THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

System Expansion and Allocation in Life Cycle Assessment

With Implications for Wastepaper Management

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Technical Environmental Planning
CHALMERS UNIVERSITY OF TECHNOLOGY
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Abstract

The choice of system boundaries and allocation methods can have decisive effects on the results and conclusions of a life cycle assessment (LCA). System expansion makes it possible to model the indirect effects of a decision; however, this modelling is often based on inaccurate assumptions. Subdivision and/or allocation based on physical, causal relationships model the consequences of a decision that affects the internally used functions but not the exported functions of a multi-function process. Allocation based on gross sales value model the causes of environmental burdens.

System expansion and marginal data can be used in most LCA applications. It can be expected to contribute to individual decisions and actions that result in a lower level of environmental burdens per functional unit than would have been the case without the LCA; however, the modelling of indirect and marginal effects may be restricted by other methodological requirements, such as acceptability, feasibility, and, in some applications, the need for a detailed methodological standard.

This thesis includes a new approach to the allocation problem in open-loop recycling. This approach models the indirect effects of a change in the supply of, or demand for, the recycled material. It can be used for system expansion as well as for allocation. It takes important mechanisms into account, but the precision in the model can be poor. The 50/50 allocation method presented in an earlier paper can be regarded as an approximation of the new approach.

Important methodological issues in the environmental comparison between recycling and waste incineration of wastepaper, old corrugated board, etc. include the modelling of indirect effects and the choice of data for electricity production. The indirect effects depend on which energy source competes with energy from wastepaper incineration, on which material is replaced by the recycled paper, and on the alternative use of forest resources that are not required for pulpwood production. All of these, as well as the marginal electricity production, depend on the time-perspective and on other waste management, forestry, and energy policies.

Key words: environmental life cycle assessment, life cycle inventory analysis, LCI methodology, modelling, system boundaries, system expansion, allocation, waste management, paper recycling, waste incineration.

List of publications

This thesis includes the following papers, referred to by Roman numerals in the text:

I. **Choice of System Boundaries in Life Cycle Assessment**
Tillman, A.-M., Ekvall, T., Baumann, H. and Rydberg, T. (1994) *J. Cleaner Prod.*, Vol. 2, No. 1, pp. 21-29.

II. **Open-Loop Recycling: Criteria for Allocation Procedures**
Ekvall, T., Tillman, A.-M. (1997) *Int. J. LCA*, Vol. 2, No. 3, pp. 155-162.

III. **Comment on Critical Review of Life-cycle Assessment**
Ekvall, T. (1997) *Resources, Conservation and Recycling*, Vol. 19, No. 3, pp. 219-220.

IV. **Life-Cycle Assessment as a Decision-Support Tool - the Case of Recycling Versus Incineration of Paper**
Finnveden, G. and Ekvall, T. (1998) *Resources, Conservation and Recycling*, Vol. 24, Nos. 3-4, pp. 235-256.

V. **Key Methodological Issues for Life Cycle Inventory Analysis of Paper Recycling**
Ekvall, T. *J. Cleaner Prod.* In press.

VI. **Allocation in ISO 14041 - A Critical Review**
Ekvall, T. and Finnveden, G. Submitted to *Env. Sci. Techn.*

VII. **Environmental Aspects of Energy and Material Recovery of Paper – Today and in a More Sustainable Future**
Finnveden, G. and Ekvall, T. Submitted to *Process Safety and Environmental Protection - Transactions of the Institution of Chemical Engineers - Part B*.

VIII. **A Market-Based Approach to Allocation at Open-Loop Recycling**
Ekvall, T. Submitted to *Resources, Conservation and Recycling.*

Contents - overarching document

Preface

This PhD thesis, like most other publications, is influenced by the background of the author. My personal background includes a BSc in physics. The field of physics appealed to me because of an early inclination towards abstract and analytical thinking. This inclination was, of course, reinforced by the basic academic studies of mathematics and physics. This is probably part of the reason I tend to prefer a rational paradigm, striving towards methodological solutions and knowledge with a frame-related robustness, *i.e.*, methods and knowledge which can be accepted relatively independent of the perceptions or interests of the observer (Tukker 1998, p. 31). Furthermore, my writing has a tendency to be more abstract than necessary, which sometimes makes it difficult to read. I have tried to alleviate this problem by, for example., including concrete examples in the text.

My personal background also includes the degree of licentiate of engineering in energy systems analysis (Ekwall 1991). My experience in this branch of systems thinking made me interested in the systemic aspects of LCA. In particular, it made me interested in analysing how the system of technological activities was modelled and should be modelled in an LCI. As a result of my experience in the field of energy research, I also have a tendency to focus on energy-related environmental burdens, such as the demand for fuel and electricity and emissions to air of CO_2, SO_X, NO_X and VOC. This is a tendency that many other LCA practitioners share for practical reasons, such as the availability of data. I am, of course, aware of the fact that the scope of environmental problems is much broader.

In my work on the PhD thesis, I have benefited from input and support from a large number of senior researchers, colleagues, friends, relatives, and so on. In particular, I am grateful to the following people:

- My former supervisor, Dr. Torbjörn Svensson, and my former boss, Göran Svensson, for introducing me to the exciting and fresh field of LCA.
- My subsequent supervisor, Dr. Anne-Marie Tillman, for co-authorship, for her active support and genuine interest, for a healthy number of inspiring discussions, and for her insights and persistence, which greatly improved the thesis – particularly the overarching document.
- My early colleagues, Dr. Tomas Rydberg and Dr. Henrikke Baumann, for initial discussions on system boundaries, for co-authorship, and for their continuous feedback.
- Dr. Reine Karlsson, Prof. Bengt Steen, Dr. Sven-Olof Ryding, and other members of the Swedish Product Ecology Project for stimulating discussions that provoked me into formulating my own version of the 50/50 method.
- Dr. Göran Finnveden, Anne Rønning, and the other experts working on the Nordic Guidelines on LCA for interesting discussions on system boundaries and allocation

in general and for their feedback on the 50/50 method in particular - Göran also for co-authorship and for companionship in the debate on wastepaper management.

- Dr. Sverker Molander, Dr. Greg Morrison, Dr. Åsa Jönsson, Dr. Ulf Östermark, and the two Drs. Karin Andersson for sharing their experience and knowledge, and for their advice on the procedural aspects of postgraduate research.
- My colleagues at Chalmers Industriteknik (CIT) and elsewhere for putting the 50/50 method into practical use.
- My colleagues at CIT and Chalmers University of Technology for their friendship and for providing a fertile environment for my work.
- other friends and colleagues in the international LCA community for many stimulating meetings and valuable pieces of information - important pieces to the jigsaw puzzle were supplied by (in order of appearance) Philippa Dobson, Henrik Wenzel, Dr. Rolf Frischknecht, Prof. Roland Clift, Dr. Adisa Azapagic, Dr. Gjalt Huppes, and Dr. Bo Weidema.
- Dr. Patrik Söderholm, Prof. Thomas Sterner, Dr. Gunnar Köhlin, and other economists for setting me straight in their field of expertise.
- My friend Frank Lorentzon for guidance in the domains of philosophy.
- My friends in the Saltstänk choir for bringing heaps of deep joy into my life.
- My children, Tove and Sara, for giving my life a deeper sense of purpose.
- My wife Inger for caring, for giving me the courage to initiate a second round of postgraduate research, and for taking on more than her share of the responsibilities at home, which made it possible for me to combine the PhD research and the contract research.
- Dr. Ingvar Andersson and Elin Eriksson at CIT for allowing me the time off to complete the thesis.
- The Swedish taxpayers for financing my research.
- Last, but not least, the Swedish Waste Research Council for allocating the money for my project.

Gothenburg, April 1999

Tomas Ekvall

1. Introduction

1.1 The history of LCA: a summary

Environmental life cycle assessment (LCA) was developed from the idea of comprehensive environmental assessments of products, which was conceived in Europe and in the USA in the late 1960s and early 1970s (Hunt & Franklin 1996). The concept of LCA is here defined (based on ISO 1997) as the compilation and evaluation of the material and energy flows and of the potential environmental impacts of the life cycle of a product. The product life cycle is here defined as the system consisting of models of the technological activities used for the various stages of the product: from extraction of raw materials for the product and for ancillary materials and equipment, through the production and use of the product, to the disposal of the product and of ancillary materials and equipment, if any. In this context, the term *product* is broadly defined to include not only physical products but also services.

Originally, LCA was used as a tool by environmental consultants. Eventually, it became clear that different LCAs carried through by different consultants resulted in different and sometimes conflicting conclusions (*e.g.*, ENDS 1991; Baumann 1995, p. 14). The large differences in the LCA results could be explained, in part, by different methodological choices (*e.g.*, Ekvall 1992, paper **IV**, paper **V**).

Many initiatives were taken to harmonise LCA methodology. These efforts resulted in methodological guidelines, most of which were valid for a specific geographical area (*e.g.*, Vigon *et al.* 1993, Lindfors *et al.* 1995a), for a particular category of products (*e.g.*, FEFCO-Ecobilan 1993, Strömberg *et al*. 1997, Ekvall *et al.* 1997), or for a particular application of LCA (Ryding *et al.* 1995, Wenzel *et al.* 1997, Hauschild & Wenzel 1998). General methodological guidelines were, in some cases, developed on the basis of consensus within a relatively small group (*e.g.*, Heijungs *et al.* 1992a-b). The various guidelines include different and often conflicting methodological recommendations.

An effort to reach consensus on a broad, international level was initiated within the Society of Environmental Toxicology and Chemistry (SETAC) in 1990 (Fava *et al*. 1991). This harmonisation process soon resulted in the so-called SETAC Code of Practice (Consoli *et al.* 1993). This document describes a procedural framework for LCA. It also includes some methodological recommendations but many, if not most, methodological problems remain unsolved in the SETAC Code of Practice.

The harmonisation process continues to proceed within SETAC. Since the Code of Practice was published, different working groups have been addressing different parts of LCA methodology (*e.g.*, Clift 1996, Udo de Haes 1996). In addition, a standardisation process started within the framework of the International Organization for Standardization (ISO). The latter has resulted, so far, in a standard for the procedural and methodological framework of LCA (ISO 1997) and a standard for the first parts of the LCA procedure: the goal and scope definition, and the life cycle inventory analysis (LCI; ISO 1998a). The international standards present recommendations or requirements for several methodological issues that were not covered in the SETAC Code of Practice.

The obstacles encountered in the harmonisation and standardisation of LCA methodology spurred the interest of the academic world. Development of LCA methodology and environmental research based on LCA methodology are relatively new academic topics, but the research volume grew rapidly at several universities during the first half of the 1990s. This is reflected by the increasing number of PhD theses that include results of this research (*e.g.*, Huppes 1993, Weidema 1993a, Rydberg 1994, Guinée 1995, Azapagic 1996, Schneider 1996, Young 1996, Fallscheer 1997, Horvath 1997, Blinge 1998, Bras-Klapwijk 1998, Karlsson 1998, Andersson *et al.* 1999, Carlsson-Kanyama 1999) - at least 15 PhD theses related to LCA were presented in 1998 alone. A scientific journal dedicated to LCA research (*The International Journal of Life Cycle Assessment*) was started in 1996. In addition, a large number of scientific papers on LCA have been published in other journals dedicated to environmental science, such as *Journal of Cleaner Production* and *Resources, Conservation and Recycling*, or to specific types of products, such as *Energy Conversion and Management*.

It was established at an early stage that the appropriate methodological choices depend on the purpose of the LCA (*e.g.*, Consoli *et al.* 1993, p. 9 & 21). Several attempts have been made to structure the various applications of LCA and to describe the connection between the study goal and the methodological choices that should be made in the LCA (*e.g.*, Heintz & Baisnée 1992, Weidema 1993a & 1998, Baumann 1996 & 1998, Frischknecht 1997, Cowell 1998, Hofstetter 1998, Tillman 1998 & 1999, Wenzel 1998a). These attempts provide a structure for further methodological discussions.

One type of methodological problem in LCA is the problem of allocation. Allocation can be defined as the partitioning of environmental burdens and other material and energy flows to and from a technological activity between the products for which the activity is used. Environmental burdens are here defined as the resource demand, the emission of pollutants, the waste generated, and the changes caused by land use of the technological system. Allocation generally becomes a methodological problem when a technological activity provides different functions for different products. The problem is to decide what share of the environmental burdens of the activity should be allocated to the product being investigated.

A large number of solutions have been suggested and applied to the allocation problems (*e.g.*, Boustead 1992, Heijungs *et al.* 1992b, Pedersen & Christiansen 1992, Vigon *et al.* 1993, Weidema 1993b, Boguski *et al.* 1994, Huppes & Schneider 1994, Rydberg 1995, Klöpffer 1996, Buhé *et al.* 1997, Kim *et al.* 1997, Anderson & Borg

1998, Frischknecht 1998, Newell & Field 1998, Wenzel 1998b, Karlsson 1998, Azapagic & Clift 1999, Lindeijer & Huppes 1999, paper **I**, paper **II**, paper **VIII**). It has been argued that the choice of approach to the allocation problems is arbitrary (Thomas 1977, p. 3; Spreng 1988, p.140). International standard ISO 14041 stipulates a specific procedure for dealing with the allocation problems (ISO 1998a); however, this procedure has been criticised because it does not take into account the fact that different approaches to the allocation problems result in different types of information (paper **VI**), nor does it take into account the relationship between the method and the study goal (*e.g.*, Baumann 1998, Tillman 1999, paper **II**). The same ranking order of methodological approaches to the allocation problems is recommended for all LCA applications.

1.2 Goal and scope of the postgraduate research

As a PhD student and as a contract researcher at Chalmers Industriteknik since 1991, I have aimed at contributing to the development of LCA theory and LCA practice. An LCA can be regarded as the construction and analysis of three interlinked models of the technosphere, the ecosphere and the valuesphere (Hofstetter 1998). The construction of the three models corresponds to different phases of the LCA procedure as described in ISO 14040 (ISO 1997): life cycle inventory analysis (LCI), classification/characterisation, and weighting. My research so far concerns primarily LCI methodology, *i.e.*, the modelling of the technosphere, which aims at the compilation and quantification of the relevant material and energy flows. The framework of the LCI methodology has been described elsewhere (*e.g.*, Hunt *et al.* 1993, Guinée *et al.* 1993). I have focused mainly on the allocation problems and the definition of the boundaries of the system investigated, but I have also touched upon other methodological issues, such as the choice of data sources.

Funtowicz & Ravetz (1991) distinguish between three levels of uncertainty that all occur in an LCA (Hoffman *et al.* 1995):

- Technical uncertainty: the lack of precision in the LCA results because of factors such as uncertainties in the data.
- Methodological uncertainty: the lack of reliability in the LCA results and the uncertainty as to whether an adequate LCA procedure and methodology have been used.
- Epistemological uncertainty: the uncertainty as to whether there are important environmental aspects that are completely unknown or that for other reasons cannot be included in the LCA.

