life cycle inventory analysis can be used without any extensions.

4.1.3.2 System Modeling

The LCC is based on the LCI and no additional extensions are necessary for the goal of the study (see the previous Section). Therefore, from a costing point of view, the model can be based on the following data (compare Section 4.1.1.3):

- Quantity and costs of materials needed for the part (reference flow of the LCI), differentiated by matrix and reinforcement of the composite materials
- Costs for manufacturing processes (production of final component from materials), including costs for material losses
- Costs for the allocated fuel used in the use phase
- Costs for the incineration of the components after shredding at the end-of-life of the car

For the calculations the data for material costs as displayed in Table 4-5 were used.

Table 4-5: Material cost data used for the study on the automotive front subframe system

Material	**Minimum [€/kg]**	**Mean [€/kg]**	**Maximum [€/kg]**	**References**
Polyamide (PA)	2.35	2.40	2.45	[European Plastics News 2/99]
Polypropylene (PP)	0.48	0.50	0.52	[Euwid Kunststoff, 1999]
Glass Fibers	2.00	2.75	3.50	Estimation based on [Kunz et al. 1997]
Hemp Fibers	0.55	0.78	1.00	[Hanfnet 1999]

For the manufacturing process (injection molding), an average value per kg of component was used, since it is the same process for both alternatives. It was assumed that the injection molding of hemp fiber reinforced polypropylene has costs identical to costs for the glass fiber reinforced polyamide. A value of 1.70 € per kg (from calculations based of data of [Granta Design 2004] was used.

There were also some confidential data of Ford available, which can not be published. However, internal comparisons with corresponding data of Ford, confirmed the general validity of the data used for material and manufacturing cost [Schmidt 2000a].

A similar procedure as for manufacturing was used for the end-of-life costs, which can be limited to the costs of incineration of the component after shredding (see above). Incineration costs vary highly from region to region in Europe, and are often politically influenced. Therefore it is difficult to find exact data. For this study, an average value of 150 € per ton of reinforced plastic was assumed, partly based on [Brandrup et al. 1996]. Due to the aforementioned reasons this value bears a high uncertainty, though does not significantly influence the results of the study (see Section 4.1.3.3). Therefore, a further analysis of these costs was not necessary for the goal and scope of the analysis.

The temporal life time of the component is identical to that of the complete car (see Section 4.1.1.3), which is assumed to be 12 years. This life time is relevant for the calculation of the discounted future costs for gasoline and end-of-life treatment. However, a variation within the probable range (about 10 to 16 years, depending on the specific market region) would not have a significant influence on the results. Therefore, no variation of the lifetime is taken into account.

Table 4-6: Important data and assumptions for the LCC model

Data/assumptions	**Default**	**Minimum influence of use phase**	**Maximum influence of use phase**
Effect of weight on gasoline consumption [l/100kg * 100km]	0.35 (see Section 4.1.1.3)	0.1 [Schmidt et al. 2004]	0.5 [Schmidt et al. 2004]
Price of gasoline consumption [€/l]	1.0 (based on prices in Europe in 2001)	0.5 (estimation for low price regions)	1.5 (estimation for possible prices in future)
Service life of car [km]	150 000 (see Section 4.1.1.3)	-	250 000 (estimation)
Discount rate	5% [EC 2003a]	10% (see Section 3.4.3.2)	0% (see Section 3.4.3.2)

The default values for the service life of the car and the weight dependent fuel consumption are identical with those used for the LCA modeling and therefore correspond to the LCA results presented in Section 4.1.1.5. While for the LCA a sensitivity analysis was not necessary, minimum and maximum values were used in addition for the analysis of the life cycle costs (see Section 4.1.3.3) in order to show the influence of data and assumptions (such as the discount rate) where there is no scientific agreement due to involved value choices and/or different use scenarios.

4.1.3.3 Results

Figure 4-7 shows the default scenario, i.e. the results of the LCC calculations using the default values for manufacturing and end-of-life costs as well as the default data from Table 4-6. For the material prices data ranges were analyzed (min, mean, max according to Table 4-5).

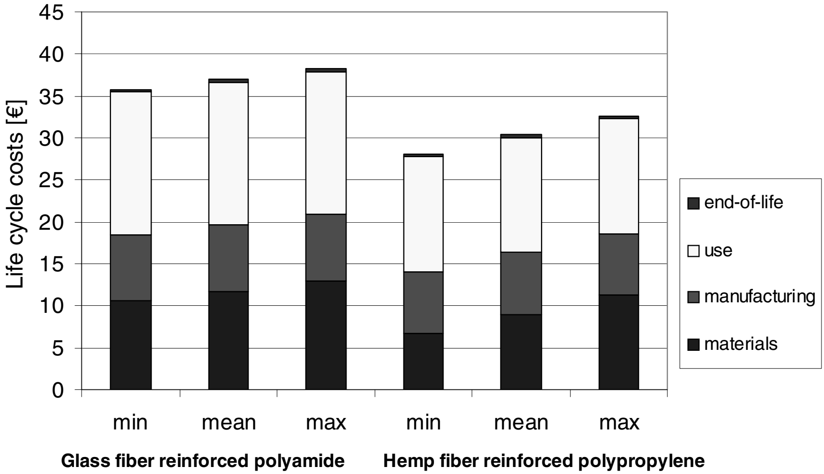

Figure 4-7: Default (average) scenario: comparison of the life cycle costs of the front subframe system from PA-GF with that from PP-HF. Minimum and maximum prices for the materials as well as default data for the effect of fuel savings (0.35 l gasoline per 100 km and 100 kg), fuel price (1 € per liter), and discount rate (5%) were used; service life: 150 000 km, lifetime: 12 years

Analyzing the results of the default scenario, the hemp fiber reinforced polypropylene component shows potentially lower life cycle costs. Compared to the glass fiber reinforced polyamide, the life cycle costs are about 10% (PP-HF, max vs. PA-GF, min.) to 24% (PP-HF, min vs. PA-GF, max.) lower. The main differences result from the use phase, though the material production is also of high importance, especially since the range of the material costs of the PP-HF is quite large, differing almost by a factor of 2. With the maximum value for the material price of PP-HF the overall differences are reduced to the differences by the use phase.

Figure 4-8 shows the maximum scenario, i.e. the results of the LCC calculations using the default values for manufacturing and end-of-life costs as well as the maximum data (regarding influence of use phase) from Table 4-6. For the material prices data ranges were analyzed as in Figure 4-7 (min, mean, max according to Table 4-5).

With such a scenario, the lighter component from PP-HF leads to cost savings of around 20%; differences in the production of the materials only play a minor role in this comparison.

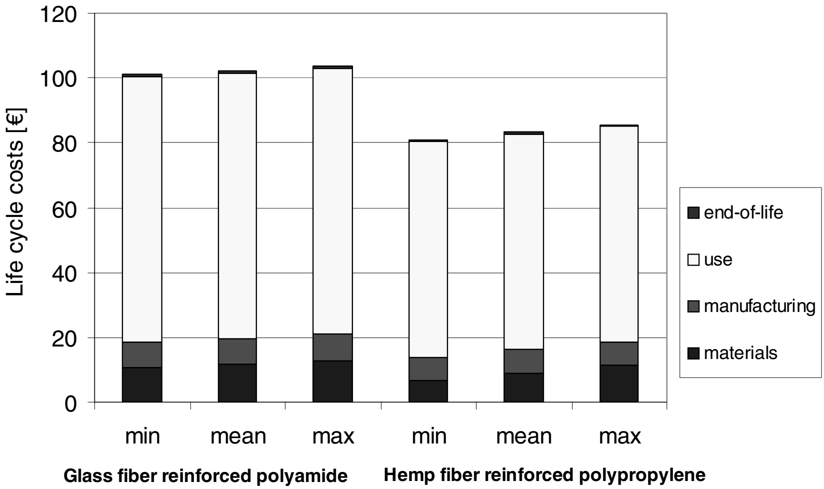

Figure 4-8: Maximum scenario (for influence of use phase): comparison of the life cycle costs of the front subframe system from PA-GF with that from PP-HF. Minimum and maximum prices for the materials as well as maximum data for the effect of fuel savings (0.5 l gasoline per 100 km and 100 kg), fuel price (1.5 € per liter), and discount rate (0%) were used; service life: 250 000 km, life time: 12 years

Finally, Figure 4-9 shows the minimum scenario, i.e. the results of the LCC calculations using the default values for manufacturing and end-of-life costs as well as the minimum data (regarding influence of use phase) from Table 4-6. For the material prices data ranges were analyzed as in Figure 4-7 and Figure 4-8 (min, mean, max according to Table 4-5).

In this scenario, the differences are between nearly 0% and about 30%, also in favor of PP-HF. In this case, the differences are reduced essentially to the differences in material costs and the use phase only has a very small influence.

Overall, Figure 4-8 and Figure 4-9 show results of exaggerated extreme scenarios, while the default scenario seems to be the most probable and the reality will be between the minimum and the maximum scenario. However, the extremes give an indication of the range of changes that can result from different assumptions, data, and different use patterns (influencing mainly the service life of the car, but also the driving style).

