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Eco-efficiency in extended supply chains - methodological development with regulatory and organizational implications

Thesis for the degree of doktor ingeniør

Trondheim, September 2006

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*One giant leap for a man, a small step for mankind
(Free interpretation)*

Abstract

The purpose of this thesis is to contribute to the development and understanding of eco-efficiency assessments for extended supply chains. Using a methodological approach, the main outcome is a consistent methodology to assess the eco-efficiency for extended supply chains. The methodology allows comparisons both between different extended supply chains, e.g. the life cycle of different products, and also within an extended supply chain to reveal which processes are the most important for the value performance and the environmental performance, and to determine the contributions of supply chain partners to this performance. The methodological development also includes a proposal for how land use impacts on biodiversity should be included in the environmental assessments.

The methodological approach is used to reveal regulatory and organizational implications for the extended supply chains. In a case study on furniture production the assessments of eco-efficiency are used in two ways: to provide recommendations about new regulations that could motivate improvements in eco-efficiency performance of the products; and, to suggest organizational changes that could and should be performed to realize the potential for improvement.

This thesis demonstrates that the methodological, regulatory and organizational aspects of eco-efficiency are closely interlinked and must be used in combination to realize the potential for improvement.

The methodological recommendations are believed to be valid for all extended supply chains, while the regulatory and organizational implications are case specific. The approach used to derive with the recommendations should be transferable to other extended supply chains.

This thesis shows how measures of eco-efficiency can be used in the search for a path to sustainability. However, eco-efficiency must not be misinterpreted as sustainability since eco-efficiency only deals with relative and not absolute values, and does not incorporate social issues.

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The thesis is written as a compilation of five research papers. Several people have contributed to raise the quality of these. Most of them are recognized in the acknowledgements in the papers, but I would in particular thank my co-authors on two of the papers for their contributions; Alexander Dahlsrud, Christian Solli, and Anders Hammer Strømman, in addition to Annik Magerholm Fet. Several anonymous reviewers have also made their contributions. In addition, Roland Clift and Sigurd Støren were appointed critics on a 'trial dissertation' and provided useful comments towards the finalizing of the thesis.

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Trondheim, September 2006



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- A List of abbreviations
- B Eco-efficiency in extended supply chains: A case study of furniture production
- C Eco-efficiency in redesigned extended supply chains; furniture as an example
- D Investigation of relationships in a supply chain in order to improve environmental performance
- E Environmental impact and added value in forestry operations in Norway
- F The importance of land use impact on biodiversity in an assessment of environmental performance of wood products

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1 Introduction

1.1 Background

This doctoral thesis is carried out as an integrated part of the research project 'Productivity 2005 Industrial Ecology' at the Norwegian University of Science and Technology (NTNU). The main objective with the project, as stated in the research plan, is *'to raise the level of expertise at NTNU, and disseminate knowledge on product, production and recycling systems through research and networking in such a way that the Norwegian manufacturing industry has access to candidates, expertise and methodology that will help companies implement more eco-effective and competitive solutions in such systems'* (Brattebø and Hanssen 2000).

P2005 Industrial Ecology is divided into two core projects and several horizontal activities. The core projects are 'Eco-effective products and production systems' and 'Eco-effective recycling systems and producer responsibility'. This thesis is a contribution under the first core project. The horizontal activities have focused on life cycle assessment (LCA), eco-responsible corporate culture, use of terminology, communication of industrial ecology, and consequences for industry resulting from the implementation of industrial ecology.

Brattebø and Hanssen (2000) identify three general research subjects within the core project on eco-effective products and production systems.

1. Methodologies for quantification of eco-effectiveness with regard to products, companies and networks of companies, and how to use this information in specific industrial cases.
2. Governmental regulations and financial instruments as promoters or barriers to development of eco-effective solutions in product and production systems.
3. Best organization, organizational learning and new ways of managing eco-effective companies and networks of companies in relation to product and production development.

These questions are later made more specific for eco-effective supply chains by Fet and Johansen (2000).

1. A methodological approach: What is an eco-efficient supply chain and how should eco-efficient supply chains be characterized, assessed and denoted?
2. A regulatory approach: How can measures on eco-efficiency bring forward regulations and standards for material efficiency, economic efficiency etc. in relation to the functional unit in a specific business sector?
3. An organizational approach: What benefits can be achieved through eco-efficient supply chains and company networks, and how should organizations develop to optimise eco-efficiency?

These questions constitute the basis for the formulation of this doctoral thesis.

1.2 Research goal and questions

The tripartition of the research focus both from Brattebø and Hanssen (2000) and Fet and Johansen (2000) is used to set the scope of this thesis. Three main questions are identified together with a number of sub-questions:

1. What is an eco-efficient supply chain?
 - a. What is a supply chain?
 - b. What is eco-efficiency?
 - i. How do we measure environmental performance in a supply chain?
 - ii. How do we measure value performance in a supply chain?
2. How can regulatory aspects be used to improve eco-efficiency?
3. How can organizational aspects be used to improve eco-efficiency?

The title of the thesis, '*Eco-efficiency in extended supply chains – methodological development with regulatory and organizational implications*', is a reflection of these questions.

1.2.1 *Eco-efficiency in the supply chain*

Huppés and Ishikawa (2005) state that there exists no agreed-upon method for assessing eco-efficiency and the concept of eco-efficiency and the related methodologies need clarification. Huppés and Ishikawa (2005) call for the development and clarification of terminology and basic method choices. Further, the tendency is to assess eco-efficiency for sites and not for larger systems. Even if this seems to be changing, assessments of eco-efficiency in supply chains are immature compared to assessments of single processes and production sites.

This thesis focuses on eco-efficiency in extended supply chains and is a contribution to understanding the applications of eco-efficiency. The scope of this thesis includes the investigation of how it is possible to assess and improve the eco-efficiency for existing products and use this knowledge in the development of new products. Measures of eco-efficiency can also be used as an element in marketing and as a part of the decision basis when procurements take place. An expansion of the system boundaries from the traditional site specific assessments to assessments including the extended supply chain is a prerequisite for these uses.

To be able to answer the first research question a number of sub-questions are identified. It is necessary to establish sound definitions of a supply chain and of eco-efficiency. To make it possible to assess eco-efficiency in supply chains, it is also necessary to clarify how environmental performance and value performance in a supply chain should be assessed.

1.2.2 *Regulatory aspects and improved eco-efficiency*

In society there is a general agreement that regulations are needed to address specific market failures (Bleischwitz et al. 2004). However, it is often argued that authorities should set targets for improvements, while industry should be allowed to determine how these targets can be met (Porter and van der Linde 1995; van den Akker 2000; Bleischwitz 2003).

New regulations might be introduced by both the authorities and industry. Independent of origin, new regulations will in most situations change the conditions for business and can therefore be used to motivate improvements in environmental performance.

The scope of this thesis includes investigation into how measures of eco-efficiency at a product level can be used to suggest regulations that effectively motivate improvements in environmental performance along the extended supply chains of the products.

1.2.3 *Organizational aspects and improved eco-efficiency*

Companies rely increasingly on their suppliers for competitive success (e.g. Hahn et al. 1990; Lambert and Cooper 2000) and this includes an increasing dependency on the environmental performance of the suppliers by extension. When evaluating the environmental issues related to a product's life cycle it is not sufficient to focus on single companies; the extended supply chain must be considered.

The scope of this thesis includes an exploration of how organizational aspects can be used to improve eco-efficiency. This includes both vertical relationships within the supply chains and also horizontal relationships within an industry sector.

1.3 Research methodology and outcome

The approach to the research questions has been through a combination of case studies and a recombination of already published theories and knowledge. Two case studies are conducted; one case study on furniture production and one case study on forestry and the delivery of logs to factories. The research has been exploratory by nature and based on the processes of systems engineering applied to understanding the interactions between elements in a supply chain.

A pragmatic approach is used to develop a methodology to assess the eco-efficiency of supply chains. Environmental assessments are often too complex to ever be 'complete' or free from assumptions (Wrisberg et al. 2002). However, in the absence of scientifically based recommendations, the chance of making a wrong decision is equal to the chance of making a right decision. It is thus better to use some assessments than none at all (Schaltegger and Sturm 1992; Bleischwitz et al. 2004).

The outcome of this thesis is a proposal for how eco-efficiency in extended supply chains should be assessed and understood, and how regulatory and organizational aspects influencing the supply chains could be used to improve the eco-efficiency of the product from a supply chain perspective. The proposed methodology makes it possible to assess the present eco-efficiency performance of a product and set targets for improvement in eco-efficiency. The resulting measurements together with knowledge of regulatory and organizational aspects make it possible to identify the path from the present situation to the target situation (Figure 1).

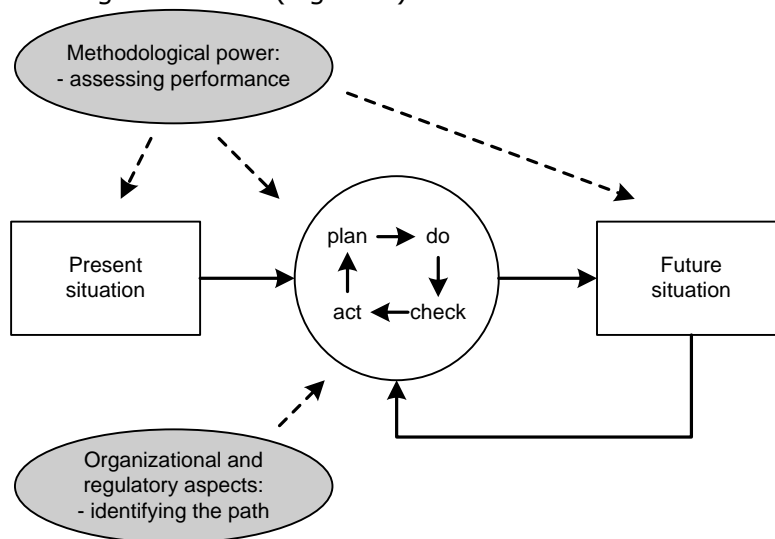


Figure 1 – Use of eco-efficiency assessments to identify the present situation, set targets, and identify the path to reach them

1.4 Structure

The principal work of this thesis is presented in the five research papers included as appendices (Appendices B-F). These are all independent papers dealing with the identified research questions based on relevant theory and case studies. The purpose of the following chapters is to place the papers in a context and provide a more thorough discussion of both the methodologies used and the findings.

The context for the thesis is presented in chapter 2. This chapter discusses the most important environmental issues within industry. The ideas of sustainable development and sustainability are presented in 2.1. The role of business and individual companies, and the motivation for environmental engagement are described in section 2.2, while supply chain considerations are introduced in section 2.3. In section 2.4 the concept of

industrial ecology is presented as a possible way for companies to actually address sustainability. Section 2.5 introduces eco-efficiency with a brief overview of the history and development of eco-efficiency both as a concept and as a tool for assessments, followed by a discussion of the present use of eco-efficiency.