My research has dealt primarily with the uncertainty as to whether an adequate LCI methodology has been used. The aim has been to reduce this methodological uncertainty. Technical and epistemological uncertainties are addressed very briefly in paper **I**, paper **V**, and paper **VIII**.

Bras-Klapwijk (1998, pp. 187-188) distinguishes between two different paradigms for the view of the function of studies in public decision-making processes: the rational paradigm and the discourse paradigm. She presents three characteristics for each paradigm. In the rational paradigm, decision-making is regarded as a rational process involving a single policy-maker or a group acting in agreement. An objective analysis is regarded as possible or something that should be striven for. And the analyses provide conclusions that can be adopted directly by policy-makers who translate these conclusions into policies. In the discourse paradigm, decision-making is regarded as a discourse process involving different governmental and social actors with different perceptions and interests. The conclusions of an analysis depend on these perceptions and interests, and this should be explicitly acknowledged. And conclusions based on such subjective perceptions and interests are considered to be valuable input to the discourse. As explained by Bras-Klapwijk (p. 188), there exists ”a continuum of ideas between the extremes of the rational and the discourse paradigm”. The distinction between the rational and discourse paradigms is probably also relevant for private decision-making, as long as there is more than one stake-holder.

Ideas close to the rational paradigm are predominant in the methodological development of LCA (Bras-Klapwijk 1998, p. 110). My research fits easily into the rational paradigm since it is an attempt to improve the objective, theoretical foundation for the methodological choices. The results can also be useful for discourses since my aim is to contribute to the development and application of an LCI methodology that results in knowledge with an improved frame-related robustness, *i.e.*, knowledge for which the acceptance is less dependent of the perceptions or interests of the participants in the discourse (Tukker 1998, p. 31).

My research is, however, based on the perception that an LCA practice is good if, and only if, it can be expected to result in environmentally sound individual decisions and actions. Decisions and actions are here perceived as environmentally sound if they result in a lower level of environmental burdens per functional unit than would have been the case without the LCA. The more the LCA practice can be expected to contribute to environmentally sound decision-making and actions, the better it is. In this context, a good LCA theory is a theory that provides a scientific basis for good LCA practice. It should be noted that the perception of sound decisions and actions is, in this case, based on the actual or expected consequences of the individual actions. This perception originates from a theory within normative, moral philosophy, which can be denoted situation ethics. According to an alternative theory – which can be denoted rule ethics – decisions and actions should be evaluated on the basis of whether they reflect good rules (Lübcke 1988).

I tend to write about the decision-maker as if it is a single individual or a group of decision-makers acting in agreement. I am aware of the fact that the concept of a single decision-maker or a group acting in agreement is often a simplification. Decisions are often the result of complex interaction between different individuals or groups with different perceptions and interests (Bras-Klapwijk 1998, pp. 187-188; Baumann 1998, pp. 14-16). However, I believe that the simplification is justified in this thesis, since the aim is to strive towards the generation of robust knowledge that can be accepted as valid by all participants in a decision-making process.

The aim of my postgraduate research has been to build upon the findings and experience I have made as a contract researcher, to put the findings in a broader, theoretical context, and to formulate the findings in a scientific way. The aim to contribute to the improvement of LCA theory was operationalised in my postgraduate research into four research tasks:

- Determine the consequences different system boundaries and allocation methods have for LCA results (conclusions are presented in paper **III**, paper **IV**, paper **V**, and paper **VII**) and what information is conveyed to decision-makers by LCIs with different approaches to the allocation problems (paper **I**, paper **IV**, paper **V**, and paper **VI**).
- Establish how the goal definition may be used to guide the choice of system boundaries and allocation method (paper **I**, paper **II**, paper **IV**, paper **V**, and paper **VI**).
- Refine the 50/50 allocation method for open-loop recycling – that is, when material or energy from one product life cycle is recycled into another – that was presented in an earlier paper (Ekvall 1994) with the aim of accurately taking into account the effects of recycling (paper **VIII**).
- Reduce methodological uncertainty in the environmental comparison between recycling and waste incineration of wastepaper, old corrugated board, and similar materials (paper **IV**, paper **V**, and paper **VII**).

The fourth research task was added to the aim of the postgraduate research because several of the existing case studies that I analysed as a contract researcher (*e.g.*, Ekvall 1992, Ekvall 1996) or as a PhD student (paper **IV**, paper **V**) dealt with paper or paperboard products. These case studies were, in many cases, carried through in order to compare different waste management methods for these materials. The choice between recycling the material and waste incineration with energy recovery was often the focus of these studies. Different studies resulted in different conclusions, which gave rise to a somewhat heated debate in Sweden (*e.g.*, Bruvoll & Ibenholt 1998a-b, Ekvall 1998, Robért *et al.* 1998) as well as in other countries (*e.g.*, Pearce 1997), possibly because the choice between recycling and incineration of these large material flows can be important from both an economic and an environmental perspective. The analyses of the existing case studies resulted in conclusions regarding which choices in the LCI methodology are important for the comparison between recycling and incineration of wastepaper, old corrugated board, and similar materials. They also resulted in some conclusions regarding which methodological choices are relevant for such a comparison.

1.3 Methods

The theoretical research tasks were carried out through analyses of existing case studies, literature studies, discussions with other researchers, and theoretical reflection. The papers in this thesis present methodological results, but they are based on case studies that are published elsewhere (see Figure 1). The existing case studies,

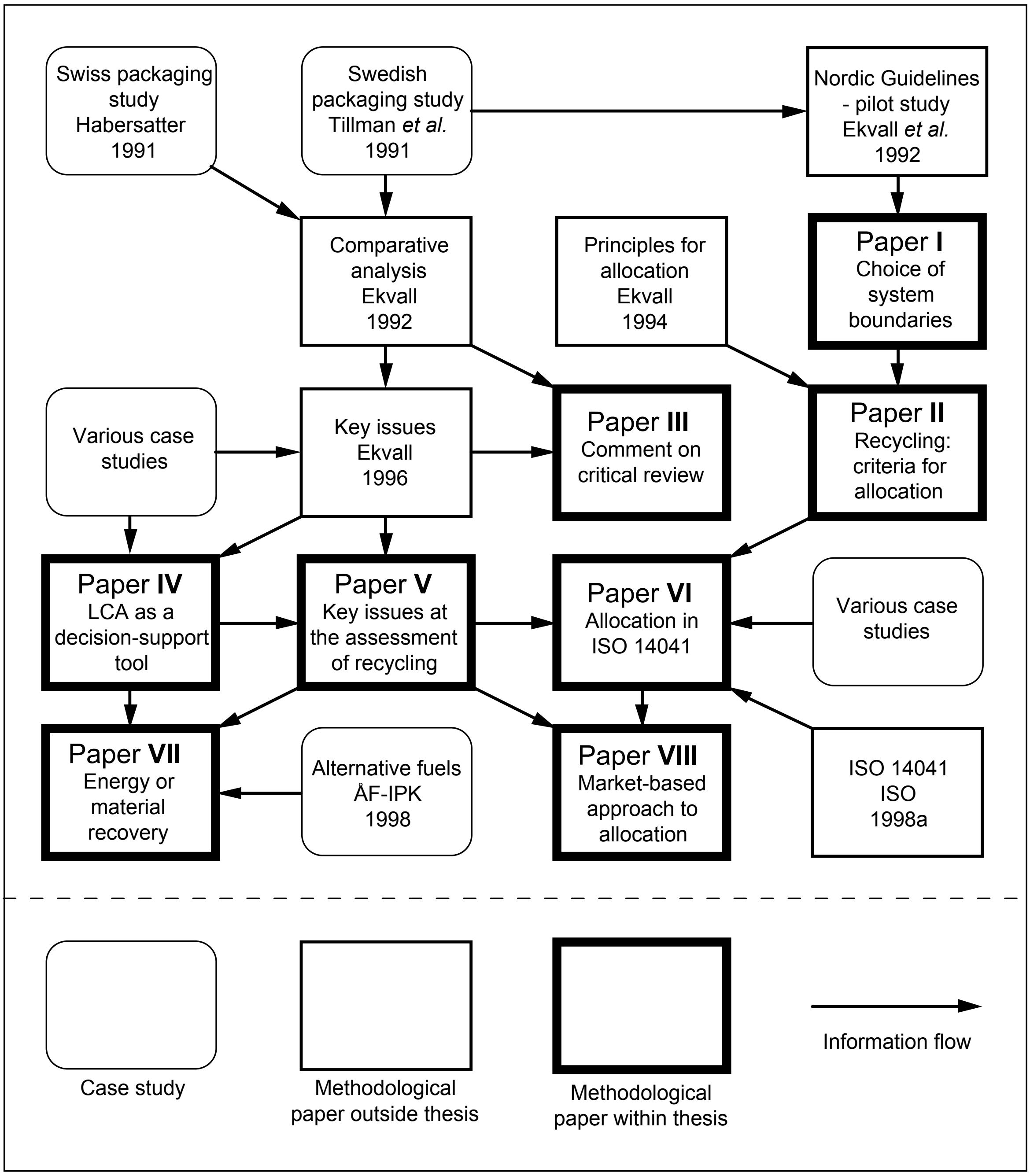

Figure 1. Simplified illustration of the main sources of the papers in this thesis. Papers to the left in the figure are generally oriented towards wastepaper management, while papers to the right to are more concerned with LCA in general.

employing different methodologies, were analysed to investigate the applicability of different methodological choices and the effects on LCA results. Literature studies and discussions with other researchers were carried out to gather relevant information on contemporary LCA methodology and other scientific areas. The research tasks are of a multi-disciplinary nature and involve elements of fields such as systems analysis, economics, management theory, and philosophy. Theoretical reflection was needed to conceive further methodological developments and to analyse the implications of different approaches to the allocation problems, such as what information is conveyed by LCAs that are carried through with different approaches to the allocation problems.

Contribution to the LCA practice was made through the theoretical research and through the dissemination of the findings. I carried out part of the dissemination work as a PhD student, but most of it was carried out as part of my activities as a contract researcher. One important part in the dissemination process was to participate in various harmonisation processes. Before the postgraduate research began, I participated in the development of the Nordic Guidelines on LCA (Ekvall *et al.* 1992, Lindfors *et al.* 1995a-b) and the Swedish Product Ecology Project (Ryding *et al.* 1995). During the time of my postgraduate research, I participated in the European LCANET project (Frischknecht 1997) and in various other harmonisation processes as a PhD student or as a contract researcher at Chalmers Industriteknik (Ekvall *et al.* 1997, Strömberg *et al.* 1997, Swan 1997, Ekvall *et al.* 1998, de Groot *et al.* 1998, Guinée *et al.* 1998), I also contributed with feedback to the harmonisation process within the SETAC-Europe working group on enhancement of inventory methodology (Clift 1996, Clift *et al*. 1999) and to the international standardisation of LCI methodology (ISO 1998a). Currently, I participate in the continuing harmonisation processes within the SETAC-Europe working group on scenario development (Fleischer 1998) and COST Action E9 (COST 1997). As a contract researcher, I have also been able to utilise and disseminate the theoretical findings as, for example, a project manager (*e.g.*, Sjöberg *et al.* 1997, Ekvall *et al.* 1998) and a reviewer (*e.g.*, Brännström-Norberg *et al.* 1996) of case studies. The theoretical findings have also been disseminated through the public debate on recycling and waste incineration (*e.g.*, Ekvall 1998, Robért *et al.* 1998). All of these activities involved a mutual exchange of knowledge and ideas with LCA researchers and other experts; hence, these activities have been as much a part of the learning process as they have of the dissemination process.

1.4 Goal and scope of the overarching document

This text, which precedes the papers in the thesis, is not intended to be a comprehensive summary of the research presented in the papers. In particular, the assessment of recycling and incineration of paper and paperboard products is not adequately summarised in this text. For a summary of this research, I refer to paper

VII. The aim of this text is to supplement the papers by putting my theoretical findings into a broader context, validating my approach to allocation in open-loop recycling, and presenting further conclusions based on the research results and experience; hence, I chose to call this text an overarching document rather than a summary.

The overarching document includes a discussion of the allocation problems (Chapter 2) and a review of a selected number of approaches to the allocation problems. The review covers most of the approaches to multi-function processes (Chapter 3), but only a selected number of approaches to open-loop recycling (Chapter 4). The selection was made with the intention to provide an adequate background for the validation of my own, market-based approach. The review in Chapters 3 and 4 is, to a large extent, based on paper **VI**; however, the description of the market-based approach (Section 4.5) is based on paper **VIII**.

The overarching document also includes a discussion of general and goal-dependent requirements on LCI methodology (Chapter 5). This discussion is based on paper **II** and paper **V** but includes several additional arguments and conclusions. The new arguments include a discussion of the methodological consequences of a radical application of the requirement to generate information about the effects of possible actions (Section 5.2). It also includes a discussion of the various applications of LCA and how these can be classified and of the applicability of different LCI methodologies (Sections 5.5-5.6).

The validation of the market-based approach is presented in Chapter 6. It is based on the perception that an LCA practice is good to the extent that it can be expected to result in environmentally sound individual decisions and actions. Empirical verification of whether a methodological choice results in environmentally sound decisions and actions is not feasible; instead, the validation is based on the theoretical discussion of the requirements on the LCI methodology (Chapter 5). The validation includes a discussion of the following issues:

- Completeness: whether the approach takes important and relevant mechanisms into account.
- Clarity: whether the concepts and mechanisms that are used in the approach are well defined and easy to understand.
- Acceptability: whether the method can be accepted as relevant by decision-makers.
- Feasibility: whether the approach is feasible and easy to apply.
- Applicability: the limitations of the applicability of the approach.
- Precision: the expected precision of the results that are obtained through this approach.

Finally, the overarching document includes a summary of the conclusions presented in the thesis. This summary is presented in Chapter 7.

2. Allocation problems

In the current international standard, ISO 14040, allocation is defined as the partitioning of material and energy flows to or from an activity to the product system under study (ISO 1997, p. 5); however, environmental burdens other than material and energy flows, such as land transformation, can also be partitioned in an allocation.

Before an allocation can be performed, it must be decided which material and energy flows represent burdens – which should be allocated – and which material and energy flows represent functions – to which the burdens should be allocated. Functions include the production of products (including materials, energy carriers, and services) and the processing of waste. In the context of LCA, the distinction between a waste flow and a flow of products is often based on the economic value of the flow: a flow with a positive economic value is a product flow, and a flow with a negative economic value is a waste flow (Heijungs *et al.* 1992b, pp. 23-24; Huppes 1993, pp. 187-189; Frischknecht 1998, pp. 95-96). Based on this distinction, functions are represented by outflows of material and energy with a positive economic value (products) and inflows with a negative economic value (waste). Burdens are represented by outflows of material and energy with a negative economic value (emissions and waste) and inflows with a positive economic value (natural resources, raw materials, intermediate products, etc.).