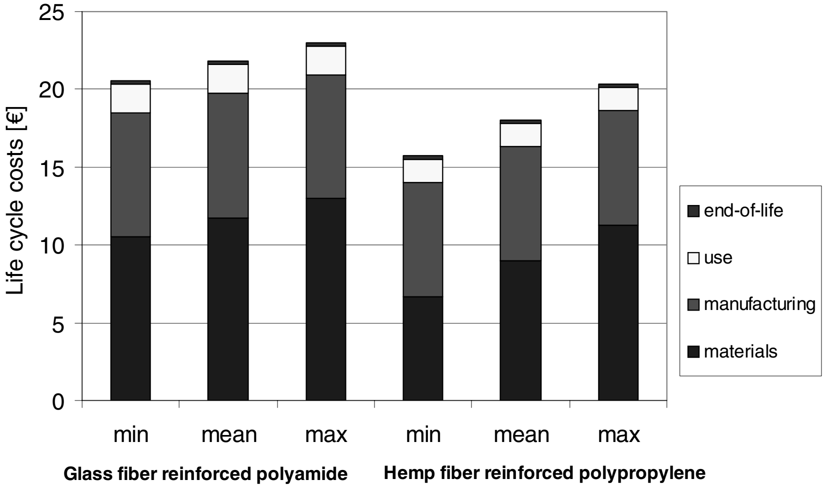

Figure 4-9: Minimum scenario (for influence of use phase): comparison of the life cycle costs of the front subframe system from PA-GF with that from PP-HF. Minimum and maximum prices for the materials as well as minimum data for the effect of fuel savings (0.1 l gasoline per 100 km and 100 kg), fuel price (0.5 € per liter), and discount rate (10%) were used; service life: 150 000 km, life time: 12 years

4.1.3.4 Interpretation and Combined View of LCC and LCA

The results presented in the previous Section show clearly that there are potentials to reduce the life cycle costs of the front subframe system by using hemp fiber reinforced polypropylene in place of glass fiber reinforced polyamide. Only if, with the natural fiber reinforced material as a starting point, the most disadvantageous assumptions in regards to material price (maximum for PP-HF and minimum for PA-GF) within the extreme and exaggerated minimum scenario for the influence of the use phase are assumed, is the overall difference zero. In all other cases there are advantages for PP-HF, with the prerequisite that the automotive component from this material can be injection molded with costs similar to those of PA-GF.

In addition, there seem to be potentials in regards to the price of the natural fiber reinforced plastic. If the plastic and the fibers can be purchased at significantly lower prices that the conventional composite, the price range giving an implication for this potential, cost benefits can be expected in any case.

Therefore, from the point of view of LCC, the recommendation is to invest in R&D for the manufacturing of the component from natural fiber reinforced thermoplastics. If the manufacturing problems can be solved with reasonable efforts, a better economic performance can be expected, especially since this technology could also be used for many other components in the automotive and other sectors.

In addition to R&D for the manufacturing, from an economic point of view, the market of hemp fibers with regards to supply, price stability and price development, should also be closely observed. Before investing, a feasibility study taking into account the series number of this and similar parts for the material should be carried out in order to avoid unexpected developments later (as e.g. rocketing prices due to rising demand and supply shortages).

4.2 Municipal Wastewater Treatment

The main parts of this Section on an LCA and LCC case study of municipal waste water treatment, which employs the developed life cycle inventory based LCC method (see Section 3.4), has been published as part of an article in the journal Environmental Progress [Rebitzer et al. 2003a]. This very detailed and comprehensive life cycle assessment would not have been possible without the contributions from [Braune 2002] and [Stoffregen 2003], who collected and compiled the life cycle inventory data.

4.2.1 Goal and Scope

When assessing the environmental impacts and costs of options for the treatment of municipal wastewater, one cannot only focus on the quality of the end-product, the cleaned water, or the costs for the operation of the wastewater treatment plant, respectively. The impacts and costs caused by the operation of the plant as well as by upstream processes (for e.g., the production of ancillaries) and downstream operations (e.g., treatment and transport of produced sludge) have to be taken into account as well. This is of relevance, because these activities by themselves also use water and other resources and cause pollution and costs and one has to avoid activities that cause more harm than good. The aforementioned assessment is only possible with a systemic life cycle approach as outlined in the previous Section of this paper, since the consequences of the involved relevant activities are taken into account and their interrelations and trade-offs have to be modeled.

For this case study the choice and application of ancillaries is of particular interest (e.g., purely inorganic coagulation vs. combined inorganic coagulation and organic flocculation) and has never been studied before. In addition, the trade-offs between application of chemicals and downstream benefits have to be examined. Here, dewatering of sludge is an important issue [Farinato and Huang 1999]. The choice of the ancillaries for dewatering can have important implications on the overall environmental impacts [Lyons and Vasconcellos 1997] and costs, as will also be shown in this paper.

For the impact assessment phase, CML 2001 [Guinée et al. 2002] was selected, though only focusing on globally and regionally oriented impact categories (GWP, ODP, RDP, Nutrification, Acidification).

Different resulting options for using ancillaries for coagulation and flocculation as well as different options for the final disposal and treatment of wastewater sludge (incineration vs. application in agriculture) have been studied. The chosen reference flow (see the previous Section, here identical with the functional unit in LCA terms [ISO 14040: 1997]) to be assessed is the treatment of the average amount of a typical municipal wastewater (typical contamination) per year and person in Switzerland. As a cost perspective (see previous the Section) the perspective of a company or municipality operating the wastewater treatment plant is chosen, because the aforementioned questions regarding ancillary use and downstream treatment are taken by this actor in the life cycle.

In order to assess, and compare, different treatment options, a modular system model has been developed, allowing the assessment of different strategies. This is used to create a basis for the planning of new wastewater treatment plants (in the sense of Design for Environment, see [Rebitzer and Schmidt 2003]), as well as to assist decisions in existing plants for the treatment of municipal wastewater. Figure 4-10 shows the principal system model used in this study.

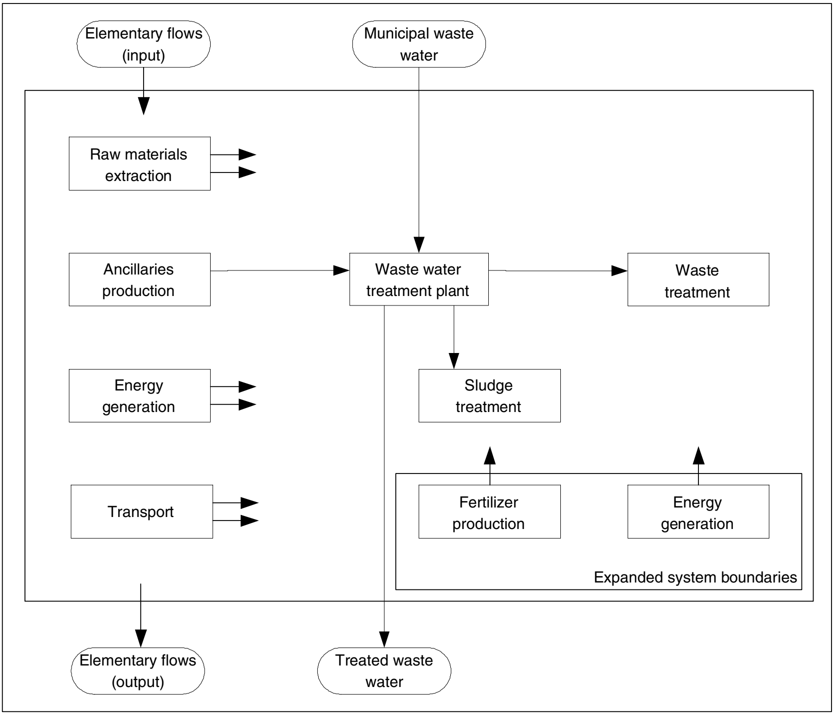

Figure 4-10: Model of the LCA system for municipal wastewater treatment [Rebitzer et al. 2002]

4.2.2 System Modeling

In order to model the complete system as shown in Figure 4-10 efficiently and targeted at the decisions that have to be made in the planning and operation of wastewater treatment plants the most important parameters and options were selected to be variables in the model:

- Composition of wastewater
- Application of chemicals for phosphorous removal and sludge treatment (conditioning and dewatering)
- Technical specification of the wastewater treatment plant (WWTP)
- Type of sludge disposal (agriculture or incineration)

- Transport distances between treatment plant and sludge disposal location

These various parameters are briefly discussed in the following.

Composition of Wastewater: required inputs such as energy demand and chemicals needed as well as generation of waste and emissions depend on the composition of the water to be treated (e.g., carbon, nitrogen, and phosphorous concentration). Therefore, the most important parameters characterizing municipal wastewater can be varied in the model. For the study at hand the typical composition of municipal wastewater in Switzerland was employed.

Application of Chemicals in the WWTP: the model is built around three scenarios, whereas Scenario A represents treatment without using any chemicals for phosphorous removal and sludge treatment. Scenario B assumes the application of iron sulfate for phosphorous removal without any other measures. Scenario C describes the application of polymeric flocculants (polyacrylamides) for sludge conditioning and dewatering in addition to the inorganic chemical use as in Scenario B. A novel detailed life cycle inventory of polymeric flocculants has been established in order to carry out the study.

Specifications of the Wastewater Treatment Plant: a representative wastewater treatment plant of a specific size range (for 10,000 to 50,000 person equivalents[31]) was modeled, the different process steps adapted to the strategies for chemical use (see above). Figure 4-11 shows an example of the treatment plant for the option of using both inorganic coagulants and organic flocculants in the process. It represents the general specifications of the plant in Scenario C.

Sludge Disposal Options: one principal decision in wastewater treatment concerns the selection of the disposal strategy for the resulting sludge. In this model, the two most significant options in practice, incineration and application of the sludge for agricultural purposes, are taken into account. This is clearly an important decision, as the results show (see below), but also a highly political issue in regards to the discussions e.g., in Europe concerning agricultural practices and incineration of waste.

Transport Distances: depending on the regional population structures, the existing infrastructure, and availability of disposal locations (centralized or decentralized) different transport distances for the sludge result.