Chapter 3 represents an overview of the most important analytical tools and methodologies used in the thesis. System engineering is used as a methodological superstructure for the thesis. An introduction to the concept of a system and how a system is constructed is presented in section 3.1.1 and the systems engineering process used for this thesis is described in section 3.1.2. The variety of ways of assessing environmental performance and value performance are presented and discussed in section 3.2.

The case studies used in the thesis are divided into two main thrusts; the furniture case presented in section 4.1, and the timber case in section 4.2. The cases are also discussed in the papers. The three first papers cover furniture production and the last two are from timber logging. Not all papers include all research questions, but through the five papers all identified research questions are addressed and discussed against both cases (see Figure 2).

Chapter 5 thoroughly explores the implications of the results presented in the papers. The main findings are highlighted and as clear recommendations as possible on the research questions are given. This chapter is divided in accordance with the research questions where the methodological questions are treated in section 5.1, regulatory issues are presented in section 5.2, and organizational issues are presented in section 5.3. In the last section in this chapter (5.4), the relationships between these three areas are discussed in order to put focus on the interaction between them and the necessity to focus on all three simultaneously in order to achieve significant improvements.

Six appendices are included. First a list of abbreviations used in the thesis is included (Appendix A). Then the five papers follow consecutively (Appendices B-F). Details on publication status and authorship are given in front of each paper.

Figure 2 is an overview of the construction of the thesis. The research questions are placed in a context and an analytical framework. The approach in the papers is a result of the identified research questions and available analytical tools. Together with relevant literature these form the basis for the recommendations given. The findings on all research questions can be reduced to the central issue for sustainability; how to achieve real improvements. Not all papers contribute to all research questions and this is shown with the lines in the figure. Dotted lines indicate that the question is addressed but not as the main topic of the paper.

The thesis is closed with chapter 6 where a final evaluation of the degree of goal achievement in the thesis is presented together with some ideas for further research.

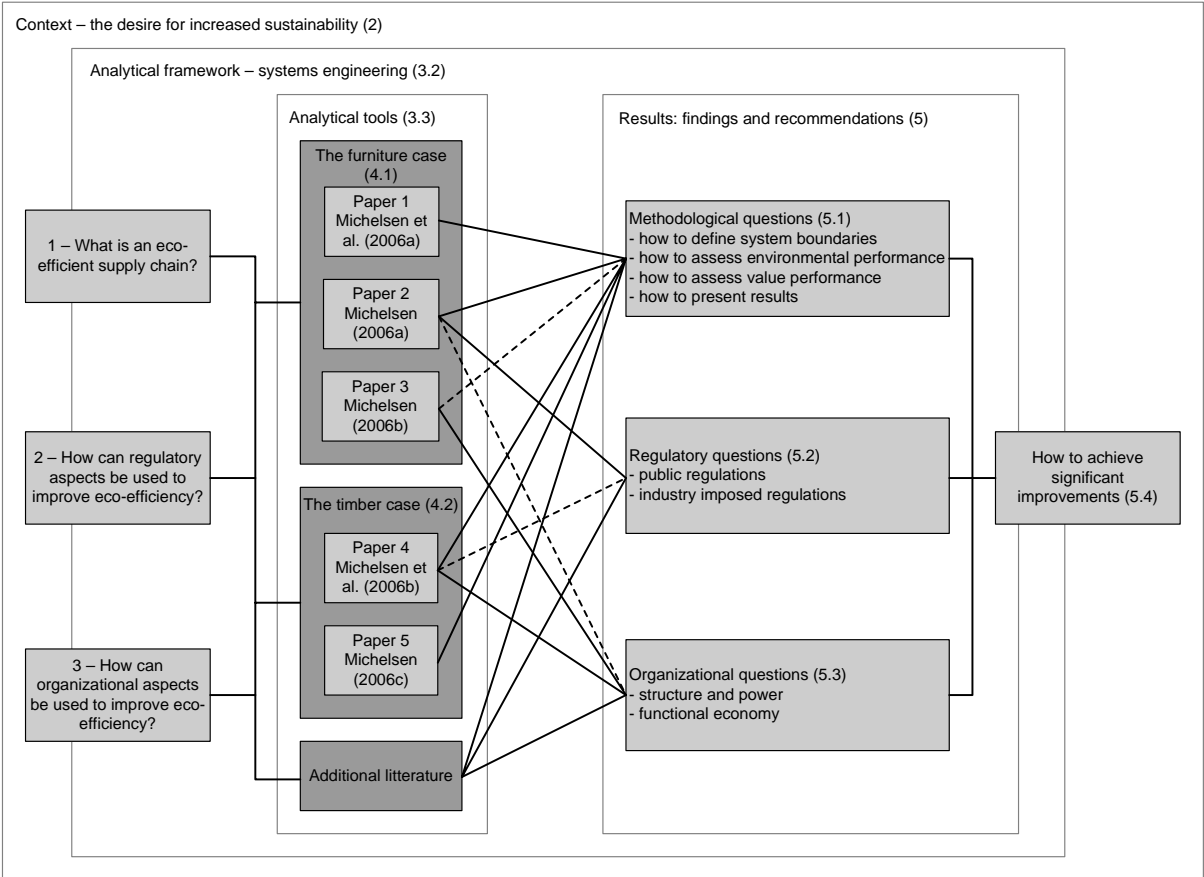


Figure 2 – The context and structure of the thesis

2 Sustainable Development, Industrial Ecology and Eco-efficiency

The context for this thesis is the need for increased sustainability. In this chapter the ideas of sustainable development and sustainability are discussed along with the role of business. Industrial Ecology is introduced as a possible way for companies to address sustainability, and assessments of eco-efficiency is here of high relevance.

2.1 Sustainable development

To reach environmental sustainability is one of the most pressing tasks for humanity and identified as one of the millennium developments goals by the United Nations (2006). The term 'sustainable development' gained attention with the World Commission on Environment and Development (1987), also known as the Brundtland Commission, who defined sustainable development as a '*development that meets the needs of the present without comprising the ability of future generations to meet their own needs*'. The definition is rather loose which also is criticized (e.g. Dyllick and Hockerts 2002; Huesemann 2003; Bartelmus et al. 2004). It fails to define what human needs actually are and also lacks a focus on environmental issues as a key concern.

A range of definitions are developed in response to this, and already in 1995 more than forty working definitions of sustainable development had appeared (Hajer 1995). A common trend is the increased focus on what has come to be known as the triple bottom line; concern for environmental, economic and social aspects simultaneously (e.g. Roome 1998).

These definitions are also criticized (see e.g. Simpson et al. 2004). Ehrenfeld (2005) attacks the term sustainable development in itself since it has incorporated a presumption on economic development. The jeopardy, he claims, is that the development only goes in a less unsustainable direction which is not the same as sustainability. Ehrenfeld (2000) argues that the whole idea of sustainable development, in particular as defined by the World Commission on Environment and Development (1987), is tied to one specific model of the world that at least in some situations fails to describe the real situation. As an alternative Ehrenfeld (2005) uses the term sustainability which he defines as '*the possibility that all forms of life will flourish forever*'. For human beings, this not only comprises survival and maintenance of the species, but also a sense of dignity and authenticity (Ehrenfeld 2005). Another interpretation is given by Clift (2000) who says that sustainability can be regarded as the target, while sustainable development is the process for achieving it.

The criticism against sustainable *development* as the target can be seen as a recognition of the fact that present human activity per se is a major component of the problem (e.g. Meadows et al. 1972; Daly 1990; Postel et al 1996; Vitousek et al. 1997; Clift 2003). To separate between the different factors creating an impact on the environment, Ehrlich introduced in 1968 what later has been known as the Ehrlich equation:

$$I=P \times A \times T \quad (1)$$

where I is the total environmental impact, P is the population, A is the consumption per capita (affluence) and T is a technology factor (Ehrlich 1968). This is sometimes called the 'master equation' (Graedel and Allenby 1995; Ehrenfeld 2000) and is specified by defining A as GDP/person and T as environmental impact/unit of GDP (Graedel and Allenby 1995).

The overall lesson is that total environmental impact is complex in its origin and can not be expressed by a linear relationship to one simple factor. Another severe problem with sustainability is that it is not really possible to measure it. Ehrenfeld (2000) says that it is only possible to retrospectively see if something has been sustainable. Nevertheless,

current knowledge does at least give a clue about what is necessary to move in a sustainable direction and often the focus is on doing things better than today (e.g. Schmidt et al. 2004).

In operational terms Daly (1990) identifies three general rules for sustainability;

1. The harvest rates on renewable resources should equal regeneration rates.
2. Waste emission rates should equal the natural assimilative capacities of the ecosystems into which the wastes are emitted.
3. Use of non-renewable resources must be paired with compensating investments in a renewable substitute.

The last rule is a quasi-sustainable use of resources according to Daly (1990), but the idea is that by the end of the life of the non-renewable resources, there is an annual sustainable yield equal to the previous use of the non-renewable resources, e.g. potential for production of biofuels instead of extraction of fossil fuel. In addition, Daly (1990) emphasises the need to extract more value per unit of resource, i.e. the efficiency with which the resources, renewable as well as non-renewable, are utilized.

2.2 Business and environmental performance

The role of business in the struggle to reach sustainability has been debated. The Business Council for Sustainable Development (BCSD - see Schmidheiny 1992) and later the World Business Council for Sustainable Development (WBCSD) argues that business should have a leading role, while e.g. Bakan (2005) argues that business hardly can be given the responsibility for anything except their own profit. Nevertheless, business is an essential part of society and thus essential for the direction of the development. John Browne, CEO of BP, claims that '*business is essential to delivering sustainability, because only business can produce the technological innovations and deliver the means for genuine progress on this front*' (in Handy 2002). And, he continues, business need a sustainable planet since they want to do business again and again. The natural environment serves industry in particular as a supplier of resources and as an absorber of emissions. Neither of these is unlimited.

However, even if business is an essential part of a move towards a more sustainable society, it is not obvious that the initiative to such a change primarily will originate from business. There are numerous examples of companies placing environmental issues on the agenda and according to Welford (1998) there are at least four reasons why environmental issues need to be addressed by companies; consumer pressure, potential cost savings, legislation and ethics. Similar ideas are presented by Brezet and van Hemel (1997), Hall (2000), Banerjee et al. (2003), Forman and Jørgensen (2004) and Simpson et al. (2004) to mention a few. Each of these drivers for environmental management is discussed.