The validity of the economic distinction between products and waste flows can be discussed. The European Union defines waste, without referring to the economic value, as a substance or object that is included in one of several, specified waste categories and that is the subject of recycling or waste disposal (RVF 1998); however, my research has not dealt with this part of the allocation problem. In the following, it will simply be assumed that it has been decided which burdens should be allocated and which material and energy flows represent functions to which the burdens should be allocated. The remaining part of the allocation problem is then to decide which share of the burdens should be allocated to the system being investigated.

A multi-function process is here defined as a process that fulfils more than one function. It may be a production process with more than one product, a waste management process dealing with more than one waste flow, or a recycling process providing both waste management and material production. An allocation problem arises when a multi-function process fulfils one or more functions in the system investigated and a different function, or set of functions, in other systems. The problem is to decide which share of the environmental burdens of the process should be allocated to the system investigated, that is, included in the LCI of the system investigated. Heijungs & Frischknecht (1998) defined the allocation problem in mathematical terms.

A special allocation problem arises in open-loop recycling. Here, the primary production, recycling, and final waste management of the material in the life cycle of the product investigated fulfil functions for other product life cycles as well. The methodological problem is how to allocate the environmental burdens of these processes between the cascade products, *i.e.*, the products that, through recycling, utilise the material in the life cycle investigated (paper **II**).

The two types of allocation problems have several aspects in common. Multi-function processes and open-loop recycling are both cases of multi-function systems (Azapagic 1996, p. 30; Azapagic & Clift 1999). More specifically, the primary production, recycling and final waste management activities involved in open-loop recycling are multi-function processes since they provide different product life cycles with different functions: primary material to the first life cycle in the cascade, and recycled material to other life cycles. Both types of allocation problems also arise because of the export of functions (paper **VI**). Exported functions are functions that are generated in one product life cycle but utilised in another product life cycle. Exported functions may include the functions of co-products that are used in other life cycles, the processing of waste from other life cycles, and the raw material supply and waste management of recycling processes.

It may even be difficult to distinguish one type of allocation problem from the other (Finnveden 1994). A recycling process may be regarded either as a multi-function process or as a part of the open-loop recycling activities. Waste incineration with energy recovery can be regarded either as a multi-function process or as a case of open-loop recycling (Frischknecht 1998, p. 106).

In spite of the similarities, the two types of allocation problems are separately dealt with in this thesis, because different approaches to the allocation problems are feasible and relevant for the different problems (paper **VI**). One reason for this difference is that the case of open-loop recycling requires the co-ordinated allocation of environmental burdens from different processes – primary material production, recycling, and final waste management – that are often widely separate in time and space and that are under the influence of different sets of decision-makers (Frischknecht 1997, p. 77; Frischknecht 1998, pp. 97-99; paper **II**). In the following, the concept of multi-function processes is used to denote multi-function processes other than the ones involved in open-loop recycling.

In the context of LCA, the concept of allocation is used to denote a part of the modelling of the technological system. Outside the area of LCA, such as in the area of economic planning, the concept of allocation is sometimes used in another sense. Allocation of certain resources can denote the decision to set these resources apart for a particular purpose (Anon. 1995). Allocation in the sense it is used in LCA is not a necessary part of an economic assessment of a system, even if the system includes activities that fulfil functions for other systems. The modelling of the system can be carried out based on the total, unallocated costs and the total earnings (Frischknecht

1998, p. 100), because the economic value of the exported functions – *i.e.*, the functions supplied to other systems – are defined by their price. If the exported functions had a well-defined environmental value, allocation would not be necessary in LCAs or similar environmental assessments. In fact, the act of allocation in an LCA can be regarded as an attempt to assign an environmental value to the exported functions (Frischknecht 1998, p. 100). This environmental value is multi-dimensional since the environmental burdens allocated are described through a set of parameters.

3. Existing approaches: multi-function processes

Several methodological approaches have been presented for dealing with the allocation problem that occurs in multi-function processes. Most of them are at least mentioned in this chapter. The allocation procedure presented in the international standard for LCI - ISO 14041 - is reviewed in paper **VI**. Various methodological approaches are also reviewed in other publications (Lindfors *et al.* 1995c, Strömberg *et al.* 1997) and by other authors (*e.g.*, Heijungs *et al.* 1992b, Weidema 1993b, Azapagic 1996).

3.1 ISO 14041

This section essentially presents a summary of the review in paper **VI**. The international standard requires that the following approaches be considered in the following order of preference (ISO 1998a, p. 11):

- Allocation should be avoided, wherever possible, either through division of the multi-function process into subprocesses and collection of separate data for each subprocess, or through expansion of the systems investigated until the same functions are delivered by all systems being compared.
- Where allocation cannot be avoided, the allocation should reflect the physical relationships between the environmental burdens and the functions, that is, how the burdens are changed by quantitative changes in the functions delivered by the system.
- Where such physical, causal relationships alone cannot be used as the basis for allocation, the allocation should reflect other relationships between the environmental burdens and the functions.

I have found no case study in which an allocation problem was completely eliminated through division of the multi-function process. This is not very surprising since the allocation problem can be eliminated through subdivision only when the multi-function process consists of single-function subprocesses and when environmental data can be obtained for each of these subprocesses (see Figure 2). This, in turn, requires that the subprocesses are physically separate in space or time, such as in batch production of different products. When it is possible, subdivision contributes to a more accurate picture of the consequences of any action that affects the production volume of functions used within the life cycle, while the production volume of exported functions remains unaffected; however, if the internally used function is significantly affected, the exported functions will be unaffected only if the subprocesses are economically independent of each other as well as physically separate.

The international standard states that the recommendation to divide the multi-function process into subprocesses applies not only when subdivision eliminates the allocation problem but also when it reduces the allocation problem (ISO 1998a, p. 19). This recommendation was presented earlier by authors such as Huppes (1993, pp. 209-211) and Knoepfel (1994). It has been used in several case studies to reduce the allocation problems that occur with multi-function processes such as combined extraction of crude oil and natural gas (Knoepfel 1994), diesel production at a refinery (Furuholt 1995), a sawmill (Erlandsson 1996), a metal smelter (Sunér 1996), and dairies (Høgaas Eide & Ohlsson 1998). If the approach to the residual allocation problem is based on physical relationships between the environmental burdens and the functions, this combined approach will contribute to a more accurate picture of the consequences of any action that affects the production volume of internally used functions, while the production volume of exported functions will remain unaffected. If the residual allocation is based on other relationships, the combined approach results in an inconsistent blend of different types of information. For further details on subdivision, see Section 2.1 in paper **VI**.

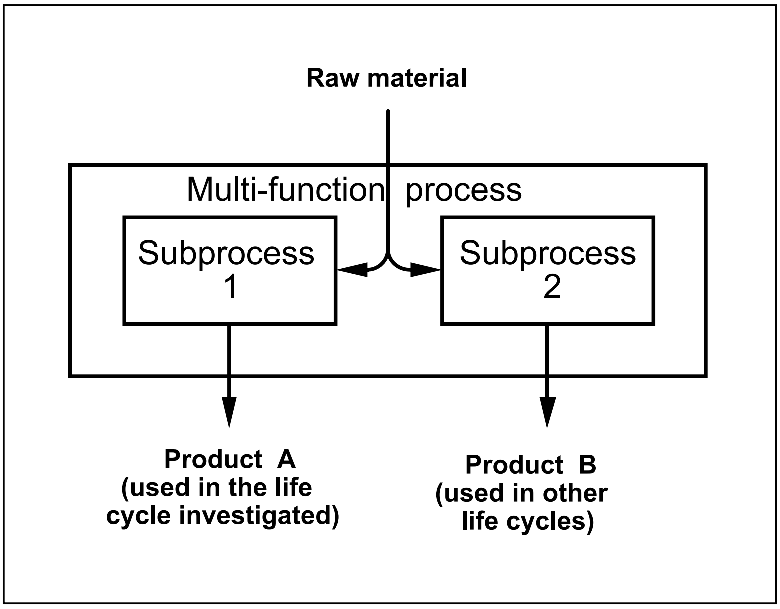

Figure 2. A multi-function process that consists of two single-function subprocesses.

System expansion was discussed at an early stage by, for example, Heintz & Baisnée (1992). Allocation has been avoided through system expansion in many case studies (*e.g.*, Pedersen 1991, Tillman *et al.* 1991, paper **VI**). These demonstrate that system expansion is possible for a wide range of allocation problems and LCA applications. In principle, system expansion requires only that there is an alternative way of generating the exported functions and that data can be obtained for this alternative production (see Figure 3).

When the production volume of exported functions is changed, the environmental burdens of activities outside the life cycle of the product investigated are likely to be affected. These effects are denoted in the following as indirect effects. System expansion means that the indirect effects can be taken into account (Ekvall *et al.* 1992, paper **I**). If the system expansion is properly conducted, it results in a more comprehensive picture of the consequences of actions that affect the production volume of both internally used and exported functions. Unfortunately, system expansion is often based on inaccurate data and assumptions, which are used to model the effects of actions on the production volume of exported functions as well as the indirect effects of changes in the exported functions.

System expansion requires the collection and processing of additional data. This extra work is justified only when the system expansion can be expected to result in information that is significant for the conclusions of the LCA. This is likely to be the case if the indirect effects are important enough to be significant for a decision and if the uncertainties in the indirect effects are not too large. For further details on system expansion, see Section 3 in paper **VI**.

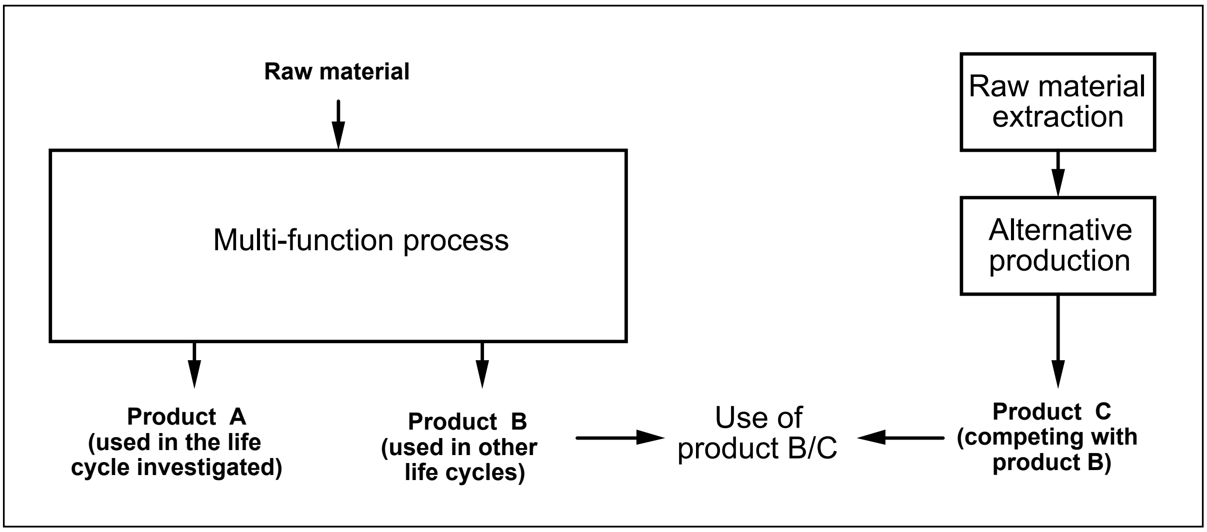

Figure 3. A multi-function process that produces a co-product (B) that competes with another product (C). System expansion in this case means that the production of product C is included in the system investigated. The use and waste management of products B and C can also be included in the expanded system if the environmental burdens from the use and waste management of product C are different from the environmental burdens from the use and waste management of product B.

Different interpretations of the ISO recommendation to allocate based on the physical relationships between environmental burdens and functions are possible. One interpretation is that the environmental burdens allocated to a function (Q_i in Figure 4) should be the burdens avoided if that function is no longer delivered while the other functions are unaffected ($\mathbf{B}$-$\mathbf{B}_0$). Such an allocation is feasible only when the functions can be independently eliminated. The vector notation is used here because the environmental burdens of the multi-function process ($\mathbf{B}$) are described through a set of parameters.

Another interpretation of this approach is that the environmental burdens allocated to each of the functions should reflect how the environmental burdens of the process are changed by marginal changes in the functions delivered by the process. In other words, the allocation should be proportional to the partial derivatives at the point of operation ($\partial\mathbf{B}/\partial Q_i$; Azapagic & Clift 1994, Azapagic 1996, Azapagic & Clift 1999). With this interpretation, the approach requires only that marginal changes in the functions can be independently made without a discrete change in technology or other discontinuities. The environmental burdens can be calculated, for example, through the use of linear programming.

This method is not applicable when the partial derivatives at the point of operation are not defined. This is the case if the functions are produced in fixed ratios (Azapagic & Clift 1994), if the point of operation is at the capacity limit of total production in the process (Clift *et al.* 1999), or if the ratio between any set of functions is maximised at the point of operation. A possible example of the latter is combined heat-and-power production, where the amount of electricity produced per GJ of heat may be maximised for that technology. Another possible example is waste incineration with energy recovery, where the amount of energy produced from each tonne of waste may be maximised for that technology.

Just like subdvision, this allocation approach will contribute to a more accurate picture of the consequences of any action that affects the production volume of internally used functions, while the production volume of exported functions remains unaffected. For further details on allocation based on physical, causal relationships, see Section 4 in paper **VI**.