[31] In the region (Canton) Vaud of Switzerland, for instance, 181 out of 183 plants are of sizes below 50 000 person equivalents. In Switzerland overall, less than 5% of all waste water plants have a capacity above 50 000 person equivalents.

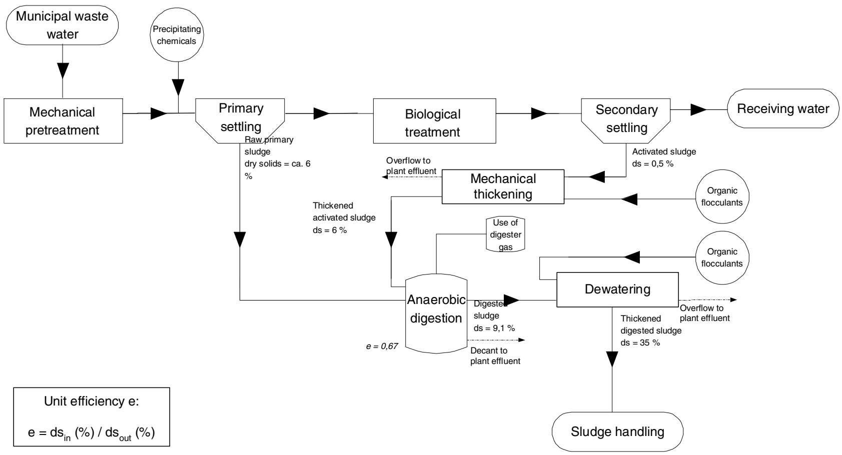

Figure 4-11: Specification of the wastewater treatment plant, here for scenario C [Rebitzer et al. 2002]

Table 4-7 summarizes the basic wastewater treatment scenarios and additional assumptions.

Table 4-7: Studied wastewater treatment scenarios and assumptions for the treatment of typical municipal wastewater in Switzerland

	Scenario A	Scenario B	Scenario C
Inorganic chemical for phosphorous removal (coagulation)	-	Iron sulfate	Iron sulfate
Organic chemical for sludge dewatering (flocculation)	-	-	Cationic polyacrylamides
Specification of WWTP	10,000 to 50,000 person equivalents	10,000 to 50,000 person equivalents, adapted to aforementioned chemical use	10,000 to 50,000 person equivalents, adapted to aforementioned chemical use
Sludge disposal	Incineration or agriculture	Incineration or agriculture	Incineration or agriculture
Transport distances for sludge disposal	40, 100, 200 km	40, 100, 200 km	40, 100, 200 km

4.2.3 Results

In this Section the results related to Scenario C (application of polymeric flocculants (polyacrylamides) for sludge treatment in addition to the inorganic chemical use) with incineration of the sewage sludge are presented and discussed in detail.

Overall, compared to A and B, Scenario C was found to be preferable in regards to environmental impacts and life cycle costs both for incineration and agricultural application of the sludge, assuming an average distance between the wastewater treatment plant and the final sludge disposal of 40 km. Assuming the same transport distances the ceteris paribus comparison of incineration and agricultural application shows that agricultural application is preferable; which can be explained mainly by the substitution of fertilizer in the latter option. In the detailed example below, however, incineration is assessed, since it is or will be mandatory in many countries in the near future. The general environmental rankings of the options were obtained by examining impacts on global warming, acidification, nutrification, ozone depletion, and resource consumption. An additional assessment on the basis of human and eco-toxicity is in progress and will complete the study.

Figure 4-12 and Figure 4-13 show the results of Scenario C with sludge incineration for the environmental impacts (here global warming) and costs, respectively. If one compares analogous results for the other scenarios and disposal (agricultural use) options one finds that sludge transport (from the treatment plant to final disposal) and sludge drying (before final disposal) are the main variable drivers, i.e. drivers which change from option to option, of impacts and costs for all scenarios. Therefore, efforts to minimize these impacts and costs are advisable, because these can be influenced by the existing and studied alternatives (see Table 4-7).

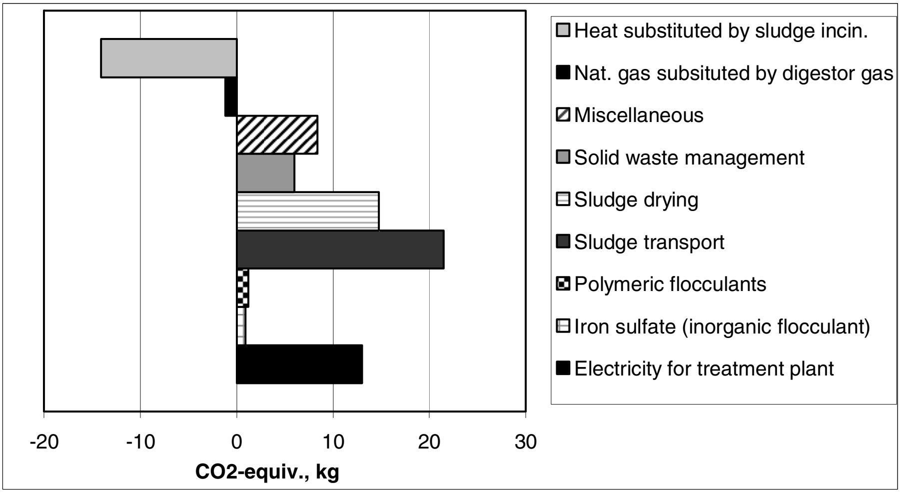

Figure 4-12: Global warming impacts of the different elements of the system of wastewater treatment, expressed in CO_2-equivaltens (scenario C, with incineration of sludge), assuming 40 km transport distance and a sludge with a dry content of 35% leaving the wastewater treatment plant

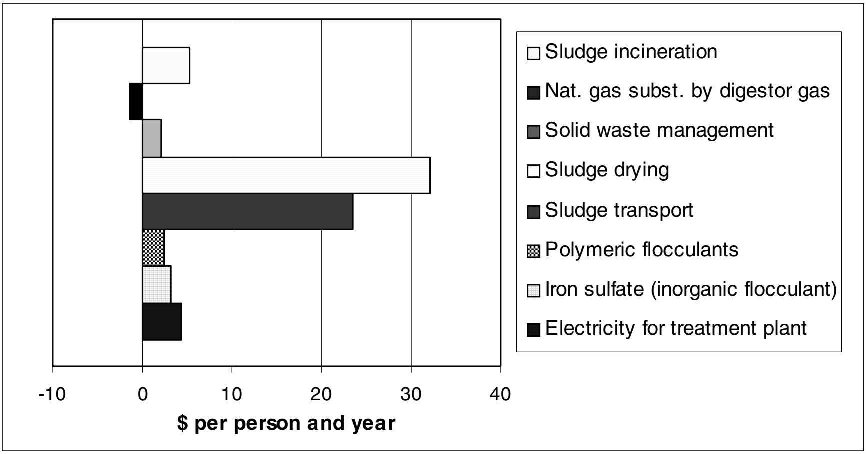

Figure 4-13: Costs of the different elements of the system of wastewater treatment (Scenario C, with incineration of sludge), assuming 40 km transport distance and a sludge with a dry content of 35% leaving the wastewater treatment plant

Measures to reduce the efforts (impacts and costs) for sludge transport and drying relate to

- minimizing the transport distance of the sludge and
- minimizing the water content of the sludge.

The minimization of the transport distance, which often exceeds 100 km, is limited by existing infrastructure and regional characteristics and is therefore difficult to achieve. However, the water content can be reduced quite easily by the application of advanced flocculation chemicals. While the application of better and or additional flocculants leads to higher burdens and costs in production (the flocculants cost up to $3 per kg), it results in significant cost savings in drying and transport, offsetting the additional chemical costs in the wastewater treatment plant by far. One can even say that flocculant costs exceeding $5 per kg or the consumption of higher amounts, if they result in a higher dry substance, are still economically beneficial (see below and Figure 4-14).

The significantly variable parts of the life cycle costs of municipal wastewater treatment (transport and sludge drying, see above) as functions of the dry substance of the sludge leaving the WWTP and the transport distance are illustrated in Figure 4-14. The calculations assume a price of approximately $2 per kg flocculant achieving a dry substance of 30% and $2.50 achieving 35% (currently possible with high performance wastewater-specific flocculants). Wastewater treatment plants without flocculation can achieve about 10% dry substance resulting in extremely high life cycle costs of water treatment. For 40 and 45% dry substance (only theoretical achievable at the moment) flocculant costs of $3 and $6, respectively, were assumed. Essentially, Figure 4-14 determines how much additional money can be spent on improved sludge dewatering by flocculation (and development for improved flocculation polymers and processes), including investments costs, without sacrificing the cost savings in sludge drying and transport.