2.2.1 Customer pressure

The importance of customer pressure is well documented. Hall (2000) has shown a clear relationship between the pressures companies experience and the actions initiated. The environmental performance of products is also often important when decisions on procurement take place (i.e. Dahl et al., 2002; de Bakker et al., 2002) even if this is far from universal (Vogtländer et al., 2002; Banerjee et al. 2003).

There is however reason to believe that this trend will continue with the increasing focus on 'green procurement' as a catalyst (Sips 2000; The European Commission 2004, Michelsen et al. 2006a). In Norway the Public Procurement Act states that all official bodies have a legal obligation to take environmental performance of products into consideration when new acquisitions are planned (The Norwegian Ministry of Government Administration and Reform 1999), and The European Commission (2003) has announced ambitious goals for green procurement. The Public Procurement Act was not instantly

implemented in procurement practices, but there is now a significant increase in environmental demands in announcements of tenders in Norway. In 2004 some sort of environmental requirement was put forward in 58 percent of the tender announcements, and in 2005 this number had increased to 66 percent (Solevåg 2005). There is no doubt that environmental performance in at least some cases is vital for the ability to win a tender, and The Court of Justice of the European Communities has clearly stated that differences in environmental performance is a legal criteria for contract awards (e.g. the 'Concordia case'¹).

Using statistics for the year 1998 from the OECD, in Norway, public procurement represented 19% of GDP, while the same number in EU 15 is 18% (OECD 2000). Given the importance of public procurement, it is expected that increased focus on environmental performance in the public sector will have a great impact in behaviour in at least some business sectors.

Zadek (2004) shows the relation between the maturity of an issue in society, e.g. environmental concern, and the necessary response from the company (Figure 3). This shows that companies can take a defensive attitude toward issues that are latent in society, but that companies increase their focus when the issue matures in society.

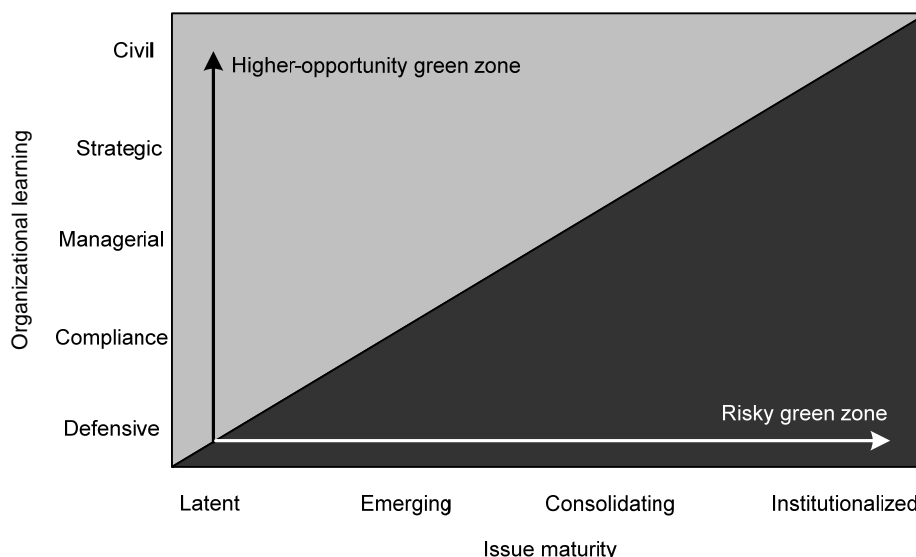


Figure 3 – The relation between the maturity of an issue in society and necessity to business response (from Zadek 2004)

This also suggests a market for green niche products (cf. Brezet and van Hemel 1997). Business opportunities exist for products that require less energy and resources, generate less waste, or contained fewer regulated substances (Hunkeler et al. 2004; von Geibler et al. 2004) and proactive companies can create their own markets with new products and thus increase their overall market share.

2.2.2 Potential cost savings

The next important driver for environmental management is opportunities for cost savings. Welford (1998) presents an overview of potential cost savings due to environmental management. Kiernan (1996) states that all over the world the link between environmental performance, competitiveness and bottom-line financial results is growing tighter every day. One important cause is the use and loss of resources such as energy and raw materials. All wastes are potential raw materials, and a decrease in generation of waste and consumption of energy and raw materials will in most cases

¹ Judgement of the court 17 September 2002, <http://curia.eu.int/>

result in lower costs and reduced environmental impact simultaneously. This is one of the core issues in eco-efficiency that will be discussed more in detail in section 2.5. Porter and van der Linde (1995) have found that just the mere focus on measuring environmental performance alone leads to enormous opportunities to improve productivity, and the view that there is a trade-off between financial and environmental issues should be rejected, at least as a general rule. There are however differences between different industry sectors (Handfield et al. 1997; Banerjee et al. 2003) and in some cases a trade-off between environmental and financial concerns may exist but becomes less relevant the longer a company has dealt with such issues (Handfield et al. 2005).

2.2.3 Legislation

When it comes to environmental regulations, there is still a common view that these hamper productivity growth (Telle and Larsson 2004). However, Telle and Larsson (2004) do not find support for this view. On the contrary, they find a positive relation between regulations and productivity growth when reductions in emissions are defined as a product. This is of increasing relevance, e.g. with tradable emission quotas for greenhouse gasses.

Some companies have seen it as an advantage to be ahead of regulations to avoid ad hoc actions (Klassen and McLaughlin 1996; Lamming and Hampson 1996; Hunkeler et al. 2004; Zadek 2004) and also influence development of new legislation (Barret 1992; Taylor 1992; Chynoweth and Kirschner 1993, Handfield et al. 1997). Since legislation often is based on best practice or best available technology, proactive companies have the possibility to set the standard and thus define the rules for more reactive competitors. Regulations will always be behind best practice and according to Handy (2002) companies need to take the lead in areas such as environment instead of being put on the defensive. Figure 4 shows a schematic picture of this where the trend setters have higher environmental costs in the beginning (at time t_1), which eventually level off (t_2). Followers, on the other hand, experience escalating costs caused by the need to take more ad-hoc actions to catch up, e.g. due to stronger regulative pressure (from Brezet and van Hemel 1997).

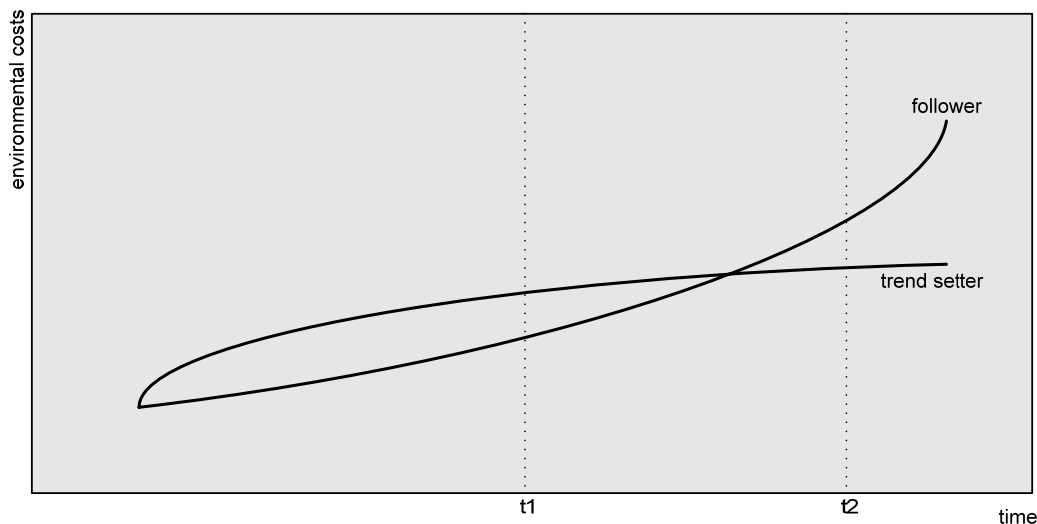


Figure 4 – Development of environmental costs for proactive trend setters and reactive followers (from Brezet and van Hemel 1997)

Sharma and Vredenburg (1998) found that companies that keep ahead of regulations also benefited in other ways. They often were given the 'benefit-of-the-doubt' in case of minor infractions and were also able to go through public consultation hearings and approval processes for new developments much faster. All this reduced costs

significantly. Porter and van der Linde (1995) and Handfield et al. (1997) also show that new environmental regulations might make companies aware of existing cost saving potentials and thus be beneficial. Hunkeler et al. (2004) show that companies tend to only chose the 'low-hanging fruits' with a fast return on investments. Profitable innovations might not be considered without the motivation of other factors, such as new regulations.

It is, however, not always obvious what the outcome of proactive behaviour will be. In Norway, chromium is still used in skin and leather tanning. At the same time, chromium is on the list of priority substances to be controlled and emissions are to be substantially reduced by 2010 (The Norwegian Pollution Control Authority 2006). As a consequence of this target, some furniture companies have banned chromium in the tanning process, while others argue that this substantially reduces the quality and have done little to reduce the consumption. If chromium is forbidden in the furniture industry, the proactive companies will then have a lead on their competitors in technology change. If not, they have incurred, at least in someone's view, unnecessary costs.

Porter and van der Linde (1995) do however stress that new environmental regulation must be wisely directed; regulators must set targets for improvements and environmental impact, but at the same time create maximum opportunity for innovation by letting industry discover how to solve their problems.

2.2.4 Ethics

The last reason mentioned by Welford (1998) for a company to consider its environmental performance is ethics. Handy (2002) claims the whole purpose of a business is not simply to make a profit, but to make a profit so that the business can do something more or better. That 'something' is the real justification for the business. One severe effect of perceived unethical behaviour of a company is reduced ability to recruit new personnel, especially at the top level (Wrisberg et al. 2002).

In a study of the furniture industry in USA, the company that had the best overall environmental performance did not regard themselves as a green company (Handfield et al. 1997). Environmental issues were just an integrated part of all of its business activities. Equal effects are reported when environmental management is integrated in total quality management; the companies increase their environmental performance as a consequence of their quality management (cf. Klassen and McLaughlin 1993; Lamming and Hampson 1996).

2.3 Product life cycles and supply chain management

As stated earlier, companies increasingly rely on their suppliers for competitive success. This means that not only the individual companies are competitors, but also the supply chain as a unit (Hahn et al. 1990; Christopher 1998; Handfield and Nichols 1999; Kaplinsky 2000; Lambert and Cooper 2000; Mentzer et al. 2000; Mont 2002; Hagelaar et al. 2004). As a consequence, companies have also experienced an increasing dependency on the environmental performance of the suppliers, especially when it comes to the performance of products (see Michelsen et al. 2006a). Concepts as Life Cycle Management (LCM) and Supply Chain Management (SCM) and corresponding policy principles as Extended Producer Responsibility (EPR) and Integrated Product Policy (IPP) have occurred as a response to this.