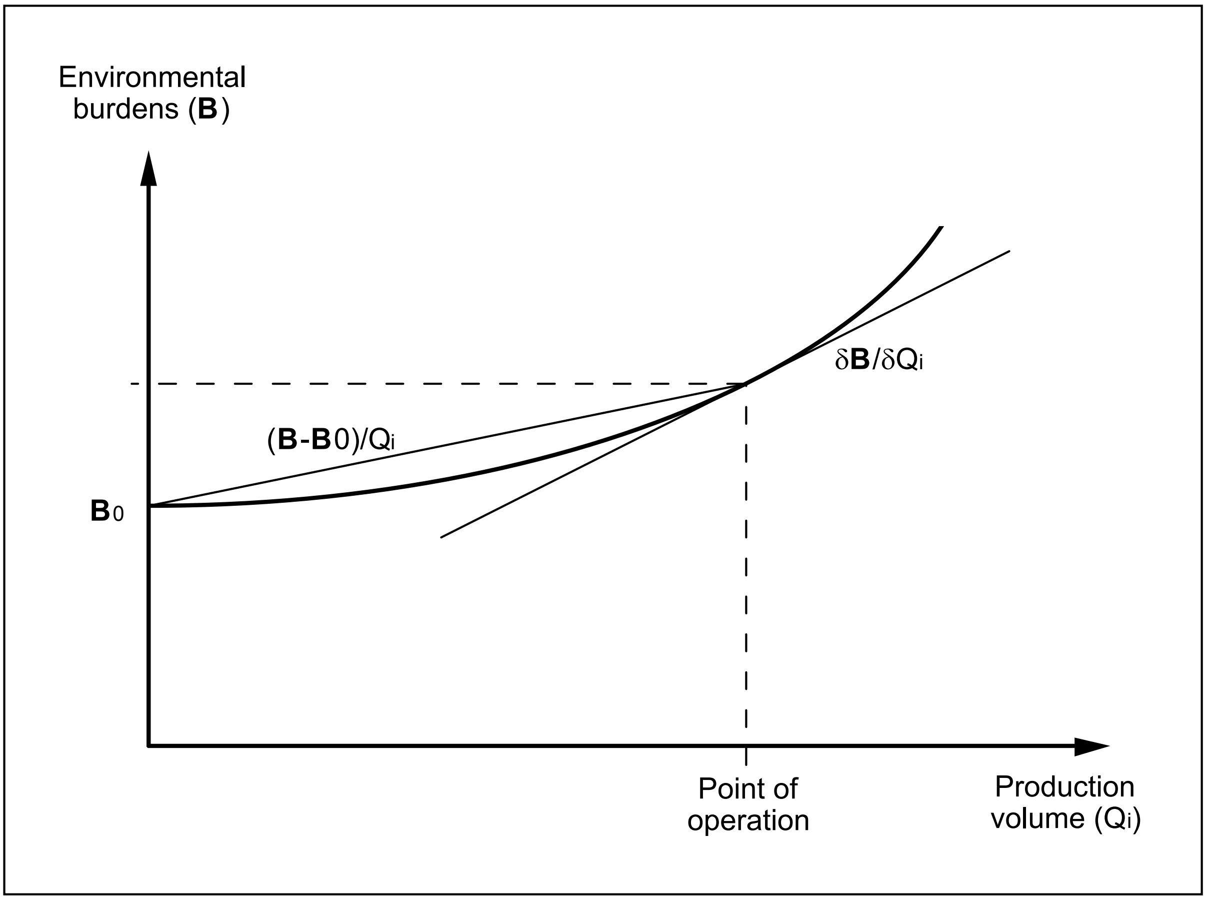

*Figure 4. Illustration of two different interpretations of allocation based on physical relationships. The environmental burdens vector (**B**) of a multi-function process is a function of the production volume of the different functions provided by the process. The graph illustrates how the environmental burdens vector may depend on the production volume of one of the functions (Q_i) if the production volume of the other functions is constant.*

The last option in the ISO procedure allows for allocation in proportion to the economic value of the products or functions. Allocation based on the gross sales value of the functions was recommended at an early stage by, for example, Huppes (1993, pp. 209-211). The motive is that the multi-function process exists because of the expected economic proceeds. This approach provides information about the causes of the activity and its environmental burdens, but not about the consequences of possible actions (paper **II**).

If it is generously interpreted, the last option in the ISO procedure also allows for allocation based on relationships that are not causal. This includes allocation in proportion to a physical property of the products – such as mass, energy content, exergy content, volume, area, or molar content – without reference to the causal relationships. In some cases, this allocation may coincide with allocation based on physical, causal relationships. However, when the allocation is not based on an accurate model of causal relationships, it will not provide reliable information on the consequences of our actions. For further details on allocation based on other relationships, see Section 5 in paper **VI**.

3.2 Frischknecht

Frischknecht (1998) presents three allocation methods for joint production processes, *i.e.*, multi-function processes in which different co-products are produced in fixed proportions. A distinction is made between joint production in which all co-products are under the control of the same decision-maker(s) and joint production in which different decision-makers control different co-products (Ibid. p. 99). In the first case, Frischknecht suggests two different allocation methods, depending on whether any feasible change in the production rate of the firm has a significant effect on the price of the co-products (Ibid. pp. 100-115). Both methods are developed from existing allocation methods in cost accounting. They are based on the ability of the products to bear the responsibility for environmental burdens and still be competitive. They require that the environmental burdens are quantified in monetary units; the competitiveness is measured in terms of comprehensive prices, which is defined as the sum of production costs, profits, and environmental costs. In this sense, the allocation is a combined economic and environmental evaluation of the multi-function process as a whole, compared to the production of the competing products. This ”enviro-economic competitiveness” approach results in information about which investments to make in order to deliver the functions at the lowest comprehensive prices. When the products compete in the market on the basis of the comprehensive price, this approach can also provide information about the indirect, ”enviro-economic” effects of more general actions.

The enviro-economic competitiveness approach is demonstrated for the case of combined heat-and-power production (Frischknecht 1998, pp. 165-196). Investment decisions about such a plant may be based on comprehensive prices as defined by Frischknecht. A problem with this approach is that investment decisions and competition in the market can, in many other cases, be based on ordinary prices, quality aspects, etc. and not on the comprehensive prices. Another problem is the large uncertainties involved when environmental burdens are quantified in monetary units (*e.g.*, Baumann & Rydberg 1994). A third problem is that the information generated through this combined economic and environmental evaluation is inconsistent with other information typically generated in an LCA. Frischknecht aims at integrating the economic and environmental assessments, but an LCA is generally considered to be an environmental assessment only. This does not, of course, prevent LCA practitioners from using economic concepts and mechanisms in the quantification of environmental burdens (Huppes 1993, paper **VIII**) and, in spite of the problems involved, in the aggregation of different environmental impacts into a single environmental index (Steen & Ryding 1993, Steen 1996).

When different decision-makers control different co-products, the environmental burdens are allocated to the co-products in shares that are defined after negotiations between the decision-makers who control the co-products (Frischknecht 1998, pp. 115-118). The aim of the negotiation is to obtain a distribution of the social costs – that is, the sum of the production and environmental costs – that the decision-makers can accept as fair.

3.3 Ekvall & Finnveden

The review of the ISO procedure (paper **VI**) resulted in a suggestion for a revision of the procedure. This suggestion is valid if the purpose of LCA is to increase our ability to anticipate the environmental consequences of our actions:

- Use the most easily applicable allocation method when the choice of allocation approach is expected not to be important for any decision that is based on, or inspired by, the LCA results.
- When the allocation can be important for a decision but the effects on exported functions are not expected to be important for any decision, avoid allocation through subdivision, allocate based on the physical, causal relationships between the functions and environmental burdens, or use an adequate approximation thereof.
- When the effects on the exported functions can be important for a decision, use system expansion or an adequate approximation thereof.

When the effects on the exported functions can be significant, but the uncertainties are too large to make system expansion feasible, the LCA practitioner should clearly state that a course of action might have important but unknown indirect effects on other life cycles.

4. Existing approaches: open-loop recycling

Many different approaches have been presented to the allocation problem caused by open-loop recycling of material or energy. Several of these have been reviewed in other publications (Ekvall 1994, Lindfors *et al.* 1995c, paper **II**, paper **VI**) and by other authors (*e.g.*, Klöpffer 1996, Schneider 1996, Fallscheer 1997). This chapter presents an overview of methodological approaches that are relevant for the validation of the market-based approach presented in Section 4.5.

4.1 ISO 14041

The international standard states that the procedure presented in Section 3.1 also applies to open-loop recycling (ISO 1998a, p. 12). However, subdivision is not effective in reducing this allocation problem, because the subprocesses of the primary material production and final waste management fulfil a function for all of the cascade products (paper **VI**). Allocation based on physical causality is not applicable for open-loop recycling, because the causal relationship between the functions and the environmental burdens of raw material extraction and final waste management is economical instead of physical (see Section 4.2).

System expansion can be applied when the allocation problem is caused by an inflow of recycled material to the life cycle of the product investigated, as long as data can be obtained for the alternative fate of the material. It can also be applied when the allocation problem is caused by an outflow of recycled material, as long as the recycled material competes with other materials and data can be obtained for the production of the competing materials. System expansion potentially contributes to a more comprehensive picture of the consequences of any action that will affect the flows of recycled materials. Unfortunately, system expansion is often based on inaccurate assumptions regarding which materials are replaced (paper **VI**). Technical report ISO TR 14049 indicates that recycled plastics and aluminium should be assumed to replace primary material of the same type (ISO 1998b). Karlsson (1995, pp. 59-61; 1998, pp. 33-35) presents a method for calculating the environmental value of recycled material from the product investigated that is based on the assumption that the recycled material replaces primary material. Fleischer (1994, Fleischer & Schmidt 1995 & 1996) argues that the alternative production must be based on primary material since the system investigated does not include alternative sources of recycled material. However, it is reasonable to expect that the recycled material may, in part, replace recycled material from other sources (paper **VI**). It may also replace other types of material (Ekvall & Wenzel Christensen 1993, Mølgaard 1995, Fletcher & Mackay 1996) or no material at all.

The international standard allows for a few additional options for dealing with this allocation problem (ISO 1998a, p. 12). If recycling does not cause a change in the inherent properties of the material, the allocation may be avoided by calculating the environmental burdens as if the material were recycled back into the same product. Otherwise, the allocation can be based on physical properties, economic value, or the number of subsequent uses of the recycled material. None of these approaches results in reliable and accurate information on the consequences of our actions (paper **VI**).

4.2 Linear programming

Azapagic & Clift (1999) demonstrate that linear programming (LP) can be used not only for multi-function processes but also for open-loop recycling. In the latter case, the environmental burdens are calculated from an LP model that includes all products into which the material is recycled; hence, it can be regarded as a case of system expansion.

Azapagic & Clift (1999) indicate that the LP model results in an allocation based on physical, causal relationships. In fact, the LP model is defined by several constraints, some of which reflect physical constraints while others are based on assumptions concerning factors such as how the production of other products is affected by an increased flow of recycled material from the product investigated. This effect depends on market mechanisms and not on physical, causal relationships. In general terms, the LP model results in an allocation based on the constraints of the model, regardless of what these constraints reflect - physical or economic causal relationships, assumptions etc.

The LCI results can reflect the consequences of actions that affect the flows of recycled materials; however, this requires that the LP model be based on accurate information regarding market mechanisms. This information is as important to the accuracy of the LP model of open-loop recycling as it is to system expansion in general (see Section 4.1).

4.3 Quality degradation

When material becomes less useful as the result of open-loop recycling, Karlsson (1995, pp. 102-105; 1998, p. 38) suggests that the environmental burdens of primary material production and final waste management should be allocated to the product investigated in proportion to the loss of ”usefulness” between the material in the product investigated and the product in which the material is subsequently used. Similarly, Wenzel (1998b) proposes that the environmental burdens should be allocated in proportion to the loss of ”material grade”. If usefulness and material grade are measured in terms of physical properties, economic value, or the number of subsequent uses of the recycled material, this approach is consistent with the international standard (ISO 1998a, p. 12). It is based on the recognition that man-made materials are valuable resources and that primary material production and final waste management are both necessary to obtain this resource.

Quality reduction can affect the environmental consequences of recycling. For instance, if open-loop recycling of a material into another product means that the quality of the material in that product is reduced, this may affect the environmental burdens of the production process and the recyclability of the subsequent product. It should be emphasised that these effects are not necessarily proportionate to the quality reduction; hence, allocation in proportion to quality losses does not provide reliable information about the environmental consequences of the quality reduction.

4.4 50/50 approaches

When material is recycled from one product into another product, the disposal of the first product and the production of primary material to the second product are less than if there had been no recycling. On the other hand, there is a need for a recycling process, and the use of recycled material in the second product may affect the environmental burdens of other activities in that life cycle. Vigon *et al*. (1993, pp. 87-91) present three different approaches to open-loop recycling. In one of these, the effects of recycling are equally distributed between the two products. This 50/50 method is only defined for the case in which the material is used in two products. It is not stated how the environmental burdens should be allocated if the material is used in three or more products before final disposal.

A slightly different 50/50 method is described in an early conference paper (Ekvall 1994) and in several subsequent publications (Lindfors *et al.* 1995a&c, paper **II**, paper **V**). Here, the environmental burdens of primary production and final waste management of the recycled material is equally distributed between the first and the last of the products in the cascade, that is, between the product in which the primary material is used and the product where the material is lost from the technosphere. The environmental burdens of each recycling process are allocated in equal shares between the adjoining upstream and downstream products, that is, between the product that is collected and sent to the recycling process and the product in which the material from the recycling process is used. This method is defined for cascades with any number of products. In this 50/50 method, the allocation concerns the environmental burdens of the *actual* primary production and waste management, while the allocation in the 50/50 method of Vigon *et al.* concerns the primary production and waste management that is *avoided*.

4.5 Market-based approach

Paper **VIII** presents a methodological approach that can be used for allocation as well as for generating the information that is required for system expansion. It is based on a conceptual model of the market for recycled material, which connects the life cycle of the product investigated to other life cycles (Figure 5). This approach aims at describing the indirect effects of a change in the flow of recycled material to (Y) or from (X) the life cycle of the product investigated. The indirect effects depend on how the market reacts to the change, and this reaction depends on political constraints, price elasticities, and so on. If the market for recovered material is sufficiently free for the recycling rate to be decided by economic forces, an increase in the quantity supplied by the product life cycle investigated is likely to result in a reduction in the price (P) of the recovered material. The effects of this price reduction depend on how sensitive the collection for recycling of other products (S) and the demand for recovered material (D) are to changes in the price. When authorities decide the recycling rate, the indirect effects of open-loop recycling depend on the details of the regulation.