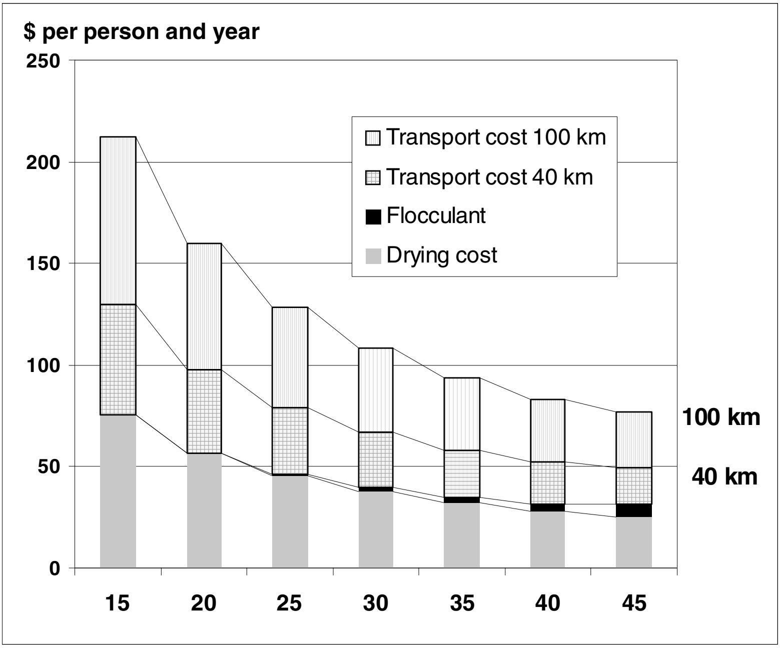

Figure 4-14: Significant variable costs of municipal wastewater treatment as a function of sludge dry substance and disposal transport distance

Clearly, this waste water treatment case illustrates a win-win nature in regard to environmental impacts and life cycle costs. Other, specific, conclusions include the following:

- In all scenarios, the contribution of chemicals (flocculants) is less than 10% of the cost and environmental impacts of drying and transport.

- For transport distances greater than 40 km, the transport activities dominate the impacts and costs.

Figure 4-15 plots the life cycle costs versus impacts on global warming for various options that are based on the scenarios of Table 4-7.

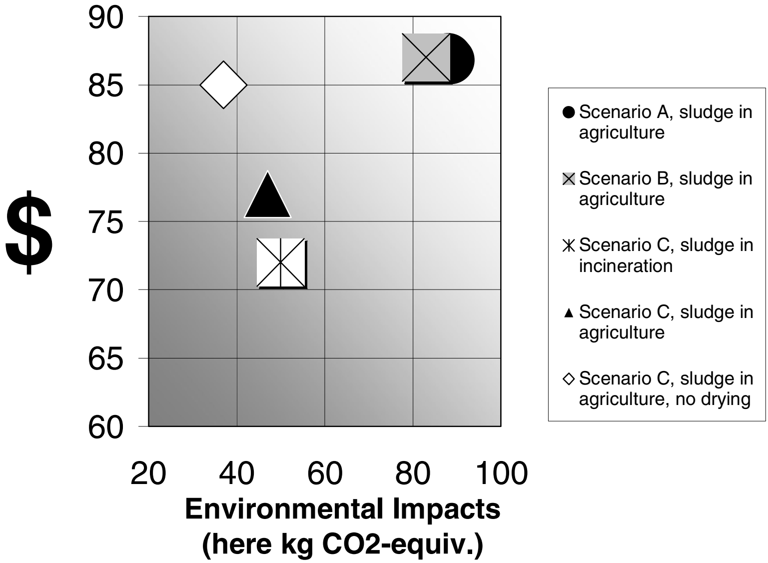

Figure 4-15: Combined view of life cycle costs and environmental impacts of the various options (excerpt, portfolio representation), all with the assumption of 100 km transport distance

In summary, one can state that any activities to improve the flocculation process and thus increasing the dry substance of the sludge are beneficial both to the environmental as well as to the economic bottom-line. This illustrates well how the coordination of the supply chain can lead to important economic and environmental savings, as was mentioned at the outset.

5 Conclusions and Recommendations

The goal of this thesis was to contribute to the enhancement of the application efficiency of life cycle assessment (LCA) for industrial uses. Two principal strategies were followed:

- To simplify the methods of LCA application and
- To use LCA as a basis for life cycle costing.

For the first part, the methodological developments related to the Modular LCA based on foreground processes showed a clear benefit of this approach in the context of industrial decision-making. With this method it is possible to facilitate the application of LCA to a large extent. However, for this approach a fairly complete database is required, and, therefore its applicability for completely new product systems is limited.

For situations where no data and models are available, a framework and specific methods were developed with the aim of guiding the system boundary selection in regards to cut-offs ('micro boundary selection'). For this, the concepts of (i) process specific cut-offs and (ii) baseline approximation were introduced:

(i) For the part related to process specific cut-offs, one can conclude that the chosen approach delivered very useful results for defining cut-off criteria. As a recommendation one can derive to employ cut-offs in the range of 4-5%, if a coverage of 80% of the impacts is regarded as sufficient. Such a coverage should be sufficient for most applications in the material production sectors, since differences of +/- 10%, when comparing alternatives, are commonly considered as non-relevant, due to the inherent uncertainties. However, these figures should be further verified and tested by additional analysis and case studies in various other industry sectors.

(ii) The idea to use input-output LCA as a baseline for estimating the overall impact of an (unknown) product system model of (process) LCA[32], in the context of industrial decision-support, however, seems to be less promising. For one, the sectors of input-output analysis and therefore the unit-processes of input-output LCA are usually not sufficiently specific to differentiate between alternatives within one sector. Secondly, one cannot find a systematic trend between results of input-output LCA and (process) LCA. Specifically, price variations of commodities over time (market fluctuations) and even more due to different product characteristics (different products or commodities produced by one sector, see above), in addition to problems of considering impacts

[32] In this thesis, the term LCA is used for the process oriented assessment methodology according to [ISO 14040: 1997], while LCA based on macro-economic input-output analysis is always denoted as such.

from imports and exports, lead to high variations. These variations do not allow to derive general recommendations for using input-output LCA as a baseline for approximation.

The use of LCA models for life cycle costing is the first consistent attempt to cover the economic dimension of sustainability in parallel to the environmental assessment. This combination of approaches has proven to be very useful and valid, also demonstrated in the case studies from the automotive sector and for wastewater treatment. LCC based on the life cycle inventory of an LCA is a path that should be followed further, since the additional effort needed for an LCC, once the product system model is available, is minimal. In addition, it is a step closer to covering and addressing the life cycle perspective in sustainability.

In relation to the issues outlined above, the thesis also points to recommendations for further research and development:

- In addition to the tool and methodology-oriented work presented in this thesis, future research on the application of LCA should also focus to a larger extent on organizational aspects of using environmental and sustainability assessment methods. Only if these methods can be linked to business processes and different functions within an enterprise or another organization, can the application in the 'real-world' be ensured. Suitable tools and methods need to be embedded in an organization.

- Related to the Modular LCA approach, but also to the LCA methodology in general, one can identify a clear need to develop life cycle impact assessment methods for characterizing the impacts of using water resources, preferable as a function of the regional context. Similarly, the assessment of the impacts of waste generation needs substantial improvement. Ideally, the latter should be developed to a point, where only the elementary flows resulting from final waste treatment and disposal are taken into account. As an intermediate step, however, one could also work on the development of validated waste indicators, based on detailed studies of the impacts of waste management processes.

- It is recommended to carry out further research related to the influence of process-specific cut-offs. As a next step, one could also assess more complex products from various sectors and also the influence on the complete life cycle, including the use and end-of-life phases.

- In order to further establish the economic dimension of sustainability, the existing approaches should be consolidated and one should strive for general scientific agreement, in order to create the basis for harmonization and possible standardization, similar to the development of the LCA methodology in the recent years. However, one should also elaborate the integration of additional economic aspects that cannot be (directly) captured by the proposed LCC method, such as influence on employment levels or income distributions. Additional work should also focus on the interfaces and commonalities between life cycle focused environmental, economic, and social assessments. Only if the latter can also be assessed in a (semi-) quantitative way, can sustainability truly be integrated into the decision-making framework of organizations (‘only what can be measured, can be managed’).

Reliable, internationally accepted, and easy to use methods that are embedded into the organizational structure of companies or other organizations and tailored to the specific management culture are needed to move ‘sustainable development’ from being a goal or vision to a integral element of management processes. The author hopes that this thesis will contribute to these developments by providing some elements on the everlasting path to a more sustainable society.

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9 Abbreviations

ABC	Activity Based Costing
ABS	Acryl Butadiene Styrene
APME	Association of Plastics Manufacturers in Europe
CBA	Cost Benefit Analysis
CEDA	Comprehensive Environmental Data Archive
CFC-11	Trichlorofuoromethane
CML	Centrum voor Milieukunde Leiden, The Netherlands
CO_2	Carbon Dioxide
DALYs	Disability Adjusted Life Years
DfE	Design-for-Environment
EAA	European Aluminium Association
EC	European Commission
EHS	Environment, Health, and Safety
Eiolca	Economic Input-Output Life Cycle Assessment
EMAS	Environmental Management and Audit Scheme
EMS	Environmental Management System
EOL	End-of-Life
EPA	Environmental Protection Agency
EPD	Environmental Product Declaration
ERP	Enterprise Resource Planning
EuP	Energy-Using Products
FCA	Full Cost Accounting
FEFCO	European Federation for Corrugated Board Manufacturers
GWP	Global Warming Potential
IAI	International Aluminium Institute
IISI	International Iron and Steel Institute
I/O-LCA	Input-Output Life Cycle Assessment
IPP	Integrated Product Policy
ISO	International Organization for Standardization
IT	Information Technology
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle inventory

LCIA	Life Cycle Impact Assessment
LCM	Life Cycle Management
MS	Microsoft
NGO	Non-Governmental Organization
NO_x	Nitrogen Oxides
OECD	Organization for Economic Co-operation and Development
OpenLC	Open-Source Toll for Life Cycle Assessment
PA	Polyamide
PA-GF	Polyamide – Glass Fiber reinforced
PE (HD)	Polyethylene high density
PLA	Product Line Analysis
PM 2.5	Particulate Matter (diameter: 2.5 micrometers)
PP	Polypropylene
PP-HF	Polypropylene – Hemp Fiber reinforced
PO_4^{3-}	Phosphate
PPP	Pollution Prevention Pays
PSM	Product System Modeling
R&D	Research and Development
RTI	Research Triangle Institute
SD	Sustainable Development
SETAC	Society of Environmental Toxicology and Chemistry
SME	Small and Medium Sized Enterprises
SO_2	Sulfur Dioxide
TCA	Total Cost Accounting
TCO	Total Cost of Ownership
TRI	Toxic Release Inventory (USA)
UNEP	United Nations Environment Program
WBCSD	World Business Council for Sustainable Development
WCED	World Commission on Environment and Development
WG	Working Group
WWT	Wastewater Treatment
WWTP	Wastewater Treatment Plant

10 Units

Table 10-1: Definition of units used in this thesis

GPa	Gigapascal
kg	Kilogram
kg/dm^3	Kilogram per Cubic Decimeter
kgeq	Kilogram Equivalents
kJ	Kilogram
kJ/m^2	Kilojoule per Square Meter
km	Kilometer
Mg	Megagram
MJ	Megajoule

11 Glossary

The definitions of terms rely to a major extent on the definitions given in the ISO standards and the references cited. However, some of them are new, defined differently, or are used in a new, different context.