The pressure to provide environmental information and carry out improvements is not evenly distributed in the supply chain. The end producers are in general more exposed than their suppliers (Hall 2000), and an important task is therefore to disperse the focus on environmental performance throughout the supply chain.

According to Cox (1999), a great deal of supply chain management today can be tracked back to the 'lean thinking' originally introduced by Toyota. Interestingly, one of the important characteristics was the focus on the elimination of waste in all processes, both internally and externally. The motivation was mainly financial, but improvements in environmental performance, measured as waste reduction, were obtained in the entire supply chain. Hunkeler et al. (2004) also focus on the relation between environmental and economic aspects in the supply chain and state that the LCM approach includes both, closely connected to sustainable development.

Hunkeler et al. (2004) define LCM as '*an integrated framework of concepts and techniques to address environmental, economic, technological and social aspects of products, services and organizations*'. Also others stress the environmental content in LCM where the temporal boundaries are set by the product life cycle (see Seuring 2004 for an overview). One of the most central analytical tools in LCM is Life Cycle Assessment (LCA) which will be described in section 3.2.

Life cycle considerations, including resource use, transportation, product, component, or material reuse or recycling, and disposal are most effective when the entire life cycle is assessed (Wrisberg et al. 2002; Hunkeler et al. 2004). The strength of a life cycle perspective is the possibility afforded to identify trade-offs between manufacture, use, and end-of-life treatment and to avoid different types of problem shifting (Wrisberg et al. 2002). This knowledge is crucial in the product design phase since often more than 70% of the environmental burdens and costs are fixed during this phase (Blanchard 1991; Fabrycky and Blanchard 1991; Handfield et al. 1997; Asiedu and Gu 1998; Rebitzer 2002).

Even though the supply chain under many circumstances is the competing unit, there are of course conflicts between participants in supply chains (Cox 1999) and there is no direct link between LCM and an actual management of the supply chain (Seuring 2004). Here the concept of Supply Chain Management (SCM) appears. Christopher (1998) defines a supply chain as '*the network of organizations that are involved, through upstream and downstream linkages, in the different processes and activities that produce value in the form of products and services in the hand of the ultimate consumer*.' All supply chains are in principle infinite, and the criteria for selection of boundaries must be set to make them manageable. In other definitions the importance of information flows is given more emphasis (e.g. Handfield and Nichols 1999; Seuring 2004). As the many definitions of SCM show, the environmental focus is not as pronounced in SCM as in LCM (Seuring 2004). In cases where environmental issues are an important part of the SCM, this is sometimes highlighted through the use of terms as Environmental Supply Chain Management (ESCM), Green Supply Chain Management or Sustainable Supply Chain Management (cf. Forman and Jørgensen 2004; Seuring 2004).

One interesting point is that it seems as those working with SCM with an environmental content also include the end-of-life treatment (e.g. Handfield and Nichols 1999; Hagelaar et al. 2004), while this is rarely the situation when the environmental focus is absent. Thus, the environmental focus results in a life cycle focus on the products, which is not a part of traditional SCM.

Christopher (1998) has expanded the definition of SCM to what he calls the 'Extended Supply Chain' (ESC) which includes use as well as end-of-life treatment of the products. This term emphasises the focus on the companies involved and incorporates the life cycle perspective. This is in accordance with the perspective in this thesis where the products and their life cycles are in focus. The term 'extended supply chain' is thus used to describe the systems in the case study on furniture production (see section 4.1; Michelsen 2006a; Michelsen et al. 2006a).

It is important to recognize that when a supplier is selected, more than the requested item is delivered. The waste and emissions created during the production, and the contribution to waste and emissions during use and end-of-life treatment of the final product, is also delivered and factors into the overall performance. To measure this performance, it is important for an end-product manufacturer to pass on the focus on environmental performance to other actors in the supply chain.

However, a company's network horizon is often rather narrow (Håkansson and Johanson 1992; Lambert and Cooper 2000; Holmen and Pedersen 2003). Even if they require information on environmental performance from the actors they know, this will in most cases be a limited part of the supply chain. In most cases it is neither possible nor practical to have too much knowledge about a large part of the supply chain (Håkansson and Snehota 1995). An additional problem is that the most severe environmental impacts in general originate from the early stages of the supply chain and especially during extraction of raw materials (Clift and Wright 2000; Dahlström and Ekins 2006) and thus far away from the focus of the manufacturers that provide the final products for the market. It is thus not sufficient to have information on environmental performance of first tier suppliers, it is also necessary to get information from their suppliers and sub-suppliers.

A possible solution to this dilemma is to use environmental performance criteria as an order qualifier and make sure that the suppliers bring this forward to their suppliers during procurement decisions (e.g. Handfield et al. 2005). When effective, this gives a cascade effect in the supply chain. The timber case described in section 4.2 provides an example of this (see also Michelsen et al. 2006b). Buyer-supplier relations play an increasingly important role in the strategies of firms, also when it comes to environmental performance (Handfield et al. 1997; Hall 2000; Håkansson and Waluszewski 2002; Handfield et al. 2005) and purchasing strategies can be the first step in developing a supply chain strategy (Pagell and Krause 2002; von Geibler et al. 2004).

A last issue that should be mentioned is the tendency that small and medium sized enterprises (SMEs) often lack the capacity to implement environmental improvements (Handfield et al. 2005). Stokes and Rutherford (2000) have revealed that SMEs in addition have a lack of awareness about environmental legislation. As a consequence, there are several examples of companies helping their suppliers to implement environmental management systems and improve environmental performance (Taylor 1992; Handfield et al. 1997; Tukker 2004; Handfield et al. 2005). The extreme consequence is to 'insource' environmental problematic processes to secure that these are properly taken care of (Handfield et al. 1997). Make or buy questions are thus an obvious issue also of environmental supply chain management.

2.4 Industrial Ecology

A critical question is how business, and the society as a whole, actually is going to take the move in the direction of sustainability. Here the concept of Industrial Ecology (IE) is proposed to fill the gap and Allenby (1999) calls industrial ecology the 'science of sustainability'. According to Erkman (1997), the idea of IE is to understand how an industrial system works, how it is regulated, and its interaction with the biosphere. Holistic system thinking is thus essential in IE. It is commonly recognized that today's problem are results of yesterday's solutions (Graedel and Allenby 1995) and it is thus obvious that it is necessary to expand the boundaries, both spatial and temporal, to avoid that today's solutions in the same manner cause the problems of tomorrow.

Then, on the basis of the knowledge about ecosystems, it must be determined how industrial systems can be restructured to make them compatible with the way natural ecosystems function. On the one hand ecosystems provide information on what to do. It is debatable whether an analogy exists between natural ecosystems and industrial systems, or if natural ecosystems only should be used as metaphors (e.g. Korhonen

2004). On the other hand, ecosystems also provide information on what not to do through identification of existing ecological constraints for industry (Daly 1990; Harte in Anonymous 2001; Lifset and Graedel 2002).

According to Erkman (1997), the concept of industrial ecology existed before the term which appeared in the literature in the 1970s. However, present use and development of the concept originates from an article of Frosch and Gallapagos (1989) which established a new starting point for development of IE after earlier sporadic attempts. Several definitions of industrial ecology are proposed. One commonly quoted definition is proposed by White (1994) who defines IE as *'the study of the flows of materials and energy in industrial and consumer activities, of the effects of these flows to the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources'*.

The definition is tripartite. First, there is a focus on the flows of materials and energy. This is not limited to activities within industry, but includes consumer activities as well. This incorporates what is commonly known as industrial metabolism (e.g. Erkman 1999), although IE goes beyond this. Second, there is a focus on the actual effects of these flows. This means that it is not enough to study the mere attributes of the flows; the impact on the environment of the different flows must be assessed. Third, the impacts of diverse factors on the flows are included.

Ehrenfeld (2000) states that industrial ecology as it exists today has two related but distinctive shapes. One is paradigmatic, normative and metaphorical; the second is descriptive and analytic. In this thesis the main focus is on the analytical part and in section 3.2 some of the common tools used in IE will be discussed. The concept of eco-efficiency has turned out to be a central part of this (e.g. Huesemann 2003). However, the paradigmatic view is already touched upon. In section 2.2 it is shown that the earlier conceived antagonism between competitive advantage and environmental improvements and constraints do not necessarily exist and new insight on the relationship between environmental concern and business management and strategies are thus emerging.

Ehrenfeld (1997) summarizes IE in four basic rules:

1. Close material loops.
2. Use energy in a thermodynamic manner; employ energy cascades.
3. Avoid upsetting the system metabolism; eliminate materials or wastes that upset living or inanimate components of the system.
4. Dematerialize; deliver the function with fewer materials.

In particular the first of these rules is given great attention. Graedel and Allenby (1995) summarized this as the move from systems based on unlimited resources and unlimited sinks for wastes (type I systems) to systems only open for energy input (type III systems) (Figure 5). The IE challenge is to move towards this hypothetical industrial type III system as far as possible.