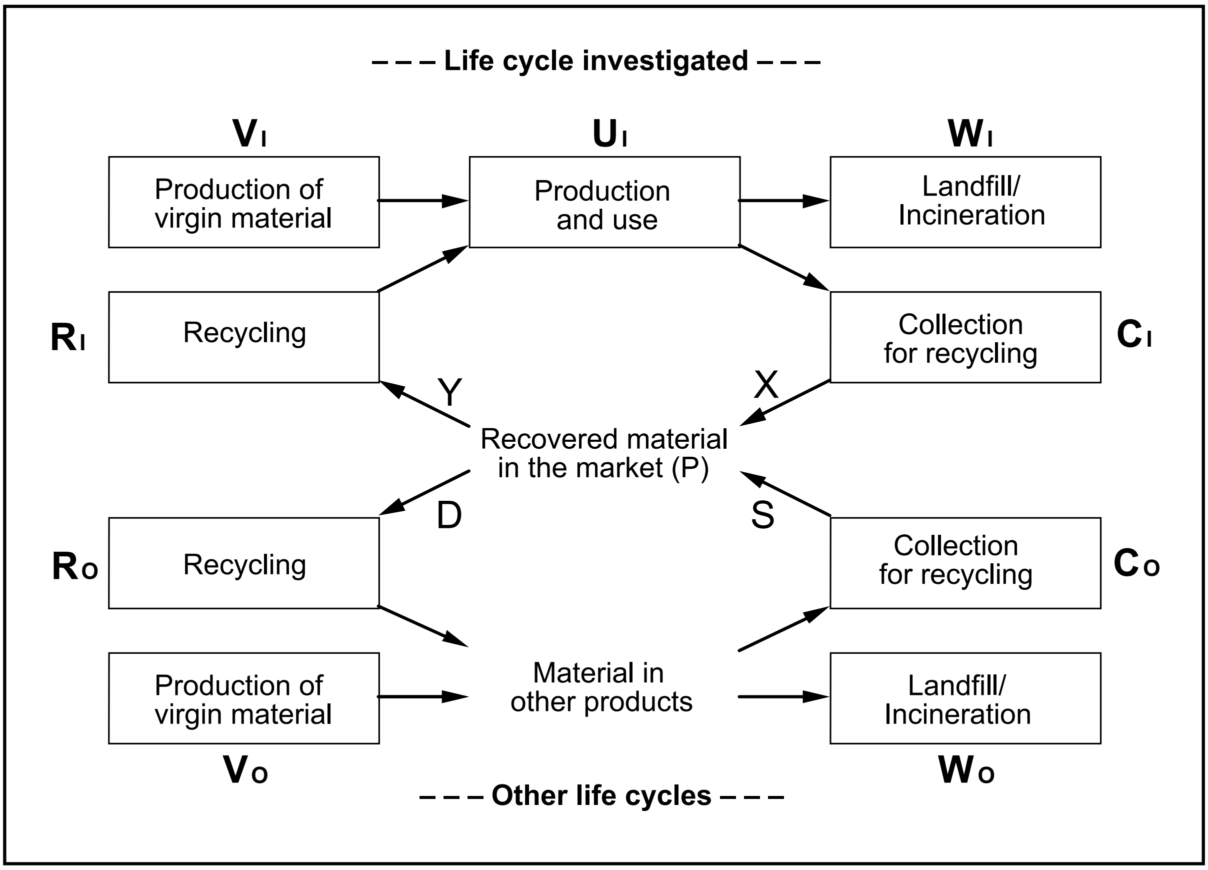

Figure 5. The conceptual model of the life cycle investigated as connected to other life cycles through the market for recycled material.

5. Requirements on LCI methodology

Three general requirements on LCI methodology are presented and discussed in this chapter: generation of knowledge, acceptability, and feasibility. These requirements are also discussed in paper **II**. The requirement to generate knowledge is elaborated on with a discussion of the implications for system boundaries and for the choice of data and with a discussion of the methodological consequences of a radical application of this requirement. In addition, goal-dependent requirements are discussed in the second half of the chapter.

5.1 General requirement I: generation of knowledge

Comprehensive and clear information

One obstacle to the harmonisation and standardisation of LCA methodology is that the information an LCA should provide is not clearly specified. According to the early SETAC definitions, an LCA should provide information about the environmental burdens associated with a product, process or activity (Fava *et al.* 1991, p. 1; Consoli *et al.* 1993, p. 5). This statement is vague because the association – that is, the connection in the mind (Anon. 1995) – between environmental burdens and the product investigated is subjective. If your association is different from mine, we are likely to find different methodological choices relevant. This can affect the choice of, for example, system boundaries, allocation methods, and/or data sources.

According to the ISO definition, an LCA should provide information about the flows and environmental impacts of a product system. This statement is vague because it is unclear whether the product system is the system investigated in the LCA or the life cycle of the product investigated in the LCA. The system investigated in an LCA may, through system expansion, include processes outside the product life cycle. Part of the product life cycle may also be excluded from the system investigated if this does not significantly affect the overall conclusions of the study (ISO 1998a, p. 5). The ISO statement is also vague because it is unclear how it should be decided which flows and environmental impacts belong to which system or which activity. The latter is also a problem with the adjusted ISO definition presented in Section 1.1.

As indicated in Section 1.2, this thesis is based on the perception that good LCA practice is an application of LCA that can be expected to result in environmentally sound individual decisions and actions. Paper **II** states that in a well-defined problem, the decision-maker knows the outcome, or the outcome probabilities, of the available alternatives (Abelson & Levi 1985). Textbooks on decision theory (*e.g.*, Grubbström 1977) and corporate finance (*e.g.*, Brealey and Myers 1984) are based on the recognition that information on the consequences of available alternatives is necessary to make a rational decision. Baumann (1998, pp. 14-16) emphasises the fact that decisions are not always rational, well-informed choices between well-defined alternative actions. Our ability to make sound decisions is constrained by lack of knowledge, limited human capacity for information processing, and conflicting goals. To increase our ability to make environmentally sound decisions, an LCA should contribute to reducing these constraints.

The constraint of conflicting goals is apparent in the weighting phase of an LCA (where different environmental impacts are weighted against each other) and in the final decision-making (where environmental goals may be in conflict with economic or other goals). Conflicting goals may also occur in other parts of an LCA, such as when it is decided how to deal with environmental issues that cannot be quantified (Bras-Klapwijk 1998, p. 185). The problem of conflicting goals is addressed by authors such as Bras-Klapwijk (1998), Hofstetter (1998), and Tukker (1998), but this subject is beyond the scope of this thesis.

To reduce the constraint of incomplete knowledge, an LCA should provide information about the environmental consequences of possible actions. To reduce the constraint of limited capacity for information processing, the information provided through the LCA should be clearly presented. An LCA can probably never provide complete information about all the environmental consequences of all decisions that can be based on, or inspired by, the LCA results; however, the more information the LCA generates, and the clearer it is presented, the more the two constraints are reduced. It can be concluded, from the perception of good LCI practice, that a general requirement on the LCI methodology is that it should result in the most comprehensive and clear information possible about the environmental consequences of decisions that can be based on, or inspired by, the LCA results.

This view of what knowledge should be provided by an LCA is consistent with a pragmatic or instrumentalistic view of knowledge. Instrumentalism is a philosophical theory that was developed by the American philosopher John Dewey from the pragmatism of William James (Russel 1948, pp. 810-828; Aspelin 1963, p. 120). Both philosophers emphasise the close relationship between the thinking process and the practicalities of life. The purpose of the thinking process is for us to obtain the knowledge we need to act. A statement is true when it gives us adequate guidance for our actions (Aspelin 1963, p. 121). According to Dewey, knowledge is precisely the same as the ability to anticipate the consequences of manipulating things in the world (Dancy & Sosa 1992, p. 97).

Instrumentalism incorporates many themes from Kant and Hegel (Dancy & Sosa 1992, p. 97). According to Russel (1948, pp. 820-821), the most important philosophical contribution of Dewey is his criticism of the traditional concept of truth. According to instrumentalism, no truth is static and final; instead, it evolves over time. Instrumentalism apparently also includes elements of idealism, stating that inquiry produces the *objects* of knowledge instead of the *knowledge* of objects (Dancy & Sosa

1992, p. 97). On this point, instrumentalism has been criticised by realists who state that objects exist independently of our thinking about them (Ibid., p. 420).

It should be noted that, while my view of the knowledge generated in an LCA is consistent with instrumentalism, instrumentalism is not a prerequisite for the general requirements of comprehensiveness and clarity. The ontological question of whether objects are real or products of our thoughts is not relevant for the general requirements. The instrumentalistic definition of knowledge is not necessary either. It is sufficient to state that the knowledge that an LCA should provide is knowledge about the environmental consequences of manipulating things in the world. Only then can the use of LCA methodology be expected to result in environmentally sound individual decisions and actions.

Implications for system boundaries

To provide information about the environmental consequences of possible actions, we require data reflecting the environmental consequences of these actions. Some of the most important effects of an action can occur outside the life cycle of the product investigated (*e.g.*, Ekvall 1992). This can be the case if the system investigated includes (paper **V**):

- Multi-function processes (activities with multiple functions).
- Open-loop recycling (recycling of material or energy from one product life cycle into another).
- Use of limited resources (resources for which there is competition).

One way to take the indirect effects into account in an LCA is to expand the system investigated to include activities outside the product life cycle that are significantly affected by an action that may result from the LCA (Section 3.1). In some cases of open-loop recycling, it is possible to take indirect effects into account through the application of an allocation method that gives approximately the same results that system expansion does (paper **VIII**).

As an example of the use of limited resources, consider the decision to buy furniture produced from particle board (see Figure 6). This decision may have a marginal effect on the demand for particle board and on the demand for sawdust for particle board production, but it is not reasonable to expect that this will affect the production of sawdust since sawdust is a by-product with low economic value from sawmills. The quantity of sawdust produced depends on the demand for construction wood and similar products. This means that sawmill and forestry operations are likely to be unaffected by decisions to buy furniture produced from particle board; instead, less sawdust will probably be available for other purposes, such as the production of heat and electricity. In this context, sawdust is a limited, man-made resource (*cf.* Section 9.2 in paper **V**). Although the decision to buy the table will probably not affect sawmill and forestry operations, it may affect the demand for alternative energy sources (Figure 6). This indicates that forestry and sawmill operations can be excluded from an LCI that is carried through to describe the consequences of buying the furniture. On the other hand, the alternative use of the sawdust and the alternative method of generating the exported function should be included in the system investigated. Apparently, there is an important difference between the requirement that an LCA provide comprehensive information about the consequences of possible actions and the old SETAC requirement (Consoli *et al*. 1992, p. 7) that an LCA cover the entire life cycle of the product investigated.

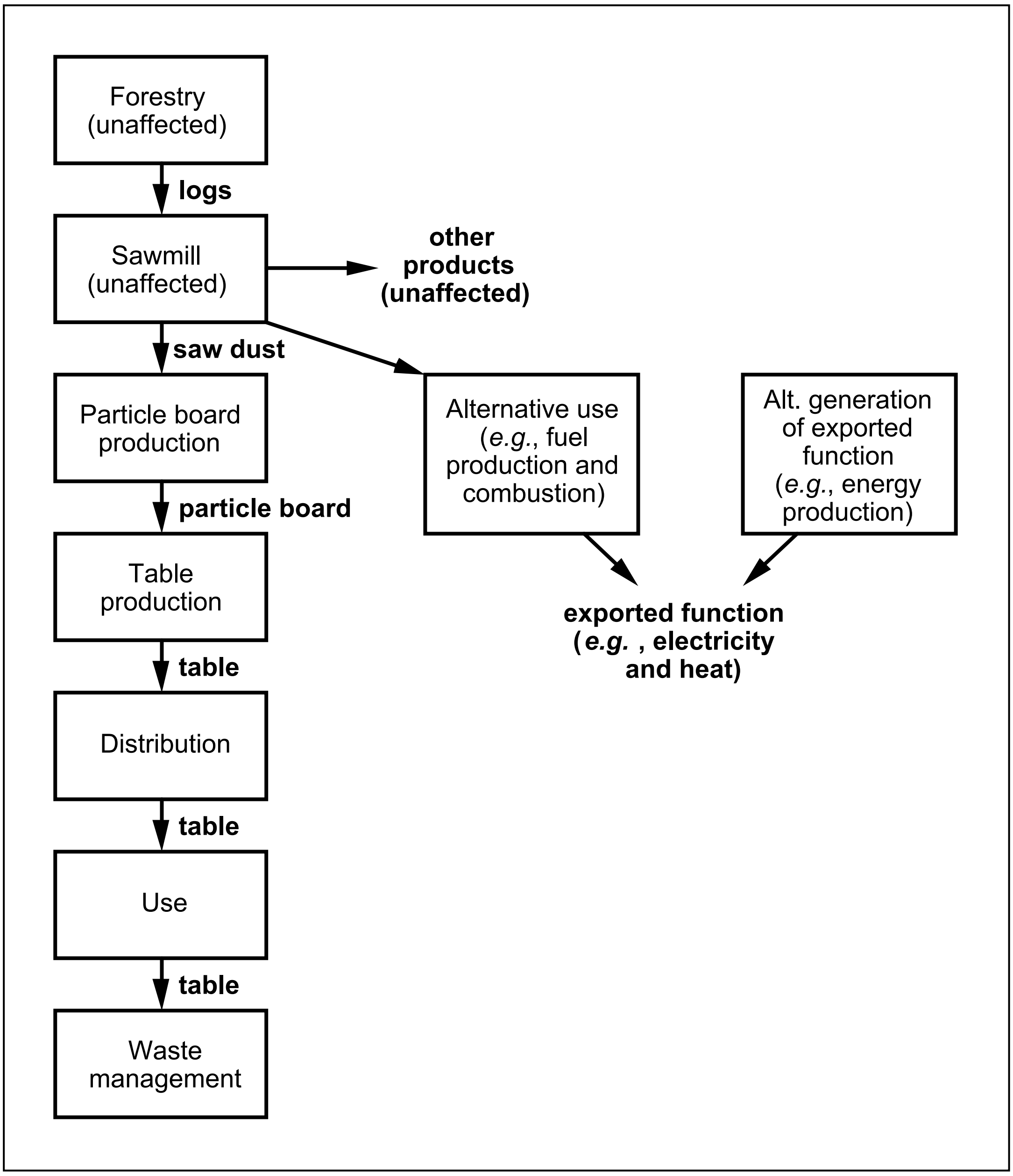

Figure 6. Simplified illustration of the life cycle of a hypothetical table. The system illustrated has been expanded to account for the alternative use of sawdust.

Implications for the choice of data

Azapagic & Clift (1999) distinguish between marginal and discrete effects on an activity. Marginal effects are infinitesimal effects on production volume. Discrete effects are substantial changes in production volume or complete changes from one activity to another (see Figure 7). Marginal effects should, ideally, be modelled using marginal data, *i.e.*, data reflecting the environmental burdens of the technology affected by a marginal change (Heijungs *et al.* 1992b, Baumann *et al.* 1993, Weidema 1993b, Frischknecht 1997, Ekvall *et al.* 1998, Frischknecht 1998, Weidema *et al.* 1999, paper **V**). Substantial effects should be modelled using incremental data (Azapagic & Clift 1999). These are likely to depend on the scale of change (Figure 7). Complete changes should be modelled using average data. Many actions can be expected to cause changes that are small enough to be approximated as marginal effects on the production of bulk materials (*e.g.*, steel, aluminium, polyethylene), energy carriers (*e.g.*, electricity, heavy fuel oil, petrol), and services (*e.g.*, waste management), for which total production volume is very high. As indicated above, decisions to buy furniture produced from particle board may have marginal effects on the production of particle board.