Building block: Existing unit process data sets, existing data models, or aggregated data from previous LCA studies or existing data bases [Udo de Haes et al. 2002].

Conventional LCA: The term 'conventional LCA' refers to the methodological procedure exactly as described in [ISO 14040: 1997; 14041: 1998; 14042: 2000], relating to the single steps as well as the order of steps. It is a detailed LCA that complies to the ISO standards without specific modifications or adaptations.

Cost management: Cost management encompasses all (control) measures that aim to influence cost structures and cost behavior precociously. Among these tasks the cost within the value chain have to be assessed, planned, controlled, and evaluated [Hilton et al. 2000; Kaplan and Cooper 1997].

Cradle-to-gate: The cradle-to-gate system only includes upstream processes of the product under study. Downstream processing of the manufactured product, its use, the end-of-life and scrap recovery processes are not considered in the inventory.

Design and Development: "Set of processes that transforms requirements into specified characteristics or into the specification of a product, process or system. … The terms "Design" and "development" are sometimes used synonymously and sometimes used to define different stages of the overall design and development process." [ISO 9000: 2000].

Design for Environment (DfE, syn: ***Eco-Design, Green Design):*** A general term for a number of methods for incorporating environmental factors into the design process [SETAC 1996]. The use of the concept, in various forms often expressed as DfX, is confirmed to the design of products and therefore to industrial applications.

Detailed LCA: LCA study in which the processes and flows within the system are modeled as complete and detailed as possible [de Beaufort-Langeveld et al. 1997].

DfE: see ***Design for Environment***

Eco Labeling: Eco labeling can be divided in different types of labeling according to the ISO Standards 14020-25 [ISO 14020: 2000; ISO 14021: 1999; ISO 14024: 1999; ISO/TR 14025: 2000]. ISO define three different types of labeling Type I, Type II and Type III: Type III is also called ***Environmental Product Declaration***. Type I labels are voluntary schemes. In order to be granted the label applicants have to fulfill a pre-set set of criteria based on life cycle considerations for the product group. Compliance with the criteria is controlled by a third party. Type II labels are also called environmental self-claims. These are single issue labels which a manufacturer can place on his product. There is no third party verification of the correctness of the claim.

Eco-Design: see ***Design for Environment***

EcoDS: see ***Environmental Conscious Decision Support System***

Eco-Efficiency "is achieved by the delivery of competitively-priced goods and services (= products, the authors) that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity throughout the life-cycle to a level at least in line with the earth's estimated carrying capacity. In short, it is concerned with creating more value with less impact" [WBCSD 2000].

Eco-indicator 95 minus: Indicator based on the Eco-indicator 95 methodology [Goedkoop 1995] where the contributions of the single indicators greenhouse gas potential and primary energy demand are subtracted; given in points.

Eco-indicator 99 minus: Indicator based on the Eco-indicator 99 methodology [Goedkoop and Spriensma 2001] where the contributions of the single indicators greenhouse gas potential and primary energy demand are subtracted; given in points.

Elementary flow: (1) "Material or energy entering the system being studied , which has been drawn from the environment without previous human transformation", (2) "Material or energy leaving the system being studied, which is discarded into the environment without subsequent human transformation" [ISO 14040: 1997].

EMAS: see ***Environmental Management Systems***

EMS: see ***Environmental Management Systems***

Environment: "Surroundings in which an organization operates, including air, water, land, natural resources, flora, fauna, humans, an their interrelation" [ISO 14001: 2004].

Environmental Conscious Decision Support System (EcoDS) is a software based decision support tool for a cost-risk evaluation of environmentally conscious alternatives, using ***Streamlined Life Cycle Assessment*** and ***Life Cycle Costing*** It includes a preliminary screening of the potentially most relevant life cycle stages and impacts and uses three metrics for comparison between alternatives: cost, impact assessment and a qualitative measure of the potential business opportunity, or risk, for a given stage-impact pair [Hunkeler et al. 1998].

Environmental Impact: "Any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organization's activities, products or services that can interact with the environment" [ISO 14001: 2004].

Environmental Management Systems (EMS) (ISO 14000 and EMAS): A corporate management strategy including an environmental policy and goals for the activities of the company. The EMS commits the company to continuous improvements of the environmental performance. [ISO 14001: 2004] In the case of ***EMAS*** there is an obligation of publishing a periodical report on the company' s activities, environmental policy and environmental performance.

Environmental Product Declaration (EPD): Declaration of a product's performance with regard to different environmental parameters (e.g. use of resources, emissions, content of chemical substances) during the products life cycle either cradle-to-grave or cradle-to-gate (type III labeling) [ISO/TR 14025: 2000].

Environmental Reporting is a means of communication, e.g. annually, of the environmental aspects of the activities of an organization [WICE 1994].

Environmentally Preferable Purchasing: Selecting products or services that have a lesser or reduced affect on human health and the environment compared with competing products or services that serves the same purpose [US EPA 1999].

EPD: see ***Environmental Product Declaration***

European Eco-Management & Audit Scheme (EMAS): see ***Environmental Management Systems***

Flow chart method: see ***Sequential Method***

Foreground unit process: Processes that are "related specifically to the product system at stake" [Udo de Haes et al. 1994, p. 11].

Functional Unit: "Quantified performance of a ***Product System*** for use as a reference unit in a life cycle assessment study" [ISO 14040: 1997].

Gate-to-gate: The gate-to-gate system only includes on-site related processes of the product under study. Upstream processes and downstream processing of the manufactured product, its use, the end-of-life and scrap recovery processes are not considered in the inventory.

Green and Sustainable Chemistry: Improvements in the ***Eco-Efficiency*** of chemical processes, products and services, to achieve a sustainable, cleaner and healthier environment and a competitive advantage [Jensen 2000].

Green Design: see ***Design for Environment***

Green Procurement is a concept for reducing the environmental burden by buying products with a reduced ***Environmental Impact*** compared to similar products. In order to help public and private procurers guidelines for green procurements can be developed.

Horizontal cut-offs*:* are based on the image of a flow chart where the flows start with resource extraction at the top and end with the final disposal at the bottom.

Independent LCA module: Classified and characterized independent LCI module resulting in an independent module providing the life cycle category indicator result.

Independent LCI module: Extended unit processes with defined input and output system flows.

Industrial Ecology: Multidisciplinary study of industrial systems and economic activities, and their links to fundamental natural systems. It provides the theoretical basis and objective understanding upon which reasoned improvement of current practices can be based. [Allenby 1999].

Integrated Pollution Prevention: see ***Pollution Prevention***

Integrated Product Policy (IPP) "is an environmental policy toolbox currently being discussed within the EU. Its aim is from green markets through an integrated use of policy tools to green consumption (demand side) and to green ***Product Development*** (supply side). As a policy concept, IPP aims to take a life cycle perspective ('cradle-to-grave'), include all relevant stakeholder viewpoints and consider (in the case of products) the ***Product Development*** process from idea generation to product management and reverse logistics. In addition, it points towards reducing resource use and the environmental impact of waste which should be implemented in co-operation with business [EC 2003b].

IPP: see ***Integrated Product Policy***

LCA: see ***Life Cycle Assessment***

LCC: see ***Life Cycle Costing***

LCIA: see ***Life Cycle Impact Assessment***

LCM: see ***Life Cycle Management***

Level of confidence: The 'level of confidence' of a simplified LCA is defined in relation to its ability to deliver a similar result as a detailed LCA, which can be seen in regards to the ranking of alternatives in comparisons and/or in absolute values of category indicator results.

Life Cycle Assessment (LCA): "Compilation and evaluation of the inputs, outputs and the potential ***Environmental Impacts*** of a ***Product System*** throughout its life cycle" [ISO 14040: 1997].

Life Cycle Costing (LCC) is "an assessment of all costs associated with the life cycle of a product that are directly covered by the any one or more of the actors in the product life cycle (supplier, producer, user/consumer, EOL-actor), with complimentary inclusion of externalities that are anticipated to be internalized in the decision-relevant future" ([Rebitzer and Hunkeler 2003], modified on the basis of the definition of [Blanchard and Fabrycky 1998]).

Life Cycle Impact Assessment (LCIA) Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential ***Environmental Impacts*** of a ***Product System*** [ISO 14042: 2000].

Life cycle impact assessment category indicators*:* Quantifiable representation of an impact category [ISO 14042: 2000].

Life cycle indicator*:* Generic term representing the life cycle impact assessment category indicators and technical life cycle indicators.

Life Cycle Interpretation: Phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are combined consistent with the defined goal and scope in order to reach conclusions and recommendations [ISO 14043: 2000].