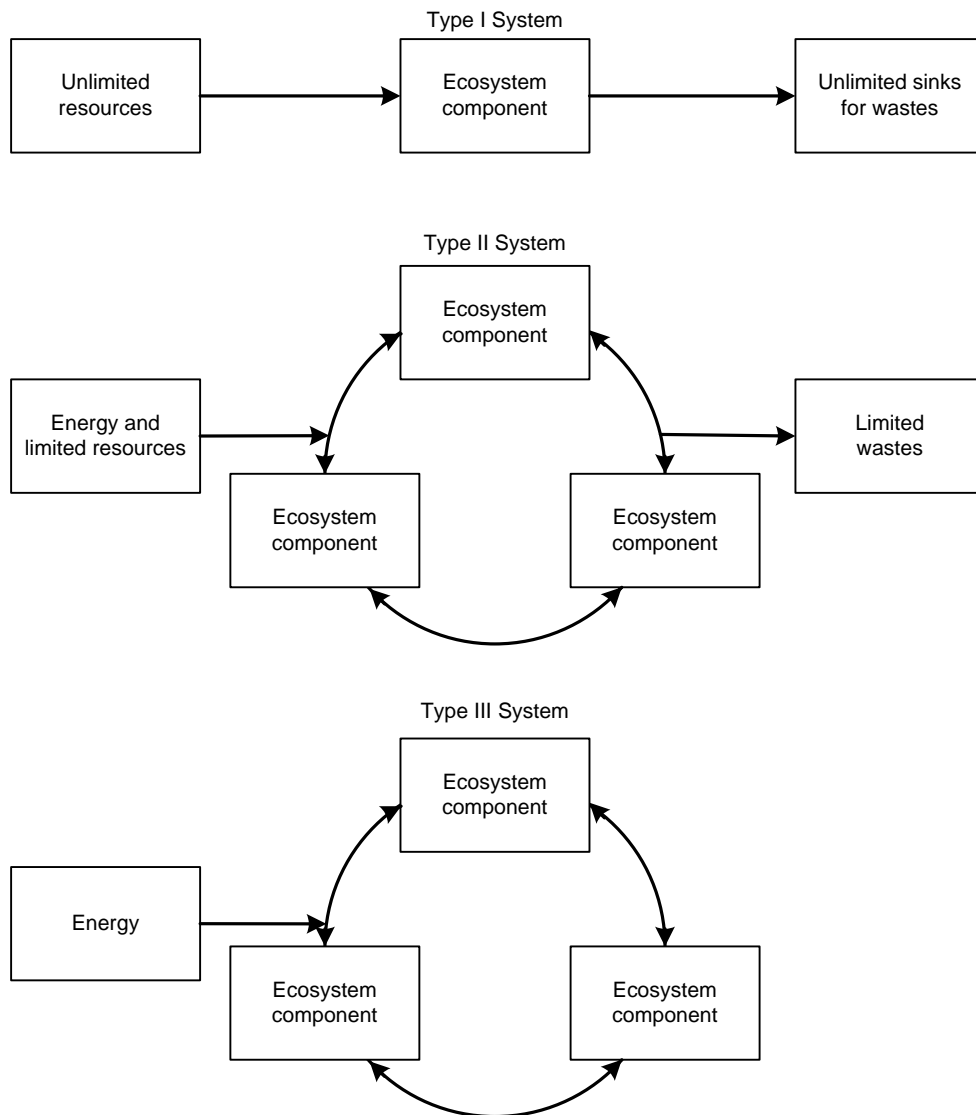


Figure 5 – One of the main issues in industrial ecology is to move from linear type I systems to type III systems with closed material cycles (from Graedel and Allenby 1995)

2.5 Eco-efficiency

The term eco-efficiency (E/E) was introduced in the late 1980s (Schaltegger and Sturm 1989) and appeared in academic literature for the first time in 1990 (Schaltegger and Sturm 1990). Eco-efficiency is seen both as a concept and as a tool where the basic idea is to produce more with less impact on nature, measured as reduced emissions or reduced raw material consumption, or both (e.g. Schaltegger and Burrit 2000). The idea is however older; when 3M introduced their Pollution Prevention Pays (3P) programme in 1975, the idea was to simultaneously make environmental and financial improvements (Ruud 2002).

Eco-efficiency soon gained popularity after Schmidheiny (1992) popularised the term in the book *Changing Course* which was presented at the United Nations Conference on Environment and Development (UNCED) summit in Rio. This book is in fact often quoted as the origin of the term eco-efficiency. Here, Schmidheiny (1992) primarily put the focus on nature's capacity to absorb wastes and called for reduced emissions along with continued economic growth. The Business Council for Sustainable Development (BCSD, since 1995 changed to the World Business Council for Sustainable Development - WBCSD) has played an important role in making the concept known within industry and

is for instance running a web-page with information and examples on eco-efficiency in industry². Within business, many look upon eco-efficiency as the main corporate response to the call for sustainable development (DeSimone and Popoff 1997; Dyllick and Hockerts 2002).

One of the most quoted definitions is from WBCSD that defines E/E as *'the delivery of competitively priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impact and resource intensity throughout the life cycle, to a level at least in line with the earth's estimated carrying capacity'* (DeSimone and Popoff 1997). Another widely used definition is from OECD that defines eco-efficiency as *'the efficiency with which environmental resources are used to meet human needs'* (OECD 1998).

A major difference between these two is the inclusion of the carrying capacity in the WBCSD-definition, while OECD looks upon eco-efficiency as a straight forward measure of the exploitation ratio of the resources that are introduced to the economy.

However, when E/E is operationalised, this distinction is of minor importance since the most common interpretation of eco-efficiency has been the ratio between a product or a service value, and the environmental impact caused by the delivery where the total volume of economic activity is not included (e.g. Verfaillie and Bidwell 2000):

$$\text{eco-efficiency} = \frac{\text{product or service value}}{\text{environmental impact}} \quad (2)$$

Using this interpretation of E/E, it is obvious that the measures of eco-efficiency only cover part of the sustainability concept. First, eco-efficiency only considers environmental and economic performance, while the social dimension of sustainability is omitted. Second, also when it comes to total environmental impact, it is apparent that only a fraction of the Ehrlich equation (equation 1) is included, namely the technology-factor (cf. Graedel and Allenby 1995, see section 2.1). Schaltegger and Burritt (2000) use the notion 'sustainable improvement' as a target for eco-efficiency. Never the less, this interpretation is often used and is in particular found useful for assessing changes over time (Fet and Michelsen 2003, see also Figure 7).

The equation above occurs in a range of varieties, but they all relate environmental and value performance to each other. Possible measures of environmental performance are given in section 3.2.1 while possible measures of value performance are given in section 3.2.2.

Eco-efficiency can also be presented in xy-diagrams as shown in Figure 6. Here the environmental and value performance is plotted in the diagram and the two are not merged into a single indicator as in equation 2. However, the eco-efficiency ratio is still used to some degree in the figure; all points found at the same side of the eco-efficiency line and with the same distance to the line will have the same score following equation 2. To have a strong sustainable improvement (cf. Schaltegger and Burritt 2000), both environmental and value performance must be improved, recognized as a win-win situation. This is shown by arrow A in the figure. If only one of the aspects is improved (B or C), there is only a weak sustainable improvement according to Schaltegger and Burritt (2000). As the figure shows, the eco-efficiency is improved also in situation B even if the environmental impact actually increases. Following equation 2 the eco-efficiency improvements are equal in situations B and C.

This is also addressed as absolute versus relative decoupling. A relative decoupling is taking place if the increase of products or service value is higher than the increase in

² <http://www.wbcsd.org/>

environmental impact (cf. B in Figure 6), as distinct from absolute decoupling where the total environmental impact decreases while the output increases (cf. A in Figure 6 - OECD 2002; Bartelmus et al. 2004).

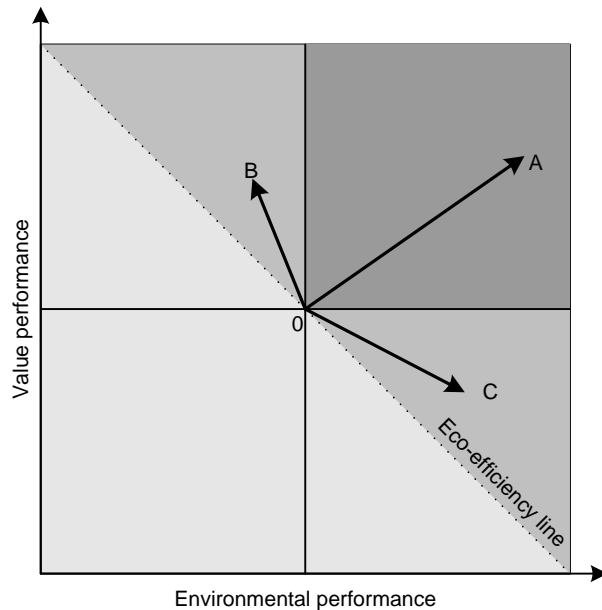


Figure 6 – The eco-efficiency matrix and differentiation between strong sustainable improvement (A) and weak sustainable improvement (B and C) (from Schaltegger and Burritt 2000)

Even though E/E does not cover the entire sustainability concept, it constitutes a good starting point for sustainability assessment if social aspects later on are added. This is already attempted by the company BASF (Kicherer et al. 2004; Schmidt et al. 2004). Eco-efficiency has also been supported very prominently as being a suitable goal for top management to adopt (e.g. Schmidheiny 1992; OECD 1998, Schaltegger and Burritt 2000) and is supported by influential organizations such as WBCSD and OECD.

2.5.1 Present use of eco-efficiency

The concept of eco-efficiency has developed quickly since its introduction. In addition to encouraging more efficient use of resources, two distinct areas for eco-efficiency assessments are identified; as a tool to measure performance at a system level (process, product, company etc.), and as a tool to compare different alternatives (benchmarking). In both situations the underlying motivation is to improve business performance.

Depending on the area of application, eco-efficiency performance is reported mainly in two ways. When performance and improvements for one system is measured, there is a clear tendency to use the eco-efficiency ratio (equation 2). This allows presentation of time scales that provide understandable information on the changes in performance (see Fet and Michelsen 2003). This has up till now been the dominate way of operationalising eco-efficiency (e.g. Verfaillie and Bidwell 2000) and there are published guidelines on eco-efficiency that focus solely on this ratio (Sturm et al. 2003).

The second major technique for presenting eco-efficiency, is the use of xy-diagrams, e.g. as portfolio matrixes. This has gradually developed (see Lee and Green 1994; Ilinitich and Schaltegger 1995; Schaltegger and Sturm 1998) and is now used by companies such as BASF (Saling et al. 2002) and TNO (Eggels et al 2001). This way of presenting eco-efficiency is also adopted at Delft University of Technology (Huisman 2003) and the Norwegian University of Science and Technology (e.g. Bohne 2005; Michelsen 2006a; Michelsen et al. 2006a). The basic concept is shown in Figure 6. Using the diagrams requires that a reference point is established, e.g. an average value (see Michelsen et al.

2006a) or a specific solution (see Michelsen 2006a). Then, other solutions (e.g. products or processes) are compared to this. In most cases relative numbers are used although absolute numbers might be used as well.

The range of applications for eco-efficiency is shown in Table 1. During the last years it has been a tendency to expand the boundaries for eco-efficiency analyses and more attention is given to the topics in the lower right corner of Table 1. This was e.g. obvious at the Conference on Quantified Eco-Efficiency in Leiden in April 2004³.

Table 1 – Applications of eco-efficiency

	Single site	Life cycle/ESC
Sector	National statistics	
Company	WBCSD ¹ , IChemE ²	
Business	WBCSD ¹ , IChemE ²	OBIA ³
Product	E.g. energy labels	Ecolabels, BASF ⁴ , EPDs

1 - <http://www.wbcds.org/>

2 - <http://www.icheme.org.uk/>

3 - see Taylor and Postlethwaite (1996), Clift and Wright (2000), Clift (2003)

4 - <http://www.corporate.basf.com/en/sustainability/>, Saling et al. (2002)

2.5.2 Targets for eco-efficiency improvements

As already stated, the basic idea of eco-efficiency is to reduce the environmental impact of economic activities. The same basic idea constitutes the basis for the 'Factor X' concepts where X denotes the reduction in environmental impact of economic activities. X is used in the range from 4 to 50 (Reijnders 1998), but 'Factor 4' and 'Factor 10' are in particular prominent and used as strategic goals within industry (Bartelmus et al. 2004). Factor 4 originates from the idea of doubling wealth while halving the resource consumption (von Weizsäcker et al. 1997). The target for improvements in eco-efficiency is thus a factor of 4. Factor 10 originates from Schmidt-Bleek (1994) and is in principle the same with the exception that he argues that it is technically feasible to increase the resource efficiency tenfold. Both concepts are primarily based on the assumption that the globally available environmental space is already overused and it is necessary to reduce the environmental pressure to about half of the present situation (Bleischwitz et al. 2004).