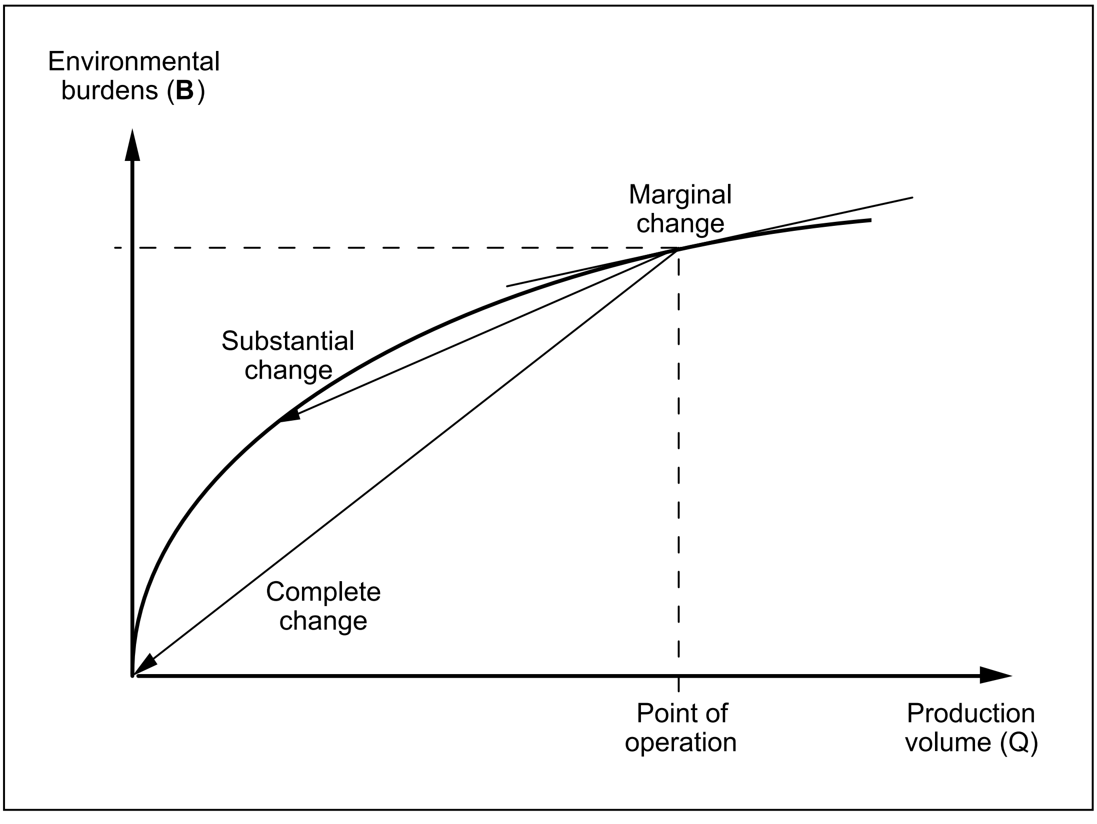

Figure 7. Marginal, substantial, and complete changes in a production process (Azapagic & Clift 1999).

Discussion

Effect-oriented LCI methodology, *i.e.*, an LCI methodology aiming at describing consequences of possible actions, should take indirect and marginal effects into account. Accurate modelling of indirect effects requires that the affected activities outside the product life cycle are identified. Accurate modelling of marginal changes requires that the marginal technologies are identified. The uncertainties can be very large in the modelling of the indirect effects (paper **I**, paper **V**, paper **VIII**) and the environmental burdens of the marginal technology (Ekvall *et al.* 1998, paper **V**). This reflects the fact that the actual consequences of an action are often uncertain. Using the terminology of Funtowicz and Ravetz (1991), this uncertainty can be classified as a technical uncertainty. Hence, if the general information requirements concerning comprehensiveness and clarity of the information are accepted, the methodological uncertainty is reduced but the technical uncertainty may still be quite large. If the uncertainties in the consequences are large enough to be significant for a decision, multiple scenarios or sensitivity analysis can be used to demonstrate the uncertainty.

5.2 Radically effect-oriented LCI

Referring to Figure 6, it is not certain that the production of particle board is affected by decisions to buy the table. Even the production of the table may be unaffected. Instead, the effect of decisions to buy the table might be that less particle board or fewer tables of this type are available for other purchasers. The indirect effects of buying the table may be that particle board is replaced by other materials in other products and/or that the production and distribution of other types of tables are increased (Figure 8). This indicates that economical analyses of the table market, the table producer, the markets for particle board and competing materials, the producer of the particle board, and so on are required to accurately estimate the environmental consequences of buying the table. And this conclusion can easily be generalised to other products. With a radical application of the requirement to generate information about the effects of possible actions, it is probably not sufficient, and perhaps not even relevant, to trace the materials in the product investigated back to the cradle – that is, to the extraction or generation of natural resources. The decision to buy the product does not necessarily mean that the amount extracted of these natural resources is increased.

The economic analyses require a different type of data and knowledge than is usually used in an LCI. This is in itself not a sufficient reason to refrain from making this type of radically effect-oriented environmental assessments; instead, it is an argument for involving economists in the case studies. An accurate environmental assessment of a decision apparently requires co-operation between economists and engineers. This has been pointed out before by, for example, Weidema (1993b).

In general terms, the consequences of an action propagate through the economic and technological systems in a chain of cause-and-effect relationships, resembling somewhat the ripples caused by a stone thrown in a lake. To provide a decision-maker (a policy maker, a product designer, a consumer, etc.) with comprehensive information about the environmental impacts of a specific action, the study should focus on the activities for which the environmental impacts are most affected by the action, regardless of whether they are located within or outside the life cycle of the product investigated. The propagation of the consequences should, ideally, be mapped as far down the cause-and-effect chains as the effects are large enough to be significant for the assessment of the action. The natural starting point of a radically effect-oriented environmental assessment of a specific action is then the point where the action is carried out – that is, the point where the stone hits the water.

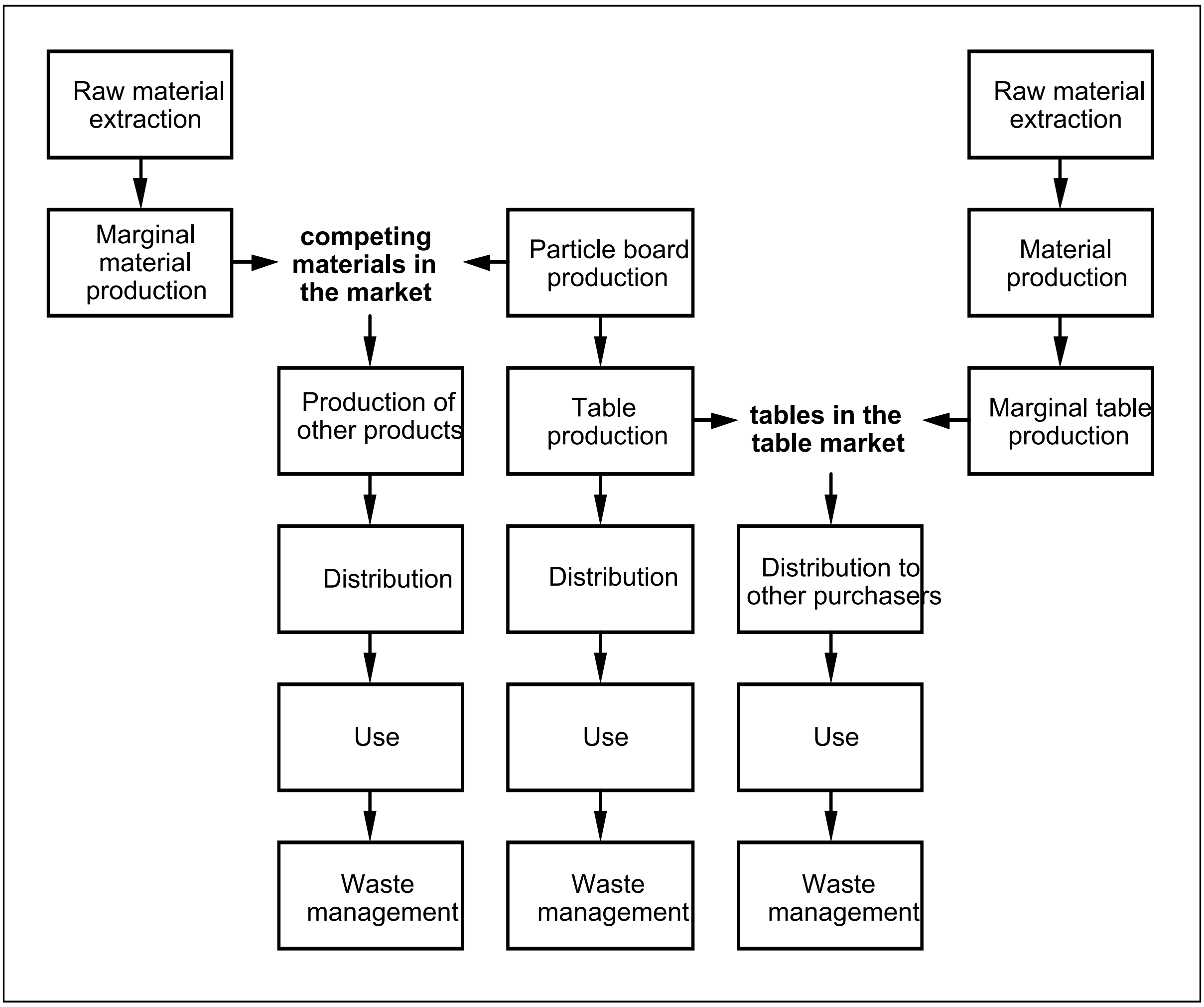

Figure 8. Simplified illustration of the system investigated in a hypothetical, radically effect-oriented assessment of the decision to buy the table in Figure 6.

If the LCI or LCA is carried through to generate knowledge and ideas for future decisions, the natural starting point depends on the intended audience: the activities where the audience can take actions. These activities are denoted as the foreground system by, for example, Tillman (1998). According to the general information requirement, the propagation of the consequences of possible actions should, ideally, be mapped as far as the effects are large enough to be significant for any decision that can be based on, or inspired by, the LCA results.

It is reasonable to expect that the uncertainties in the economic analysis will be large. As indicated in Section 5.1, this reflects the fact that the actual consequences of an action are often uncertain; however, it can be assumed that the large uncertainties make it impossible or pointless to estimate the consequences far down the cause-and-effect chains. This indicates that the boundaries of the system investigated should be defined at the point where the consequences are so small, or the uncertainties so large, that further expansion of the boundaries will yield no information that is significant for any realistic decision. In practice, it can be expected that the precision in the results as well as the system boundaries will depend on the amount of resources available for the assessment (*cf.* Section 5.4).

If the original focus of LCA on the life cycle itself is wrong, why not abandon LCA and develop a new methodology with a more relevant starting point? The system investigated in a radically effect-oriented environmental assessment (*e.g.*, Figure 8) will probably not bear much resemblance to a product life cycle. But there are strong, pragmatic reasons for remaining within the domain of LCA. The concept of LCA is well established among decision-makers, and LCA has several strong points, such as a structured procedure, an international standard under development, and established fora for methodological development and harmonisation. Hence, it is probably easier to contribute to environmentally sound decisions by adjusting the existing LCA methodology than by establishing a new, competing methodology.

5.3 General requirement II: acceptability

The decision-maker may be interested in information that is not clearly related to the consequences of possible action. Referring to Figure 6, as a consumer, I may be interested in knowing whether the wood particles in the furniture originates from a sustainable forestry, even if I know that my decision to buy the table will not change the quantity of wood harvested. In general terms, the way we perceive the environmental qualities of a product may be affected by the origin of the raw materials regardless of any causal relationships. Why is this? A possible explanation is that we do not want to be part of a system that we believe is bad, even when this system is unaffected if we leave it. Or we may consider the product to be responsible for the production of its raw materials, even when the quantity of raw materials produced is unaffected by the production of the product. The reason may be that we do not evaluate actions based only on the actual or expected consequences of individual actions but also on the basis of whether these actions reflect good rules (Lübcke 1988).

As stated in Section 1.2, the methodological recommendations in this overarching document are based on the tradition of evaluating individual actions; however, to contribute to environmentally sound decisions and actions, LCI practice should take into account the possibility that decision-makers have a different view of how actions should be evaluated. Otherwise, the knowledge generated through an LCA may not be used in the decision process. As indicated in paper **II**, the methodology used in the study must be acceptable to the decision-makers. Acceptability can be achieved by explaining the motives for the approach. One motive for accepting a methodological approach can be that it results in information about the consequences of possible actions; however, as indicated above this may not be perceived as a sufficient justification.

It is not possible, at this stage, to draw any distinct conclusions regarding which methodological choices can be made acceptable to decision-makers. The way we perceive the environmental qualities of a product depends on subjective associations (*cf.* the old SETAC definitions of LCA, see Section 5.1). The perception that the product is responsible for the environmental burdens of a specific activity is subjective (paper **II**), and the perception that an activity or action is part of a specific system is also subjective. As suggested by Gaines (1979), a system is what is distinguished as a system. A thing, or a set of things, is not a system in itself. It becomes a system only when the observer applies a systemic view to it. Presumably, the observer applies a systemic view because, with respect to his or her intent, the observer feels that it is appropriate to regard the thing, or set of things, as a system (Klir 1991).

All this indicates that the acceptability of methodological choices is based on purely subjective considerations and that little methodological guidance can be derived from the acceptability requirement; however, the strong interest in LCA for the past decade (see Section 1.1) indicates that the product life cycle is perceived as a relevant concept. This view is, to some extent, supported by Goguen & Varela (1979), who present four approaches to identifying which system is ”more coherent, more natural, more whole than others”. One of the approaches is to investigate whether the system has interesting ”emergent” properties. The product life cycle has an emergent property: it fulfils the function of the product. If the product life cycle is perceived as a relevant concept – even outside the sphere of LCA practitioners - the radically effect-oriented methodology, in which the life cycle concept is more or less forsaken, may be unacceptable to many decision-makers. If decision-makers perceive the product life cycle as a natural, coherent system, the use of the life cycle concept may also make the learning process more efficient and, hence, increase the amount of knowledge that is generated through the LCA.

5.4 General requirement III: feasibility

An action may have indirect effects that propagate through the whole global technological system. This system is too complex to analyse on the level of detail used in an LCA. To contribute to environmentally sound decisions, LCI methodology must be feasible. Hence, it is necessary to simplify the system (Ekvall *et al.* 1992). The challenge is to simplify without losing too much relevant information.