Life Cycle Inventory Analysis (LCI): Phase of life cycle assessment involving the compilation and quantification of inputs and outputs (resources, energy and emissions), for a given ***Product System*** throughout its life cycle (pre-production, production, distribution, use and recycling or disposal) [ISO 14041: 1998].

Life Cycle Management (LCM) is the application of life cycle thinking to modern business practice, with the aim to manage the total life cycle of an organization's product and services towards more sustainable consumption and production [Jensen and Remmen 2004]. It is an integrated framework of concepts and techniques to address environmental, economic, technological and social aspects of products, services and organizations. LCM, as any other management pattern, is applied on a voluntary basis and can be adapted to the specific needs and characteristics of individual organizations [Hunkeler et al. 2004].

Life Cycle Thinking "is a mostly qualitative discussion to identify the stages of the life cycle and/or the potential ***environmental impacts*** of greatest significance e.g. for use in a design brief or in an introductory discussion of policy measures" [de Beaufort-Langeveld 1997].

Macro boundary selection: Process of system boundary selection and limitation, which is mainly carried out in the goal and scope definition, for including and excluding macro elements of the product system such as the use phase of a given product , specific transport activities, or the influence of infrastructure.

Micro boundary selection: Process of system boundary selection and limitation, which is mainly carried out in the life cycle inventory phase, often in an iterative way, defining processes to include based on the relevance for the category indicator result of the macro element.

Modular LCA: An LCA based on re-usable elements. In a Modular LCA the modularity does not only cover the unit processes, but also extensions of the processes and mandatory steps of life cycle impact assessment.

Organization is a company, corporation, firm, enterprise or institution, or part of combination thereof, whether incorporated or not, public or private, that has it own functions and administration [ISO 14001: 1996].

PLA: see ***Product Line Analysis***

Pollution Prevention **(**syn**:** ***Integrated Pollution Prevention*****)** is an essential alternative to end-of-pipe approaches for dealing with environmental pollution. It was introduced 1976 by Dr. Joseph Ling of 3M [Shen 1995, p. 17] and was first implemented in 3M's Pollution Prevention Pays Program (3P) [Royston 1979]. Pollution prevention is based on technological and management advances that reduce environmental releases and the consumption of resources through integrated approaches. These include modifications in production processes, substitution of materials, recycling activities etc. [Royston 1979].

Process Product: A physical or symbolic object (good or service, respectively) which leaves a unit process and whose value in monetary terms is positive. A product is a commercial commodity.

Product Development is the "process of taking a product idea from planning to market launch and review of the product in which business strategies, marketing considerations, research methods and design aspects are used to take a product to a point of practical use. It includes improvements or modifications to existing products or processes" [ISO 14062: 2002].

Product Line Analysis (PLA): Prospective planning tool developed in Germany studying ***Environmental Impacts*** of a particular service to society [Öko-Institut 1987]. Based on ***Functional Units***. Similar to ***LCA*** but includes environmental, economic and social aspects [SETAC 1996].

Product Stewardship: Management of the sustainability aspects of products throughout their life cycles. Can be largely considered as synonymous with ***life cycle management***.

Product System: "Collection of materially and energetically connected unit processes which performs one or more defined functions" [ISO 14040: 1997]. Therefore, the product system is the system in reality.

Product System Model: Model that describes the collection of materially and energetically connected unit processes which performs one or more defined functions [Rebitzer 1999]. Therefore this is the model representation of the ***product system***.

Products are goods and services, or their utility [WCED 1987]. A "product is the result of a process. … There are four generic product categories, as follows: services (e.g.

transport); software (e.g. computer program, dictionary); hardware (e.g. engine mechanical part); processed materials (e.g. lubricant)" [ISO 9000: 2000].

Reference flow: Measure of the needed outputs from processes in a given product system required to fulfill the function expressed by the functional unit [ISO 14041: 1998].

Responsible Care is a program created in 1988 by the Chemical Manufacturers Association (CMA), now called International Council of Chemicals Association (ICCA). It is a program adopted by ICCA's members to foster environmentally responsible management of chemicals. Guiding principles and codes of management practices have been established [Shen 1995, p. 236].

Screening LCA: "A procedure that identifies some particular characteristic or key issue associated with an ***LCA***, which will normally be the subject of further, more intensive, study" [de Beaufort-Langeveld 1997].

Sequential method: Method for scaling the inventory processes not simultaneously but in a sequential way [Heijungs and Suh 2002].

Simplified LCA (syn**:** ***Streamlined LCA):*** An LCA obtained through a procedure that reduces the complexity of an ***LCA*** and therefore cost, time and effort involved in the study. "This may involve exclusion of certain life cycle stages, system inputs or outputs, or impact categories, or may involve the use of generic data modules rather than specific data for the system under study" [de Beaufort-Langeveld 1997].

Streamlined LCA: see ***Simplified LCA***

Supply Chain Management focuses on globalization and information management tools which integrate procurement, operations, and logistics from raw materials acquisition to customer satisfaction [ASUBUSINESS 2005].

Sustainable Development (syn: ***Sustainability)*** is development that "meets the needs of the present without compromising the ability of future generations to meet their own needs" [WCED 1987]. Sustainable development addresses economic, environmental and social aspects.

Sustainability: see ***Sustainable Development***

System Boundary: "Interface between a product system and the environment or other product systems" [ISO 14040: 1997].

System flow: In order to distinguish elementary flows from input or output flows which lead to the identification of a unit process, input and output flows that are output of or input to other processes are termed 'system flows'.

Technical life cycle indicators: Indicators building on the technical non-elementary flows (see ***Technical non-elementary flows***).

Technical non-elementary flows: (1) Material or energy entering the system being studied, e.g. water as resource, for which the preceding human transformation processes, e.g. extraction from specific water bodies, have not been taken into account; (2) Material and energy leaving the system being studied, e.g. waste, for which the subsequent human transformation processes, e.g. waste disposal processes, have not been taken into account.

Unit process: Smallest portion of a product system for which data are collected when performing a life cycle assessment [ISO 14040: 1997].

Vertical cut-offs: based on the image of a flow chart where the flows start with resource extraction at the top and end with the final disposal at the bottom.

12 Appendices

12.1 Automotive Front Subframe System LCI Data

Table 12-1: Cradle to gate inventory (inventory results) for the production of the front subframe system from glass fiber reinforced polyamide (without radiation, without application of cut-offs). The exact definition of the flows, including sum parameters, can be found it [Frischknecht et al. 1996].

Flow type	Elementary flow	Flow Quantity	Unit
Emission into air	1,2-Dichlorotetrafluoroethane	1.68934E-06	kg
Emission into air	Acetaldehyde	5.7742E-06	kg
Emission into air	Acetic acid	2.71345E-05	kg
Emission into air	Acetone	5.76051E-06	kg
Emission into air	Acrolein	1.1994E-10	kg
Emission into air	Aldehydes	1.6112E-07	kg
Emission into air	Alkanes	0.000136471	kg
Emission into air	Alkenes	1.38698E-05	kg
Emission into air	Aluminum	0.000148897	kg
Emission into air	Ammonia	2.842343079	g
Emission into air	Anilin	4.40899E-08	kg
Emission into air	Antimony	1.41528E-07	kg
Emission into air	Aromates	3.15762E-06	kg
Emission into air	Aromatic hydrocarbons	0.157371241	g
Emission into air	Arsenic	8.23313E-07	kg
Emission into air	Arsenic tri-oxide	1.35682E-05	kg
Emission into air	Barium	2.43057E-06	kg
Emission into air	Benzaldehyde	6.25772E-11	kg
Emission into air	Benzene	5.19163E-05	kg
Emission into air	Benzo(a)pyrene	3.50874E-09	kg
Emission into air	Beryllium	2.59394E-08	kg
Emission into air	Boron	0.000113593	kg
Emission into air	Boron tri-oxide	0.024600202	kg
Emission into air	Bromine	1.16071E-05	kg
Emission into air	Butane	0.000490941	kg
Emission into air	Butene	9.32652E-06	kg
Emission into air	Cadmium	8.70512E-07	kg
Emission into air	Calcium	0.000175335	kg
Emission into air	Carbon dioxide	27.27341068	kg
Emission into air	Carbon monoxide	0.012508202	kg
Emission into air	Chlorine	0.003192169	kg
Emission into air	Chlorodifluoromethane	1.50392E-08	kg
Emission into air	Chlorotrifluoromethane	8.63953E-09	kg
Emission into air	Chromium	1.23832E-06	kg
Emission into air	Cobalt	1.87858E-06	kg
Emission into air	Colemanite dust	2.17274E-06	kg
Emission into air	Copper	4.07845E-06	kg
Emission into air	Cyanide	3.71308E-09	kg
Emission into air	Dichlorodifluoromethane	1.37593E-08	kg
Emission into air	Dichloromethane	5.23742E-09	kg