The concept of eco-efficiency is criticized for its shortcomings. There is a lack of focus on the absolute environmental impact, local conditions are not taken into consideration (e.g. in the quality of recipients), and the result at best is that the present unsustainable practices are slowed down (McDonough and Braungart 2000; Hukkinen 2001; Dyllick and Hockerts 2002; Huesemann 2003). Huesemann (2003) pinpoints that *T* in the Ehrlich equation never can reach zero, and as long as *P* and/or *A* are growing, these factors must also be controlled to achieve sustainability (see section 2.1). Even though improvements might occur in the present total environmental impact, this improved performance will be limited as long as the growth continues and the possibilities to increase the efficiency lessened.

There are in fact several examples of how improved efficiency is eaten up by increased economic activity (e.g. Ehrlich et al. 1999). Figure 7 shows an example of this based on data from oil and gas extraction in Norway. The emissions to water per produced volume are decreased by 80% in the period from 1984 to 2004, but still the total emissions are not reduced and have in fact increased almost fourfold from 1993 until the present due to the increased production that is no longer followed by improvements in eco-efficiency. In fact, Huesemann (2003) demonstrates that improvements in eco-efficiency are a step in the wrong direction if the motivation is to enhance economic growth and not to move towards sustainability.

³ See special issue of *Journal of Industrial Ecology* Vol. 9, No. 4

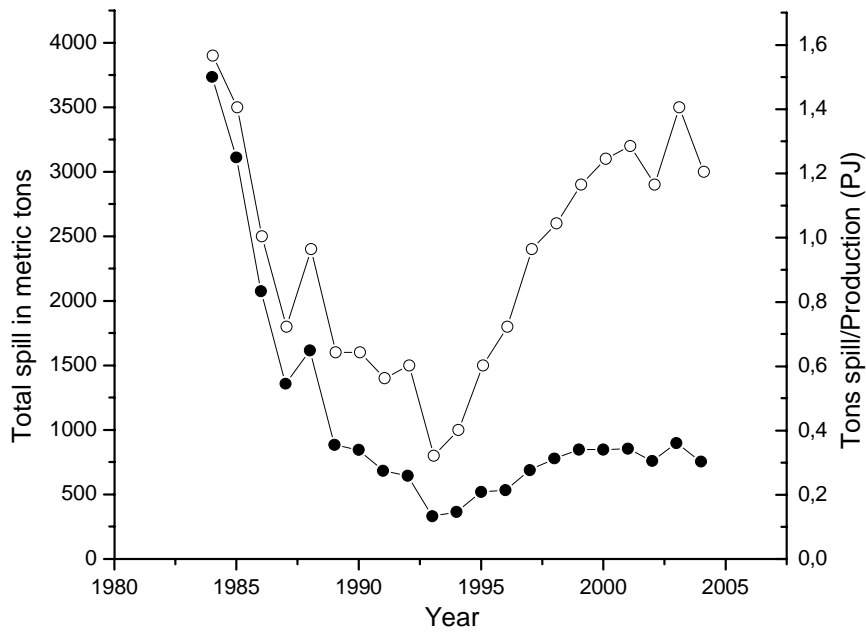


Figure 7 – Changes in absolute spills from offshore oil and gas production in Norway (open circles) and eco-efficiency measured as spills per produced PJ (solid circles) (data from Statistics Norway 2005a)

The idea of Factor X reflects a technological optimism (Reijnders 1998). Figure 8 shows different levels of environmental improvement based on level of innovation (Brezet 1997). The two first options are often called technical eco-efficiency improvements since they have an existing technical system as a starting point, while the two last options can be called functional eco-efficiency since here a function is the starting point and new technical systems are needed to ensure the required level of improvement. However, Huesemann (2003) raises the question if E/E is the right tool to assess large system improvements.

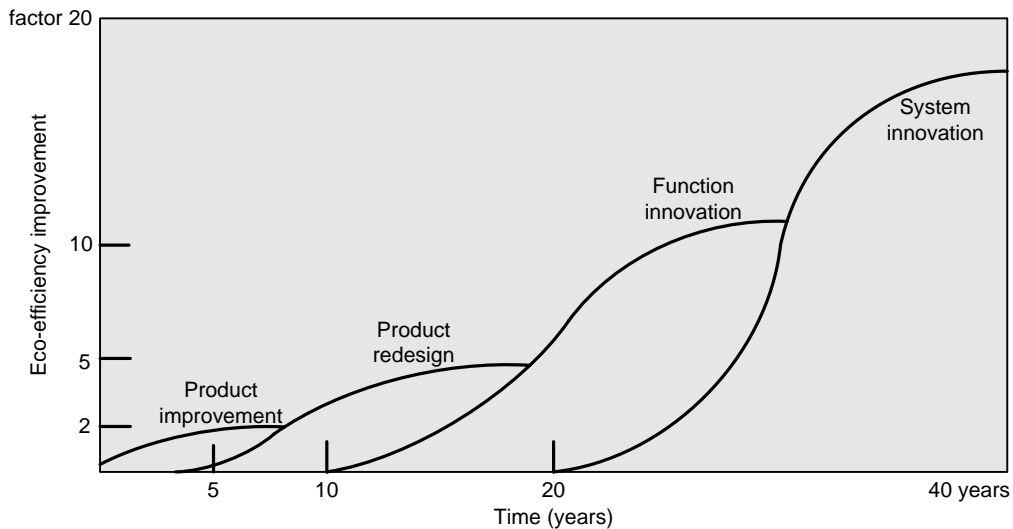


Figure 8 – Different levels of environmental improvements related to level of innovation (from Brezet 1997)

Environmental awareness is often identified as an important driver for innovations (Bleischwitz 2004) where the awareness might originate from all sources mentioned in

section 2.2. It is important to stress that environmental driven innovations do not differ from other innovations and an important question for industry is if environmentally awareness and search for environmental improvements is a part of a general business strategy (Handfield et al. 1997; Noci and Verganti 1999; Hunkeler et al. 2004). Porter and van der Linde (1995) argue that an external pressure often is necessary to make the companies aware of the possibilities.

In addition to eco-efficiency, some have introduced the term eco-effectiveness to also include the *total* environmental impact of an activity (e.g. Hukkinen 2001; Dyllick and Hockerts 2002; Figge and Hahn 2004). This is a response to the need for absolute values for sustainability measures and thus overcomes one of the shortcomings of eco-efficiency. Hanssen (1999b) defines eco-effectiveness as '*a measure first to reduce or modify the need of a certain function, and then find the most efficient solution to fulfil the function*'. According to Brattembø et al. (1999) this can be formulated as

$$\text{Eco-effectiveness} = \text{eco-efficiency} \times \text{total volume of activity} \quad (3)$$

In this section the idea of eco-efficiency assessment is introduced. In chapter 3 possible measures of environmental performance and value performance will be presented, while the applicability of the assessments will be discussed in chapter 5.

3 Analytical tools and methodology

The case work for this thesis is based on a multidisciplinary approach where different analytical tools and methodologies are combined in order to achieve the final results. To organize this process, a system engineering approach is used. The identification of a system is discussed in section 3.1.1. The system engineering approach is presented in section 3.1.2 and the analytical tools most relevant to environmental and value performance are presented and discussed in section 3.2.

3.1 Systems Engineering

Systems Engineering (SE) is regarded as both a discipline and a process. The thesis activities made use of the SE processes (section 3.1.2). Blanchard (1991) defines system engineering (SE) as *'the effective application of scientific and engineering efforts to transform an operational need into a defined system configuration through the top-down iterative process of requirements definition, functional analysis, synthesis, optimization, design, test and evaluation.'* Blanchard (1991) put emphasis on four important areas for success:

1. A top-down approach where the system as a whole is viewed.
2. A life-cycle orientation where all phases of the system are addressed.
3. A thorough identification of the system requirements.
4. An interdisciplinary approach to ensure that objectives are met in an effective manner.

The system perspective of SE is also an important element in industrial ecology (e.g. Lifset and Graedel 2002) and system oriented analytical tools are commonly applied, as described in section 3.2.

3.1.1 Defining the system of interest

The term 'system' comes from the Greek word 'systēma' meaning an organized whole. Blanchard (1991) states that a system constitutes a set of interrelated components working together with the common objective of fulfilling some designated need. He says that a system is recognized by four general characteristics:

1. A system constitutes a complex combination of resources.
2. A system is contained within some form of hierarchy.
3. A system may be broken down into subsystems and related components which interact with each other; subsystems and components are represented in respective layers of the hierarchy.
4. A system must have a purpose and be able to respond to some identified need.

The value of the system is determined by the degree to which it meets the identified need in a satisfactory and efficient manner (Blanchard 1991). Different kinds of systems can be identified (Blanchard 1991; Wrisberg et al. 2002) and as pointed out in section 1.2.1, the focus here is on product systems. In this thesis, the system-of-interest is the life cycle of the products and has a distinctly functional orientation.

There are two important dimensional aspects with such function-oriented systems according to Wrisberg et al. (2002). The spatial dimension includes processes upstream as well as downstream with respect to a core process. Analyses of such systems are thus often recognized as 'cradle-to-grave analyses'. The time dimension is also dealt with in a comparable way. Wrisberg et al. (2002) underline that processes both in the past (e.g. design and raw material extraction) and in the future (e.g. end-of-life treatment) that are necessary for fulfilling the given function, are to be included. Wrisberg et al. (2002) state that without specifying both spatial and temporal dimensions, when information is integrated, it leads to results that are space and time independent. As a consequence, for example, depreciation will normally not be used since all costs and values are related to present price or value.

The term Extended Supply Chain (ESC) was introduced earlier and there are several reasons why ESC is used to describe the product systems. First, it is useful to have a term that describes the system and not the management activities within the system, such as Life Cycle Management, Supply Chain Management, etc. Second, the alternative terms value chain and supply chain are used in several contexts and might therefore be misinterpreted. Porter (1985) originally used the term value chain in an intra-organizational context which is not appropriate for product systems. The term supply chain is more focused on inter-organizational aspects, but in many definitions it is not obvious that the term includes use and end-of-life treatment. On the contrary, this is often omitted in the interpretation of a supply chain (Christopher 1998; Pagell and Krause 2002).