A detailed LCA is beyond the budget and/or time constraints of many potential users (Svensson & Ekvall 1995). To contribute to many decisions, the methodological approaches should be easy to apply. This means that it is an advantage if the amount of information needed for the allocation approach is small and if the necessary data are easy to collect and interpret (paper **II**).

In general terms, there seems to be a conflict between the requirement to provide comprehensive knowledge and the feasibility requirement (paper **II**). One way to simplify the system is to investigate only a small part of the global technological system. In order to provide the most comprehensive information possible about the consequences of possible actions - within the budget and/or time constraints given - the system investigated should include the parts of the technological system that are expected to be most affected by such actions.

5.5 Application-specific requirements

An LCA can be carried through for many different purposes. Different applications are listed by different sources. According to the international standard (ISO 1997), LCA can assist in the following:

- Identifying opportunities to improve the environmental aspects of products.
- decision-making in industrial companies or other organisations, such as strategic Planning, setting priorities, or product or process design or redesign.
- Selecting relevant indicators of environmental performance.
- Marketing, such as environmental claims, ecolabelling schemes, or environmental product declarations.

Application typologies

As stated in Section 1.1, various attempts have been made to structure the various applications of LCA and to describe the connection between the study goal and the methodological choices that should be made in the LCA. Several of these application typologies are summarised by Weidema (1998). The overview presented in this section focus on aspects of some of the application typologies that affect the applicability of an effect-oriented LCI methodology that takes indirect and marginal effects of possible actions into account.

Many application typologies include approximately the same distinction between descriptive and change-oriented LCA (*e.g.*, Andersson *et al.* 1999). Different words are used to denote the two types of LCA (Table 1), and there are slight differences between the distinctions presented in different publications. An awareness-raising or exploratory LCA is distinguished by its purpose: by the fact that the LCA is not made to assess a specific action but for learning purposes (Baumann 1996; Cowell 1998, p. 202). The desired outcome is increased understanding of the complex system investigated and its environmental burdens. The implicit referential situation described by Heintz & Baisnée (1992) and the information-oriented LCA described by Weidema (1993a & 1998) also appear to belong to this category.

Table 1. The distinction between descriptive and change-oriented LCA in various application typologies.

Publication	Descriptive LCA	Change-oriented LCA
Heintz & Baisnée (1992)	Implicit referential situation	Explicit basis for comparison
Weidema (1993a & 1998)	Information-oriented LCA	Change-oriented LCA
Baumann (1996)	Exploratory LCA	Comparative LCA
Frischknecht (1997 & 1998)	Type 0 LCA	Type 1-3 LCA
Baumann (1998)	Life cycle accounting	Life cycle assessment
Cowell (1998)	Awareness-raising purpose	Decision-making purpose
Hofstetter (1998)	Attribution case	Change-oriented case
Tillman (1998 & 1999)	Retrospective LCA	Prospective LCA

Life cycle accounting, or retrospective LCA, is an assessment of the environmental burdens for which a product is made accountable. It is distinguished by the way the definition of system boundaries and other methodological choices are made: the methodological choices should be based on established conventions and, typically, excludes system expansion (Baumann 1998, pp. 23-24; Tillman 1998 & 1999). This is distinguished from an prospective LCA carried through to assess the consequence of a change, where the system boundaries should be defined so as to include the activities that contribute to the environmental consequences of the change. System expansion is a characteristic approach in this case. A similar distinction appears to be made by Hofstetter (1998, p. 11).

Frischknecht (1997, pp. 64-65; 1998, pp. 77-78) distinguishes between four categories of applications:

- Type 0 - status quo: describing current performance.
- Type 1 - short run: short-term system optimisation.
- Type 2 - long run: product comparison, optimisation, etc. in the long-term perspective.
- Type 3 - very long run: very long-term strategic planning.

According to Hofstetter, attribution applications include hot-spot identification, product information, and environmental labelling; however, hot-spot identification and elimination is a type 2 (change-oriented) application in the Frischknecht typology.

Applicability of effect-oriented methodology

In an early paper (paper **I**), we argued that system expansion is useful for investigating the consequences of a change. One of the drawbacks of system expansion, we argued, was that the results from different LCAs would not be additive. For this reason, we considered approaches other than system expansion to be more useful for the development of databases and accounting systems designed to describe which environmental loadings a product can be made responsible for. More recent developments have shown that additivity is not necessary for the development of databases; instead, the database format can be flexible and comprehensive enough to separately store and adequately describe incompatible sets of data (Tillman 1998).

Frischknecht (1997, pp. 76-77) argues that system expansion is not applicable for descriptive LCA. The reason is that no activity is displaced by co-products from the life cycle investigated because the production volume is constant. In his dissertation, however, Frischknecht (1998, pp. 104-105) argues that, although no activity is replaced in the reality modelled in the descriptive LCA, ”system expansion may well be used to show what *would* happen *if* a joint production process were put out of operation”. For this reason, he argues that the appropriate approach to allocation does not depend on the goal of the LCA.

As stated above, Baumann (1998) and Tillman (1998 & 1999) indicate that system expansion is characteristic for prospective LCAs, while partitioning is a characteristic part of retrospective LCAs. Baumann (1998) makes the distinction between life cycle accounting and life cycle assessment using the parallel of the distinction made in economics between accounting and investment analysis; however, there are important differences between life cycle accounting and economic accounting, both in application area and in methodology. One important application area for economic accounting is to provide a basis for calculating the amount of tax a company should pay. This is clearly not an application area for LCI. However, there are some similarities between this application and environmental product declarations, for which the accounting type of LCI has been recommended (*e.g.*, Tillman 1998 & 1999). Large economic interests can be at stake in both cases, which means that a strictly formalised method is called for. Furthermore, to be efficient, both the tax system and schemes for environmental product declarations should be based on calculations that are considered to be fair.

A description of the consequences of buying a product may, in some cases, seem unfair; however, unfairness is not an inherent property of effect-oriented LCI methodology. On the contrary, it can be argued that it is fair for the environmental product declaration to be based on an assessment of the environmental impacts that result from the decision to buy the product.

Tillman (1999) states that it may be impossible to establish consensus on a sufficient level of detail for an effect-oriented LCI methodology to be used for environmental product declarations. This is probably true, at least in the near future. A possible reason is that effect-oriented LCI methodology is relatively new and immature. The informative, awareness-raising, accounting type of LCI methodology has a longer tradition (*e.g.*, Cowell 1998, pp. 202-203). I have found no reason to assume that it is impossible to develop a theoretical foundation for at least a moderately effect-oriented methodology for which consensus and a detailed standard can be established.

Another important application area for economic accounting is to make it possible for decision-makers outside and within the company to monitor the economic performance of the company (Baumann 1998). Internal and external decision-makers may have an interest in monitoring the environmental performance as well. In this application area, the difference in methodology between economic accounting and accounting-type LCI may be important. Economic accounting deals with one monetary parameter only and is limited to the boundary of the production site or the company. Accounting type LCI deals with a relatively large number of environmental parameters and includes the whole life cycle of a product. Efficient monitoring requires that information be frequently updated. An LCI of any type can be too cumbersome to use in a continuous monitoring system, particularly in small companies and in companies with many and/or complex products. Less complex environmental performance indicators (ISO 1998c) are probably often more efficient. Feasibility and acceptability are important requirements in this application, but I have found no reason to assume that a moderately effect-oriented LCI methodology, at least, cannot be used for monitoring purposes or for identifying relevant environmental performance indicators.

A third important application area for economic accounting is to establish whether company board members and others have been responsible for any misconduct. As a parallel, an accounting type of LCI can be carried through to quantify the environmental burdens for which the product should be held responsible. An important difference is that, while the responsibilities of board members are regulated through legislation, the environmental responsibilities of the product are based on subjective perceptions as long as important methodological problems in the LCI remain unsolved. Although acceptability and feasibility are important in this LCI application, I have found no reason to assume that at least a moderately effect-oriented LCI methodology cannot be used to establish the environmental responsibility of the product.

Tillman (1999) states that additivity is important to achieve a system for environmental product declarations, in which each producer adds the environmental burdens of his/her process to the burdens associated with the raw materials and intermediate products that are used in the process. Such a system is not a requirement for environmental product declarations, but it may be a practical solution. Lack of additivity can be a problem in some effect-oriented LCAs, but in many cases it will not be a problem. Results based on marginal data are often additive, because the sum of two marginal changes is often still a marginal change. In the case of open-loop recycling, additive results can be obtained through whole-system modelling with linear programming (Azapagic & Clift 1999) or through system expansion with a market-based approach (Section 4.5). System expansion at multi-function processes often yields results that are not additive, but this makes the LCI self-inconsistent only when different parts of the system investigated utilise different functions of the multi-function process. Systems for environmental product declarations require supervision and critical reviews of the LCAs (ISO 1997). Part of the critical review process could be to check for inconsistencies caused by the lack of additivity.

It has been argued that it is important that the information provided by an environmental product declaration takes into account the environmental burdens of the whole life cycle (*e.g.*, Tillman 1999); however, the choice to buy a certain product does not necessarily affect the whole life cycle of the product (see Figure 6). If the LCA should provide as comprehensive and clear information as possible about the consequences of buying the product, only the activities that are affected by the decision should be included in the LCA.

As stated by Tillman (1999), it is difficult to estimate the scale of change that will be caused by an environmental product declaration. This is also true for awareness-raising or exploratory LCAs that are carried through for learning purposes, without a specific action in mind. As shown in Figure 6, however, production of particle board and waste management are likely to be marginally affected, regardless of the effect that an environmental product declaration has on the table market. Historically, a large share of the energy demanded by the industry has been used for the production of bulk materials, such as primary metals, pulp and paper, and bulk chemicals (Goldemberg *et al.* 1988, pp. 92 & 100), which are produced in very large quantities. The production volume of these materials is probably marginally affected by any environmental product declaration. They are also marginally affected by most actions that are inspired by awareness-raising or exploratory LCAs. Hence, for the production phase, it is reasonable to assume that most of the effects of environmental product declarations on the energy-related (at least) environmental burdens can be modelled using marginal data. The same is true for the waste management phase. The effects on the use phase of buying the product can be modelled with average data, at least if the radically effect-oriented methodology is not applied (*cf.* Section 5.2). Hence, it is reasonable to assume that most of the energy-related effects of environmental product declarations can be modelled regardless of the impact of the environmental product declaration on the market.

The unknown scale of change can cause problems in modelling activities that may be substantially affected by the environmental product declaration. Referring to Figure 6, the quantities of the table that are produced, distributed and used may be either marginally or substantially affected by an environmental product declaration. If an effect-oriented LCI methodology is applied, it is not certain whether marginal or incremental data should be used to model these parts of the system; hence, the technical uncertainty will be relatively large in those parts of the system investigated.

It has also been argued that it is impossible to know which product a purchaser would choose if he/she does not buy the product investigated (*e.g.*, Tillman 1999); however, as indicated above, the decision made by the purchaser can be defined as the choice between buying or not buying the product investigated. This implies that the LCA should be an effect-oriented comparison between two relatively well-defined options: the product and a baseline (a ”zero alternative”, *cf.* paper **I**). It is, of course, important that the environmental product declarations of competing products be based on the same baseline. This baseline could be defined when the details in the formalised method are decided upon for each specific category of products.

Awareness-raising or exploratory LCAs can be carried through before any alternative actions have been formulated. It is not possible to predict, when the LCA is carried through, which decisions will be based on, or inspired by, the LCA results. Some of the possible decisions may have no effect on certain activities, on which other decisions may have marginal or even substantial effects; hence, it is not possible to accurately model the consequences of all possible effects at the same time. This, in itself, is not an argument against system expansion that aims at including activities that can be affected by the decision-makers or against excluding from the analysis activities that the decision-makers cannot influence. Nor is it an argument against using marginal data to model activities that can be only marginally affected by the decision-makers. In other words, effect-oriented LCI methodology, with the aim of describing consequences of possible actions, can be applied in awareness-raising or exploratory LCAs, but the technical uncertainty will be relatively large.

5.6 Conclusions

As indicated in the previous section, there are differences between the descriptive/change-oriented distinctions made in different application typologies. An exploratory, awareness-raising LCA that is carried through to obtain increased understanding of the complex system investigated does not require a methodology based on established conventions. On the other hand, results from an LCA that is based on established conventions can be used to assess specific actions, such as the action of buying the product investigated.

Acceptability is important for making the LCI methodology an effective tool in environmentally sound decision-making (Section 5.3). The acceptability of the method depends on what information the decision-maker perceive as relevant for the decision-making process. Based on Sections 5.1 and 5.3, I propose that the relevant information can be described either in terms of the effects of possible actions on the total environmental burdens or in terms of the environmental identity or responsibility of the product.

Figure 9 illustrates a two-dimensional typology of LCA applications. One dimension describes the phase in the decision-making process (Baumann 1996). The other dimension describes what information is perceived as relevant for the decision. As indicated in Figure 9, all four types of LCA are possible. Effects on the total environmental burdens can be described either to assess specific actions or to generate ideas for future, environmentally sound individual decisions. Effect-oriented methodology is probably best suited for these applications since, per definition, it aims at describing the consequences of possible actions.

The environmental identity or responsibility of a product can be established with the aim of generating ideas for future decisions. They can also be established to assess specific actions for improving the perceived environmental identity or for reducing the environmental burdens for which a product is held responsible. The methodological choices are based on subjective perceptions of the identity or responsibility of the product, unless the choices are based on established conventions.

Phase in the decision-making process

Information perceived as relevant	**Exploratory** *e.g.*, key issue identification	**Comparative** *e.g.*, improvement, purchasing	Applicable methodology
Identity / responsibility			Subjective or based on convention
Effects on environmental burdens			Effect-oriented

Figure 9. Typology of LCA applications.

Effect-oriented LCI methodology can apparently be used in most LCA applications. An important exception, at least in the near future, occurs when detailed standardisation of the methodology is required, such as for environmental product declarations. The reason is that it is difficult to establish consensus on a detailed effect-oriented LCI methodology. The advantage of an effect-oriented LCI methodology is, of course, that it is likely to result in more comprehensive and accurate information about the consequences of buying a product. My conclusion is that it should be considered an important task for LCA researchers to try to develop a theoretical foundation for a detailed effect-oriented methodology for which consensus and a standard can be established. This thesis is part of that research, focusing on the modelling of indirect effects. Modelling of marginal effects is investigated by, *e.g.*, Azapagic (1996), Weidema *et al.* (1999), and Azapagic & Clift (1999). If these attempts succeed, an effect-oriented methodology may eventually be standardised on a sufficient level of detail to be used for environmental product declarations.