Flow type	Elementary flow	Flow Quantity	Unit
Emission into air	Dichloromonofluoromethane	7.44264E-09	kg
Emission into air	Dinitrogen monoxide	0.000173029	kg
Emission into air	Dust, silicotic	7.0672E-05	kg
Emission into air	Ethane	0.001334015	kg
Emission into air	Ethanol	1.15677E-05	kg
Emission into air	Ethene	2.14883E-05	kg
Emission into air	Ethylbenzene	2.19657E-05	kg
Emission into air	Ethyne	1.29075E-07	kg
Emission into air	Flue gas (empty)	1776.045845	kg
Emission into air	Fluoride	0.004539323	kg
Emission into air	Formaldehyde	3.60297E-05	kg
Emission into air	Heat, waste	0.000207495	TJ
Emission into air	Helium	0.000327503	kg
Emission into air	Heptane	9.32652E-05	kg
Emission into air	Hexane	0.000195856	kg
Emission into air	Hydrocarbons	0.228435189	g
Emission into air	Hydrocarbons, aliphatic	4.327386929	g
Emission into air	Hydrochloric acid	0.001758544	kg
Emission into air	Hydrogen	0.318229386	g
Emission into air	Hydrogen fluoride	0.000230383	kg
Emission into air	Hydrogen sulfide	0.000112247	kg
Emission into air	Iodine	5.26463E-06	kg
Emission into air	Iron	0.00010443	kg
Emission into air	Kaolin dust	9.16909E-06	kg
Emission into air	Lanthanum	7.02178E-08	kg
Emission into air	Lead	2.99013E-06	kg
Emission into air	Lime dust	3.12118E-06	kg
Emission into air	Magnesium	5.29063E-05	kg
Emission into air	Manganese	7.86904E-07	kg
Emission into air	Mercury	6.38252E-07	kg
Emission into air	Methane	0.06437068	kg
Emission into air	Methane	0.011872564	g
Emission into air	Methanol	1.35092E-05	kg
Emission into air	Molybdenum	5.59376E-07	kg
Emission into air	Nickel	2.34821E-05	kg
Emission into air	Nitric acid	3.41534E-08	kg
Emission into air	Nitric oxide	1.804461928	g
Emission into air	Nitrobenzene	1.24774E-07	kg
Emission into air	Nitrogen	0.001681104	kg
Emission into air	Nitrogen dioxide	52.07852423	g
Emission into air	Nitrogen oxides	1.665005896	g
Emission into air	Nitrogen trioxide	0.005535159	kg
Emission into air	PAH, polycyclic aromatic hydrocarbons	3.79338E-07	kg
Emission into air	Particulate matter	8.476073474	g
Emission into air	Pentane	0.000541064	kg
Emission into air	Phosphorus	2.24812E-06	kg
Emission into air	Platinum	1.1249E-10	kg
Emission into air	Potassium	1.90403E-05	kg
Emission into air	Propanal	6.25772E-11	kg
Emission into air	Propane	0.000724311	kg
Emission into air	Propene	1.99262E-05	kg
Emission into air	Propionic Acid	5.43449E-07	kg

Flow type	Elementary flow	Flow Quantity	Unit
Emission into air	Scandium	2.33994E-08	kg
Emission into air	Selenium	1.6675E-06	kg
Emission into air	Silicon	0.000547738	kg
Emission into air	Sodium	4.37767E-05	kg
Emission into air	Strontium	2.39432E-06	kg
Emission into air	Sulfur dioxide	105.0451331	g
Emission into air	TCDD equivalents	0.309539518	ng
Emission into air	Tert-butyl methyl ether	1.93033E-09	kg
Emission into air	Tetrachloromethane	8.68308E-10	kg
Emission into air	Thallium	1.71411E-08	kg
Emission into air	Thorium	4.50248E-08	kg
Emission into air	Tin	5.15072E-08	kg
Emission into air	Titanium	6.70486E-06	kg
Emission into air	Toluene	6.95059E-05	kg
Emission into air	Trichlorofluoromethane	6.39965E-08	kg
Emission into air	Uranium ore	5.05266E-08	kg
Emission into air	Vanadium	8.72831E-05	kg
Emission into air	VOC (w/o methan)	0.043730774	kg
Emission into air	Volatile organic compounds	0.001001497	kg
Emission into air	Water	7.355283412	kg
Emission into air	Xylene	9.10207E-05	kg
Emission into air	Zink	4.47859E-06	kg
Emission into air	Zirconium	2.12176E-09	kg
Emission into fresh water	Acids in general	0.000571671	kg
Emission into fresh water	Alkanes	4.87434E-06	kg
Emission into fresh water	Alkenes	4.48342E-07	kg
Emission into fresh water	Aluminum	0.000133881	kg
Emission into fresh water	Ammonia	0.000353411	kg
Emission into fresh water	AOX	6.61214E-07	kg
Emission into fresh water	Aromates	1.54978E-05	kg
Emission into fresh water	Arsenic	2.01275E-07	kg
Emission into fresh water	Barium	7.37938E-05	kg
Emission into fresh water	Benzene	4.93002E-06	kg
Emission into fresh water	Beryllium	5.12654E-09	kg
Emission into fresh water	Bis(2-ethylhexyl) phtalate	8.64583E-11	kg
Emission into fresh water	BOD	2.86504E-05	kg
Emission into fresh water	Boron	5.84199E-06	kg
Emission into fresh water	Cadmium	1.59572E-07	kg
Emission into fresh water	Calcium	0.001954049	kg
Emission into fresh water	Cesium	2.87093E-08	kg
Emission into fresh water	Chemical Oxigen Demand	0.000574366	kg
Emission into fresh water	Chlorine	0.041329452	kg
Emission into fresh water	Chlorinated solvents	1.42673E-11	kg
Emission into fresh water	Chlorine monoxide	2.63608E-05	kg
Emission into fresh water	Chlorobenzene	1.66642E-13	kg
Emission into fresh water	Chromium III	7.02605E-07	kg
Emission into fresh water	Chromium VI	1.77096E-12	kg
Emission into fresh water	Cobalt	8.74136E-10	kg
Emission into fresh water	Copper	2.43935E-07	kg
Emission into fresh water	Cyanide	7.79788E-07	kg
Emission into fresh water	Dichloromethane	3.0187E-06	kg
Emission into fresh water	DOC	8.20945E-05	kg

Flow type	Elementary flow	Flow Quantity	Unit
Emission into fresh water	Ethane, 1,1,1-trichloro-	1.13968E-08	kg
Emission into fresh water	Ethylbenzene	7.03169E-07	kg
Emission into fresh water	Fatty acids as carbon	0.000140102	kg
Emission into fresh water	Fluoride	6.04171E-06	kg
Emission into fresh water	Grease and/or oil	0.000101478	kg
Emission into fresh water	Heat, waste	6.27443E-06	TJ
Emission into fresh water	Hydrocarbons	1.36344E-06	kg
Emission into fresh water	Hydrogen sulfide	1.06088E-07	kg
Emission into fresh water	Hypochlorite	2.63606E-05	kg
Emission into fresh water	Iodine	2.87093E-06	kg
Emission into fresh water	Iron	0.00482956	kg
Emission into fresh water	Lead	3.76088E-06	kg
Emission into fresh water	Losses from limestone washing	0.047124165	kg
Emission into fresh water	Magnesium	0.000298686	kg
Emission into fresh water	Manganese	1.89832E-05	kg
Emission into fresh water	Mercury	4.53216E-08	kg
Emission into fresh water	Molybdenum	3.63028E-06	kg
Emission into fresh water	Nickel	4.22839E-07	kg
Emission into fresh water	Nitrate	0.000117011	kg
Emission into fresh water	Nitric acid	0.003791205	kg
Emission into fresh water	Nitrogen, in organic substance	5.23834E-05	kg
Emission into fresh water	Nitrogen, total	0.000334228	kg
Emission into fresh water	Non-dissolved substances	9.98793E-05	kg
Emission into fresh water	PAH, polycyclic aromatic hydrocarbons	3.74956E-07	kg
Emission into fresh water	Phenol	6.9602E-06	kg
Emission into fresh water	Phosphate	1.49598E-06	kg
Emission into fresh water	Phosphorus compounds	6.65406E-08	kg
Emission into fresh water	Potassium	0.000294912	kg
Emission into fresh water	Rubidium	2.87093E-07	kg
Emission into fresh water	Salts	0.010471503	kg
Emission into fresh water	Selenium	7.79395E-07	kg
Emission into fresh water	Silicon	1.50935E-06	kg
Emission into fresh water	Silver	1.81718E-08	kg
Emission into fresh water	Sodium	0.01392966	kg
Emission into fresh water	Dissolved solids	0.001432686	kg
Emission into fresh water	Strontium	0.000189757	kg
Emission into fresh water	Sulfate	0.027826027	kg
Emission into fresh water	Sulfide	5.40626E-06	kg
Emission into fresh water	Sulfite	5.17996E-09	kg
Emission into fresh water	Tert-butyl methyl ether	9.99801E-11	kg
Emission into fresh water	Tin	1.26206E-09	kg
Emission into fresh water	Titanium	4.39729E-07	kg
Emission into fresh water	TOC	0.001194755	kg
Emission into fresh water	Toluene	4.17537E-06	kg
Emission into fresh water	Trichloromethane	2.94744E-10	kg
Emission into fresh water	Triethylene glycol	8.20945E-05	kg
Emission into fresh water	Vanadium	1.57542E-06	kg
Emission into fresh water	Volatile organic compounds (as carbon)	1.00483E-05	kg
Emission into fresh water	Waste water	1.114721259	m^3
Emission into fresh water	Xylene	3.54695E-06	kg
Emission into fresh water	Zink	2.10593E-06	kg
Emission into sea water	Alkanes	3.00424E-05	kg