A product based system can be viewed from at least two angles; with a main focus on the actors in the production chain, or a main focus on the product itself. This is illustrated in Figure 9 and Figure 10, respectively. In Figure 9 the main focus is on the companies involved. The companies are thus identified as subordinate to the product system, or as subsystems in an SE context and the processes within each company are the system components. As illustrated in the figure, the supply chain and the extended supply chain are easily identified.

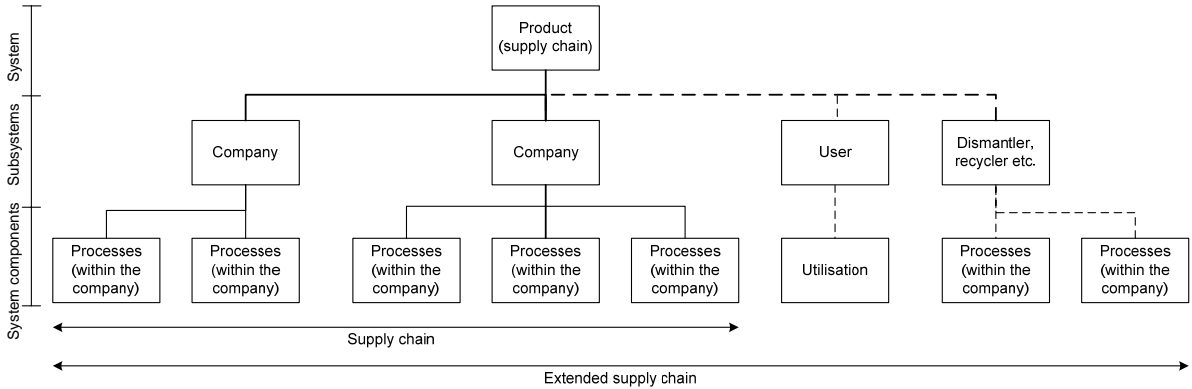


Figure 9 – Actor focused product based system (from Michelsen 2003)

The illustration in Figure 10 places more emphasis on the product itself. The subsystems here are the different components the product is composed of and the materials are the system components. Here, there is less focus on the processes necessary to bring the product into life. All these processes, including product design, must be included hierarchically in such a way that the processes necessary for producing material *n* are included in this system element. All processes necessary for producing component *n* out of the different materials, are included in this subsystem and so on (Michelsen et al. 2006a). Inclusion of these processes is indicated in the figure by the dotted life cycle circles at each system level. These life cycles do not necessarily have the same length; for example, maintenance and use of spare parts could result in different lengths of the life cycles within a system (Michelsen et al. 2006).

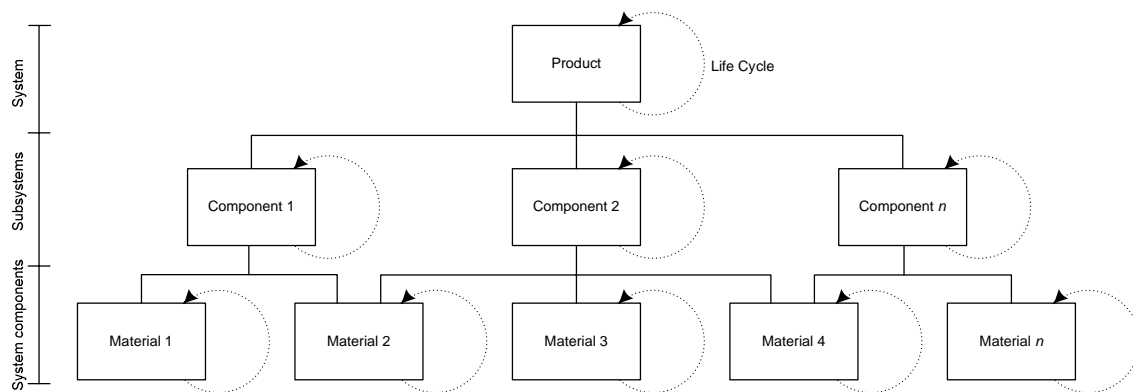


Figure 10 – Product focused product based system (from Michelsen 2003; Michelsen et al. 2006)

Figures 9 and 10 illustrate the convenience of hierarchies for decomposing large, complicated systems into parts that can be readily studied and for defining the boundaries of a system-of-interest

3.1.2 The systems engineering process

The SE process basically consists of six different steps (Blanchard 1991; Fet 1997; see Figure 11).

Step 1 – Identify needs

The starting point in a systems engineering process is the identification of a need or a desire for one or more items. This need can be based on a real or perceived deficiency (Blanchard 1991). One example of a need is upgrading a system not performing in accordance with the original requirements. In the context of eco-efficiency such needs might be that the costs of production are too high or that the environmental impact is unacceptable (see section 2.5). Blanchard (1991) also focuses on perceived deficits, such as the feeling of low performance compared to competitors. Even the lack of knowledge on eco-efficiency performance might be enough to initiate the process.

Fet (1997) divides this step into three basic questions:

- What is needed?
- Why is it needed?
- How may the need be satisfied?

In each of the case studies, the needs were given by the companies. These are presented in chapter 4. The identified needs have provided important constraints for methodological choices done in the case studies. These are discussed in both in section 3.2 and 5.1.

Step 2 – Define requirements

The requirements are the answers to the questions in the first step. Firstly, there are functional requirements that according to Fet (1997) answer the question 'what is needed?' Secondly, there are operational requirements. They reflect the needs of the customer and should answer the question 'why is the system needed?' Lastly, there are physical requirements, which reflect the needs for physical connections between subsystems and elements, the physical conditions the system will be exposed to and how the system fits into the environment (Fet 1997). These requirements are therefore in most cases related to a specific physical location, environment and application.

No formal requirements were made in the case studies. These issues were discussed continuously with the companies during the work.

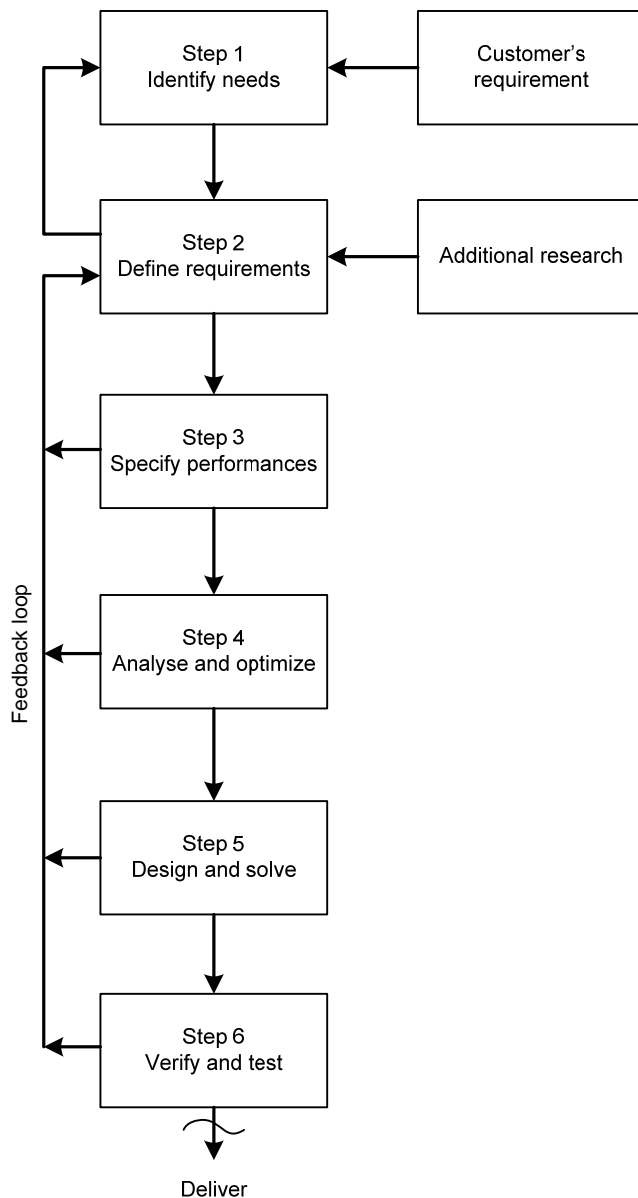


Figure 11 – The systems engineering process (from Fet 1997, modified from Blanchard 1991)

Step 3 – Specify performances

In the next step the defined system requirements should be translated into performance specifications. These are definable and measurable performance criteria for the system as a whole and allocated onto the subsystems and system elements necessary to meet the specifications set for the system.

The first three steps in the system engineering are closely linked where the outcome of the third step is the final quantification of the needs identified in the first step and defined in the second. As suggested by the feedback loop, any of these steps may be iterated as necessary to provide clarifications throughout the entire process.

In the cases used in this study, this step was also treated rather informally. The major challenge for the companies at present is to be able to assess and present the performance of the products in the first place. The focus has been primarily on identification of possible ways of assessing and presenting performance at the product level, rather than meeting specific performance targets. Both governmental regulations and recommendations from business organisations are taken into consideration here. An

important issue is to what degree the performance information should be aggregated. Different ways of presenting information on performance will be discussed in section 5.1.

Step 4 – Analyse and optimize

An essential part of the systems engineering process is a continuous analytical effort. This includes activities for evaluating different system design alternatives by carrying out trade-offs between different, and often conflicting system requirements. This is a central step in eco-efficiency analyses. Lafferty and Hovden (2002) state that there is often a real conflict between environmental and financial concerns and it is crucial that analyses on eco-efficiency clarify if this is the case. In the real world, it is not always possible to identify win-win situations (see Figure 6) and it might be necessary to prioritize between improvements in environmental and economic performance. This can for instance be done by establishing absolute performance requirements for one system element and then optimising other elements with respect to these requirements.

Sometimes it is also necessary to do a trade-off between different environmental aspects. As an example, if a component has a major impact on emission of greenhouse gasses while an alternative component has a major impact on land use, it is necessary to be able to determine which one of these is to be preferred. This could in particular be important when renewable materials are compared to non-renewable materials. This will be further discussed in section 5.1.

A challenge is to select the best possible approach using various analytical tools. Within the concept of eco-efficiency different analytical tools are advocated and some of the most relevant are presented and discussed in section 3.2.

Step 5 – Design and solve

In this step alternative improvements can be introduced. This could include improvements at all system levels – redesign and improvements of the entire system, the subsystems, the system elements or even processes and activities within the system elements. This step is beyond the scope of the case studies. Here, the focus has been on assessing the present performance and also analysing possible alterations from the present situation, but the implementation of new designs has not been performed as part of the case studies.