A more inherently permanent distinction is the one between a comparative, decision-making LCA, in which well-defined, alternative actions are compared, and an exploratory, awareness-raising LCA that is carried through to generate ideas for future comparisons and/or decisions. When the alternative actions are well defined, the type and scale of the change is known. In an exploratory LCA, the type and scale of the future change is unknown. An effect-oriented LCI methodology can be used for both purposes, but the technical uncertainties will be much larger in the second case. Two recommendations can be provided at this stage for an effect-oriented LCI methodology in an exploratory LCA. It is clear from these recommendations that the appropriate system boundaries and data sources depend on the intended audience for the LCA:

- System boundaries: the system investigated should reflect the sphere of influence of the decision-makers to which the LCA results are reported. If the LCA practitioner should strive at providing as comprehensive and clear information as possible about the consequences of possible actions, there is no reason for the system investigated to include activities that cannot be affected by the decision-makers. If the LCA results are reported to the producer of the table in Figure 6, the forestry activities can probably be excluded from the system investigated, but it may be important to include the alternative energy production. Admittedly, the sphere of influence can be fuzzy since the boundary between activities that can be affected by the decision-makers and activities that cannot be affected is not always distinct.
- Choice of data: functions (*i.e.*, the production of products and processing of waste) that are generated in very large quantities are marginally affected by most actions; hence, the production of these functions can be modelled using marginal data. This rule should be used with care if the LCA results are reported to decision-makers that are responsible for strategic planning of large-scale systems, since large-scale strategic decisions may have discrete effects, even on products that are produced in very large quantities.

6. Validation of the market-based approach

This chapter includes the validation of the market-based approach to the allocation problem caused by open-loop recycling (see Section 4.5). This validation also has implications for effect-oriented LCI methodology in general. It includes a theoretical discussion of the criteria completeness, clarity, acceptability, feasibility, applicability, and precision.

6.1 Completeness

Section 5.1 indicates that an LCA should provide the most comprehensive knowledge possible about the consequences of possible actions. This means that modelling of the technological system should take important and relevant causal relationships into account. Open-loop recycling can have important indirect effects (Section 5.1). These indirect effects depend on how the market for the recovered material reacts to a change in the supply of, or demand for, the recovered material (paper **VI**). The major advantage of the market-based approach – compared to other approaches to this allocation problem – is that the mechanisms that govern the markets for recovered material are taken into account. To my knowledge, these are not taken into account in any other approach to open-loop recycling.

Degradation of material is not taken into account in the market-based approach. If the purpose of the LCA is to enhance our ability to anticipate the environmental consequences of our actions, quality reduction is relevant only to the extent that it affects the environmental consequences of recycling. This effect is not necessarily proportionate to the quality reduction (see Section 4.3). To my knowledge, the effects of quality reduction are not adequately taken into account in any approach to open-loop recycling.

The market-based approach – as well as other approaches – takes into account the fact that one action may affect the actions of other decision-makers. Actions that result in increased collection of old corrugated containers (OCC) in one city may, for instance, result in reduced collection at other locations. However, neither the market-based approach nor other approaches take into account the fact that an action may also affect the values of other people. For example, an increased collection in one city is assumed not to affect the perception at other locations concerning at what OCC price the OCC collection should be pursued. This ceteris paribus assumption concerning the values of others is common practice in LCA, but it may be a significant source of error, since our apparent actions and the apparent results of our actions may well affect the values of other people (*e.g.*, Keyes 1982). An increase in the collection of old corrugated containers in one city may, for example, inspire decision-makers at other locations to increase collection efforts.

The limited validity of this ceteris paribus assumption affects the applicability of marginal data in general. An isolated action may have marginal effects on a market. As indicated in Section 5.1, marginal data should be used to model these effects. But if this action inspires other people to take similar actions, the scale of the change is larger and the effects on the market might be incremental rather than marginal. If these effects can be modelled, marginal data might not be applicable for the modelling.

In fact, this ceteris paribus assumption affects the validity of the whole effect-oriented methodology. A possible explanation for why an effect-oriented methodology may be difficult to accept is that we do not want to be part of a system that we believe is bad, even when this system appears to be unaffected when we leave it (Section 5.3). The reason for this may be precisely that we hope that our action will affect the values of other people, and that, in the long run, our action will contribute to a significant effect.

The market-based approach is also based on the assumption that the demand for other products and functions is unaffected by a change in the flows of recycled material across the boundaries of the product life cycle investigated. In the terminology introduced in an earlier paper (Ekvall *et al.* 1992), the market-based approach results in a technological whole system and not in a socio-economic whole system. The ceteris paribus assumption concerning the quantity of other products produced is also common practice in LCA (*e.g.*, Heijungs *et al*. 1992b); yet, the consequence is that certain economic and social causal relationships are ignored (paper **I**).

6.2 Clarity

To contribute to increased knowledge, it is not sufficient that an LCA provide comprehensive information. The information must also be intelligible and, preferably, easy to understand. This indicates that, for example, the concepts on which the methodology is based should be clearly defined and easy to understand. The market-based approach is based on the concept of price elasticity, which is well defined and established within the area of price theory (Friedman 1976). Whether it is easy to understand within the context of LCAs remains to be seen since the approach has not yet been applied in real case studies. However, experience from practice with the 50/50 method indicates that the mechanisms involved in that approach are sometimes difficult to understand for LCA practitioners. There is a risk that the difficulties are even larger with the market-based approach.

Some of the other methodological approaches to open-loop recycling are based on the concept of material quality. On a general level, the concept of quality appears to be intuitively clear, but it defies attempts to define it (Pirzig 1978). Within the context of LCA, material quality can also be difficult to define and measure because it is a complex concept (paper **II**). For the same material, different quality aspects, such as elasticity, strength, or corrosion resistance, can be important for different products.

Various attempts have been made to define or measure material quality (*e.g.*, Karlsson 1995, Ryding *et al.* 1995, Wenzel 1998b), but no definition or measurement procedure has been agreed upon internationally. It is possible that the price of the material is an adequate approximation for quality in many cases (paper **II**).

6.3 Acceptability

The acceptability of the market-based approach has not yet been investigated since the approach is new. However, the 50/50 method was recommended for hot-spot identification by Lindfors *et al.* (1995a) because, with that approach, everything that might be considered to be a hot spot is included in the system investigated. This method has also been used in several case studies (*e.g.*, Björklund *et al.* 1996, Brännström-Norberg *et al.* 1996, Engberg & Eriksson 1998), although some LCA practitioners and LCA commissioners have found it difficult to accept. One point of critique has been that the 50/50 allocation appears to be arbitrary (Trinius & Borg 1998).

The market-based approach is a refined version of the 50/50 method. It should not appear to be arbitrary since it is based on economic theory. Hence, I expect that the market-based approach will be less difficult to accept than the 50/50 method is.

6.4 Feasibility

The theoretical examples in paper **VIII** indicate that the market-based approach is feasible. Just as with other versions of system expansion, it requires the collection of data for activities outside the life cycle of the product investigated. More specifically, the market-based approach requires data on alternative material production (V_O in Figure 5) and on alternative waste management (W_O). It also requires information about price elasticities and relevant legislation. Adequate estimates for the elasticities of supply and demand can be difficult to obtain; however, once the elasticities are calculated, they can be used in other case studies with similar time-frame etc (paper **VIII**).

Most other approaches to open-loop recycling are likely to demand data on primary material production, even if the product investigated is produced from 100% recycled material (paper **II**). They are also likely to demand data on final waste management, even if the product investigated is 100% recycled after use. To apply methods based on reduction in material quality, the products that receive recycled material from the product investigated must be known. These methods also demand data on the quality of the material in the product investigated and in the products receiving this material.

6.5 Applicability

The market-based approach can be used with system expansion. The conceptual model in Figure 5 can also be used to calculate a basis for allocation if the environmental burdens of activities within the product life cycle investigated are similar to the environmental burdens of corresponding activities in other life cycles (paper **VIII**). The market-based approach is an effect-oriented approach; hence, it can be used to assess the effects of possible actions on environmental burdens (Figure 9). It can also be used to establish the environmental identity or responsibility of the product investigated, unless the approach is unacceptable to the intended audience or prohibited by the established convention.

The market-based approach provides accurate information about the indirect effects of possible actions only when the actions have marginal effects on the total market for the recycled material (paper **VIII**). The concept of price elasticity can be used only when economic forces decide the recycling rate. When authorities decide the recycling rate, the concept of a market for recovered material, which connects the life cycle investigated to other life cycles, can still be used to estimate the indirect effects of recycling; however, in this case it may be necessary to adjust the conceptual model (paper **VIII**).

6.6 Precision

The uncertainty in the ratio between the price elasticities of demand and supply is likely to be large (paper **VIII**). This means that the precision, or technical uncertainty, is large. If the market-based approach is based on the default values presented in paper **VIII**, the technical uncertainty is only slightly less than when the market mechanisms are unknown.

7. Conclusions of the thesis: a summary

As stated in Section 1.2, my postgraduate research included four theoretical research tasks. The first task was to determine the consequences of different system boundaries and allocation methods for the LCA results. Lee *et al.* (1995) claim that the decision to exclude from the LCA services such as heating, lighting, air compression, etc. in a manufacturing plant can have a decisive effect on the LCA results. That conclusion appears to be the result of large calculation errors (paper **III**). However, the decision to include or exclude indirect effects, and the methodological approach to the allocation problems, can have decisive effects on the modelling of that part of the system (Boustead 1994, Azapagic 1996) and even on the results and conclusions of the LCA as a whole (Lindfors *et al.* 1995b, Östermark & Rydberg 1995, Rydberg 1995, Strömberg *et al.* 1997, Frank *et al.* 1998, Trinius 1998, paper **IV**, paper **V**, paper **VII**).

The first research task also included an investigation of the type of information conveyed to decision-makers by LCIs with different approaches to the allocation problems. System expansion makes it possible to generate information on the indirect effects of a decision (paper **I**); however, this requires that accurate data are used for the activities that are included in the expanded system. This has often not been the case (Sections 3.1 and 4.1 in overarching document [OD], paper **IV**, paper **V**, paper **VI**). Subdivision and/or allocation based on physical, causal relationships generate information about the consequences of a decision that affects the internally used functions but not the exported functions of a multi-function process (OD Section 3.1, paper **VI**). Allocation based on gross sales value generates information about the causes of environmental burdens (Huppes 1993, paper **II**).

The second research task was to establish how the goal definition may be used to guide the choice of system boundaries and allocation method. This issue is discussed in Chapter 5, paper **I**, paper **II**, paper **IV**, paper **V**, and paper **VI**. An effect-oriented LCI methodology, including system expansion and the use of marginal data, can be used in most LCA applications (OD Section 5.6). The discussion in Chapter 5 indicates that this is a good methodological choice if the aim is to contribute to individual decisions and actions that result in a lower level of environmental burdens per functional unit than would have been the case without the LCA. An important exception, at least in the near future, occurs when a detailed standardisation of the methodology is required, such as with environmental product declarations. With a radically effect-orientated methodology, an environmental assessment requires co-operation between economists and engineers. In this context, the life cycle concept becomes irrelevant (OD Section 5.2). However, the use of an effect-oriented methodology may be restricted by the requirement to provide information that the decision-makers perceive as relevant (OD Section 5.3). The resources available for the study also restrict the precision and comprehensiveness in the modelling of indirect and marginal effects (OD Section 5.4).

The third research task was to refine the 50/50 allocation method for open-loop recycling with the aim of accurately taking into account the effects of recycling. This task resulted in the market-based approach to open-loop recycling (OD Section 4.5, paper **VIII**). The market-based approach can be used for system expansion as well as for allocation. It allows for a more accurate modelling of the indirect effects of open-loop recycling than does any other approach I have encountered, but the precision in the model is poor unless it is based on precise information on the market mechanisms (OD Chapter 6, paper **VIII**). The 50/50 allocation method presented in an earlier paper (Ekvall 1994) can be regarded as an approximation of the market-based approach (paper **VIII**).

The fourth research task was to reduce the methodological uncertainty in the environmental comparison between recycling and waste incineration of wastepaper, old corrugated board, and similar materials. Important methodological issues include the modelling of indirect effects and the choice of data for electricity production (Ekvall 1996, paper **IV**, paper **V**). The indirect effects depend on which energy source competes with energy from wastepaper incineration, on which material is replaced by the recycled paper, and on the alternative use of the forest resources that are not required for pulpwood production. Incineration of wastepaper can often be replaced by incineration of other waste flows in the short-term perspective (paper **VI**, paper **VII**). In the long run, the competing energy source depends on other decisions, such as decisions to invest in waste incineration plants (paper **V**). Recycled paper from one product or from one geographical area can be expected to replace a mix of primary material and recycled material from other products or locations (paper **VI**). This mix depends on factors such as price elasticities and/or political decisions in the markets for recovered paper (paper **VIII**). In the short-term perspective, wood that is not required for pulp production will probably not be harvested (paper **VII**). In the long-term perspective, an alternative use of forest resources might be fuel production (paper **V**). Marginal changes in the demand for electricity in the Nordic countries will affect electricity production in existing, coal-fired power plants in the short-term perspective (Baumann *et al.* 1993). In the long-term perspective, these changes are likely to contribute to affecting decisions to invest in new power plants based on coal or natural gas, or the phase out of Swedish nuclear power (Ekvall *et al.* 1998). The conclusions of a comparison between recycling and incineration with energy recovery depend on the time-perspective and on decisions on other waste management, forestry, and energy policies. To obtain the environmentally optimal wood-fibre system, recycling should, ideally, be assessed in connection with these other decisions (Ekvall 1996, paper **IV**, paper **V**).

8. Glossary

Accurate: without mistakes; the fact that the correct entity is measured, estimated, or calculated.

Allocation: the partitioning of the environmental burdens of a technological activity between the products for which the activity is used.

Effect-oriented methodology: a methodology aiming at describing consequences of possible actions.

Environmental burdens: the resource demand, the emissions of pollutants, the waste generated, and the land transformation caused by technological activities.

Exported functions: functions that are generated in one product life cycle but utilised in another product life cycle.

Function: the production of a product or the processing of waste.

Indirect effects: effects of an action on the environmental burdens of activities outside the life cycle of the product investigated.

Life cycle assessment (LCA): the compilation and evaluation of the material and energy flows and of the potential environmental impacts of the life cycle of a product.

Marginal data: data reflecting the environmental burdens of the technology affected by a marginal change.

Multi-function process: an activity that fulfils more than one function.

Open-loop recycling: recycling of material or energy from one product life cycle into another.

Precision: exactness of, for example, a measurement, estimation, or calculation of an entity.

Product: service and/or physical product.

Product life cycle: the system consisting of models of the technological activities used for the various stages of the product: from extraction of raw materials for the product and for ancillary materials and equipment, through the production and use of the product, to the disposal of the product and of ancillary materials and equipment, if any.

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