Flow type	Elementary flow	Flow Quantity	Unit
Emission into sea water	Alkenes	2.77314E-06	kg
Emission into sea water	Aluminum	3.87435E-07	kg
Emission into sea water	Ammonia	0.000175112	kg
Emission into sea water	AOX	3.80405E-07	kg
Emission into sea water	Aromates	0.000149942	kg
Emission into sea water	Aromatic hydrocarbons	0.021469474	g
Emission into sea water	Arsenic	7.77504E-08	kg
Emission into sea water	Barium	0.000578517	kg
Emission into sea water	Barium sulfate (resource)	0.00669892	kg
Emission into sea water	Bbenzene	3.005E-05	kg
Emission into sea water	BOD	7.7009E-06	kg
Emission into sea water	Boron	3.42325E-06	kg
Emission into sea water	Cadmium	1.47043E-07	kg
Emission into sea water	Calcium	0.007498099	kg
Emission into sea water	Cesium	2.31095E-07	kg
Emission into sea water	Chemical Oxigen Demand	0.355671746	g
Emission into sea water	Chlorine	0.118706711	kg
Emission into sea water	Chlorine monoxide	6.44073E-06	kg
Emission into sea water	Chromium III	2.60889E-06	kg
Emission into sea water	Copper	3.10258E-07	kg
Emission into sea water	Cyanide	3.87435E-07	kg
Emission into sea water	DOC	2.04223E-05	kg
Emission into sea water	Ethylbenzene	5.54786E-06	kg
Emission into sea water	Fatty acids as carbon	0.001198041	kg
Emission into sea water	Fluoride	2.31105E-06	kg
Emission into sea water	Glutaraldehyde	8.27027E-07	kg
Emission into sea water	Grease and/or oil	0.004896801	kg
Emission into sea water	Heat, waste	2.10671E-06	TJ
Emission into sea water	Hypochlorite	6.44073E-06	kg
Emission into sea water	Iodine	2.31095E-05	kg
Emission into sea water	Iron	2.69879E-05	kg
Emission into sea water	Lead	7.81803E-08	kg
Emission into sea water	Magnesium	0.00015487	kg
Emission into sea water	Manganese	1.10739E-05	kg
Emission into sea water	Mercury	5.26985E-09	kg
Emission into sea water	Molybdenum	7.76941E-08	kg
Emission into sea water	Nickel	5.4094E-07	kg
Emission into sea water	Nitrate	6.37092E-05	kg
Emission into sea water	Nitrite	7.89327E-06	kg
Emission into sea water	Nitrogen, in organic substance	3.01253E-05	kg
Emission into sea water	Nitrogen, total	0.00019231	kg
Emission into sea water	Non-dissolved substances	0.02069752	kg
Emission into sea water	PAH, polycyclic aromatic hydrocarbons	3.00424E-06	kg
Emission into sea water	Phenol	2.65765E-05	kg
Emission into sea water	Phosphate	7.77146E-07	kg
Emission into sea water	Potassium	0.001008691	kg
Emission into sea water	Rubidium	2.31095E-06	kg
Emission into sea water	Selenium	7.79755E-08	kg
Emission into sea water	Silver	1.38657E-07	kg
Emission into sea water	Sodium	0.072117314	kg
Emission into sea water	Sodium ion	0.885186356	g
Emission into sea water	Solved solids	0.000140716	kg

Flow type	Elementary flow	Flow Quantity	Unit
Emission into sea water	Strontium	0.001392009	kg
Emission into sea water	Sulfate	0.001858671	kg
Emission into sea water	Sulfide	3.10259E-06	kg
Emission into sea water	Tert-butyl methyl ether	5.78987E-11	kg
Emission into sea water	TOC	0.001792202	kg
Emission into sea water	Toluene	2.4966E-05	kg
Emission into sea water	Tributyltin	2.94827E-07	kg
Emission into sea water	Triethylene glycol	2.04223E-05	kg
Emission into sea water	Vanadium	7.76941E-08	kg
Emission into sea water	Volatile organic compounds (as carbon)	8.08833E-05	kg
Emission into sea water	Xylene	2.17307E-05	kg
Emission into sea water	Zink	7.77222E-07	kg
Emission into soil	Arsenic tri-oxide	1.35682E-05	kg
Emission into soil	Boron tri-oxide	0.001640013	kg
Emission into soil	Colemanite rock, coarse	0.042741337	kg
Emission into soil	Dust, silicotic	0.001464298	kg
Emission into soil	grease and/or oil	0.000224201	kg
Emission into soil	Heat, waste	6.80875E-08	TJ
Emission into soil	Limestone, coarse	0.06139886	kg
Emission into soil	Overburden	11.81539708	kg
Resource	Air	1725.662975	kg
Resource	Barium sulfate (resource)	0.033411167	kg
Resource	Bentonite	0.002199912	kg
Resource	Calcium fluoride	0.000210436	kg
Resource	Calcium hydroxide	0.066099178	kg
Resource	Calcium phosphate	0.013902375	kg
Resource	Clay	0.000131551	kg
Resource	Coal, brawn (lignite)	2.974355143	kg
Resource	Coal, hard	2.315581194	kg
Resource	Cobalt ore	3.20858E-10	kg
Resource	Colemanite rock, raw	0.420169079	kg
Resource	Disodium sulphate	0.006560054	kg
Resource	Gas, natural (0,8 kg/m^3)	0.282504131	kg
Resource	Iron hydroxide	0.016490151	kg
Resource	Iron ore	1.31514E-10	kg
Resource	Kaolin soil	15.70091013	kg
Resource	Lead ore	1.87878E-09	kg
Resource	Limestone mineral	0.60447938	kg
Resource	Mercury	7.25662E-10	kg
Resource	Nickel ore	2.13709E-13	kg
Resource	Oil, crude (860 kg/m3)	0.005183123	t
Resource	Oxygen (air)	0.005932336	kg
Resource	Palladium (in ore)	5.47454E-08	kg
Resource	Platinum (in ore)	6.17311E-08	kg
Resource	Potential energy water	1.30974E-05	TJ
Resource	Rhenium	4.57071E-08	kg
Resource	Rhodium (in ore)	5.82365E-08	kg
Resource	Rock salt (in ore)	0.002178999	kg
Resource	Sand and gravel	0.383607959	kg
Resource	Uranium ore	0.000201858	kg
Resource	Water	512.9384899	kg
Resource	Wood	1.04164E-10	t

12.2 Curriculum Vitae

Gerald Rebitzer

Nationality: German and US-American

Date of birth: 29 November 1967

Place of birth: El Paso, TX, USA

PROFESSIONAL EXPERIENCE

2003 - present	Alcan Technology & Management, Manager Product Stewardship (also: EHS auditor, Continuous Improvement project manager).
2001 - 2004	Swiss Federal Institute of Technology Lausanne, Researcher and group coordinator in Life Cycle Management (Design for Environment – DfE, Recycling, Life Cycle Assessment - LCA, Life Cycle Costing - LCC).
2001 - 2002	Consultant in DfE, recycling, LCA, LCC, software development.
1997 - 2001	Technical University Berlin, Project manager of a DfE and recycling R&D project in cooperation with five institutes and five industrial companies (Ford Motor Company, MAN Technologie, Sachsenring Entwicklungs GmbH, and others), Leader of the DfE and recycling R&D team; Acquisition and carrying out of several national and international projects in the fields of LCA, LCC, DfE, and recycling/disassembly.
1996 - 2001	Technical University Berlin, Researcher in the DfE and recycling project mentioned above.
1995 - 1996	Consultant (freelancer) in the field of municipal solid waste management in co-operation with Horn und Müller GmbH, Berlin and the Solid Waste Association of North America, Silver Spring, USA; Worked on collection systems, landfilling, and bio-mechanical treatment of waste.
1994 (Oct.-Dec.)	Solid Waste Association of North America, Silver Spring, USA; Internship: developed a guidebook for contracting in municipal solid waste management.

OTHER PROFESSIONAL ACTIVITIES

2005 - present	Editor for the Life Cycle Management Section of the International Journal of Life Cycle Assessment.
2005 - present	Member of the EAA (European Aluminium Association) Product Stewardship Working Group.
1999 - present	Reviewer for the International Journal of Life Cycle Assessment and the Journal of Cleaner Production.
1998 - present	Member of three LCA Working Groups of SETAC; Co-Chairman of the Life Cycle Costing Working Group
1996 - 2002	Member of the EUREKA working group Care Vision 2000 and the Thematic Network ECOLIFE (DfE and recycling in the electronics industry).
Publications	More than 90 publications on DfE, product development, LCM, LCA, LCC, recycling, waste management.
Memberships	Society of Environmental Toxicology and Chemistry (SETAC), International Society for Industrial Ecology (ISIE), Verein Deutscher Ingenieure (VDI).

EDUCATION

2001 - present	Swiss Federal Institute of Technology Lausanne, PhD research project “Enhancing the Application Efficiency of Life Cycle Assessment for Industrial Uses”.
2001	Swiss Federal Institute of Technology Lausanne, Graduate class in entrepreneurship (business start-up and business development).
1989 - 1995	Technical University Berlin, Studies of Environmental Engineering.
1978 - 1988	Herderschule Rendsburg, Germany, German matriculation standard (Abitur).
1985 - 1986	Pasco High School, WA, USA, US-American High School Graduation.

MILITARY SERVICE

1988 - 1989	German Air Force, Special responsibility: elected spokesman of 150 serving men.

SPECIAL SKILLS

Languages	German and English (bilingual), French (basic level).
Information Tech.	Software development (management), windows standard applications, Internet, LCA software.

SPECIAL EXPERIENCES

1985 - 1986	Rotary exchange student in Pasco, WA, USA.
Sports	Distance running (10 years competing on state and national level), bicycling (tandem), basketball.
Traveling	Tandem and bicycle touring in Europe, North America, Asia.

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12.3 List of Publications

Frischknecht, R.; Althaus, H.-J.; Doka, G.; Dones, R.; Heck, T.; Hellweg, S.; Hischier, R.; Jungbluth, N.; Nemecek, T.; Rebitzer, G.; Spielmann, M.: Selected modelling principles applied in the ecoinvent database. Japanese International Journal of Life Cycle Assessment, accepted for publication, 2005.

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Rebitzer, G.; Ekvall, T. (ed.): Scenario Development in LCA. Pensacola, USA: Society for Environmental Toxicology and Chemistry (SETAC), 2004.

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