Step 6 – Verify and test

Before delivering a system it must be tested to show that the initial needs and requirements are met. If the initial requirement was to improve the performance of the system, e.g. the environmental performance, it must be verified that the improvements are in accordance with the initial requirements. The manner in which tests are conducted is an essential part of the whole process. As mentioned above, redesign has been beyond the scope of the case studies and the outcome has been an increased knowledge on the actual performance of the present systems and identification of possibilities to improve these.

3.2 Analytical tools

Wrisberg et al. (2002) have given a comprehensive overview over different analytical tools used in the supply of environmental information. These are presented in Figure 12 where they are seen in relation to different concepts and technical elements necessary for the different tools. As shown in the figure, Wrisberg et al. (2002) make no distinction between analytical tools used in e.g. industrial ecology and eco-efficiency. A modified systems engineering process can also be recognized in the decision process. Procedural tools are also important in the implementation of a better environmental practice. The use and implementation of these have primarily been outside the scope of this work, but some relevant concepts will briefly be described in section 3.3 and further discussed in chapter 5.

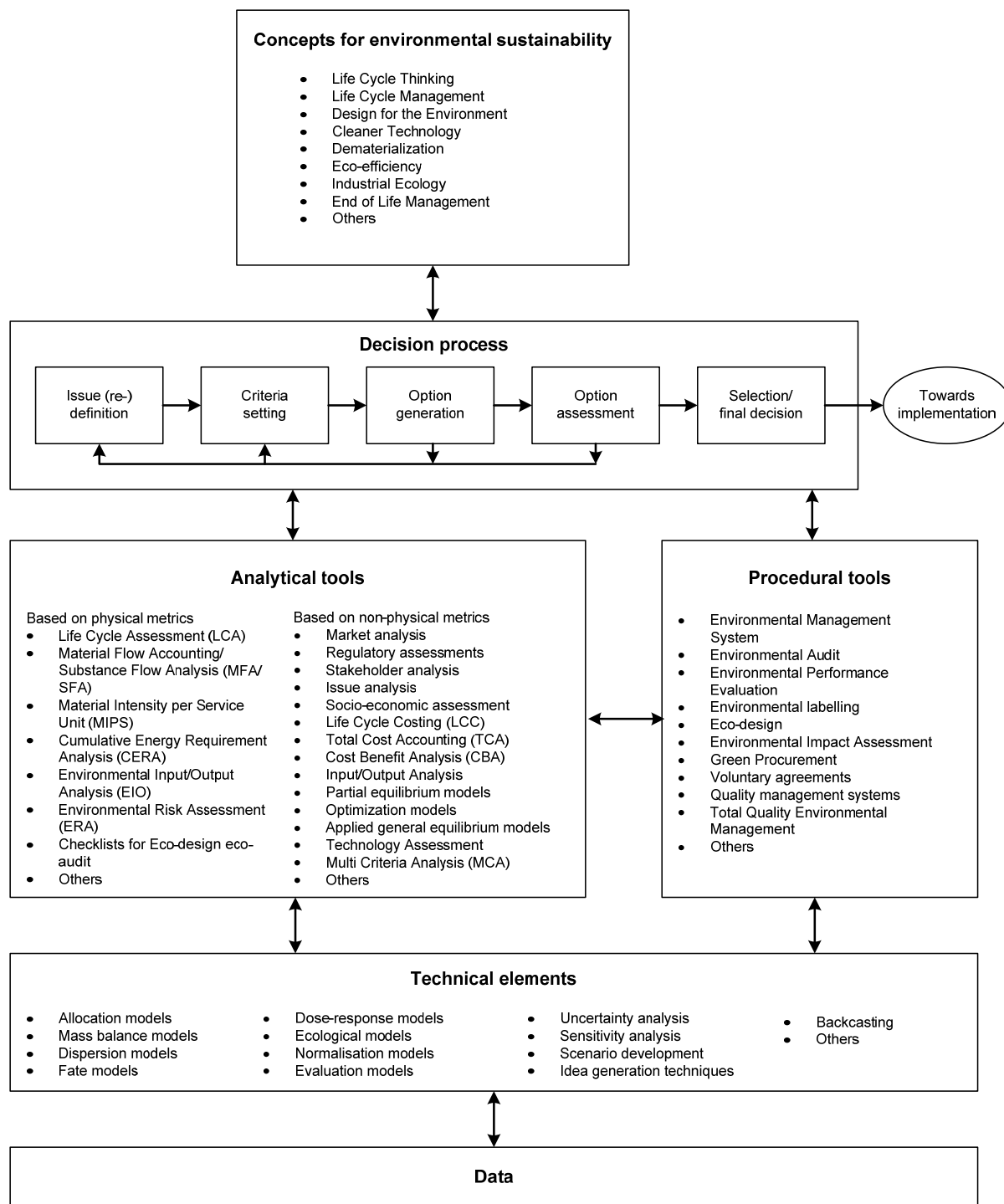


Figure 12 – Framework for supply of environmental information (from Wrisberg et al. 2002)

3.2.1 Assessment of environmental performance

As Figure 12 shows, there are many analytical tools for collecting and reporting environmental information. The problem is rather to identify the most adequate tools for analysing and optimizing according to the initial requirements. Fet (2002) has classified the different tools and methods with respect to the scope of environmental concern and time span (Figure 13).

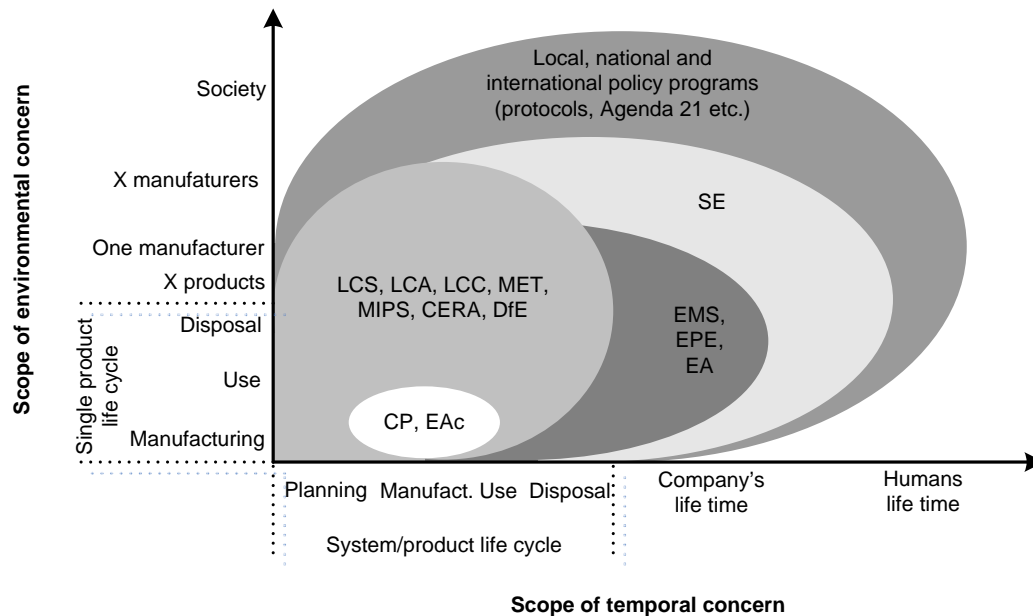


Figure 13 – A classification of methods and tools for environmental performance assessment and improvements (adapted from Fet 2002, see Appendix A for abbreviations)

As pointed out in section 1.2.1, the main focus here is on the product level and their supply chains. Several manufacturers and other actors are involved during the life cycle of a product and regarding Figure 13 a number of analytical tools stand out as the most adequate; LCS, LCA, LCC, MET, MIPS, CERA and DfE all focus on the life cycle of products and the environmental impact generated by more than one manufacturer. These will be discussed below.

The main focus here will be on Life Cycle Assessment (LCA). In addition Life Cycle Costing (LCC) is discussed in the next section. The analyses carried out in this thesis (Michelsen 2006abc; Michelsen et al. 2006ab) primarily use these two tools. Life Cycle Screening (LCS) is a simplified LCA with lower accuracy. Generic data might be used to get a first overview of a situation as is demonstrated in Michelsen (2006b).

Material Intensity per Service Unit (MIPS), Material cycle, Energy use and Toxic emissions (MET) and Cumulative Energy Requirements Analysis (CERA) can all be regarded as varieties of LCA, but with slightly different focus. MIPS focuses on the material flows as such without making any effort to prioritize between them (Schmidt-Bleek 1994) and differs in this respect from LCA (see below). MET has approximately the same focus with input-output based assessment, but here also energy is included and toxic emissions are highlighted (Brezet and van Hemel 1997). CERA only assesses the energy requirements over the life cycle of the product (Verein Deutscher Ingenieure 1997; Wrisberg et al. 2002). A major drawback of these methods is their inability to differentiate between significantly different impacts. In MIPS, for instance, an emission of 1 kg sand is not mapped differently from 1 kg mercury (cf. Reijnders 1998). These methods were thus found inappropriate for use with the case studies.

The last tool mentioned by Fet (2002) with a product life-cycle focus, is Design for the Environment (DfE). Fiksel (1996) defines this as a '*systematic consideration of design performance with respect to environmental, health, and safety objectives over the full product and process life cycle*'. It is thus not an analytical tool as such, since analytical tools are required in order to fulfil these requirements. Wrisberg et al. (2002) have categorized DfE as concept, and it will not be discussed any further in this section.

In addition, all sorts of checklists can be used (Brezet and van Hemel 1997; Wrisberg et al. 2002). As an example, the WBCSD (see DeSimone and Popoff 1997) has seven guiding principles for how to increase eco-efficiency:

1. Minimize the material intensity of goods and services.
2. Minimize the energy intensity of goods and services.
3. Minimize toxic dispersion.
4. Enhance material recyclability.
5. Maximize the use of renewable resources.
6. Extend product durability.
7. Increase the service intensity of goods and services.

However, it is important to have in mind that the differentiation of the different tools to some degree is artificial from the practitioners' point of view. They often pick bits and pieces depending on their actual need there and then (Fet 2002).

Life Cycle Assessment

Life Cycle Assessment (LCA) is defined as '*compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle*' (ISO 14040: 2006). Compared to several other analytical tools (above), the scope of an LCA is to assess the potential environmental impacts of the inputs and outputs of a system, not only the flows as such. It is a comprehensive tool with a cradle-to-grave focus and the idea is that LCA applied to a product should make it possible to assess the overall environmental burdens, identify the 'hot spots' of the life cycle and predict the effects of any proposed improvement actions.

A LCA consist of four distinct phases; definition of goal and scope, inventory analysis, impact assessment, and interpretation (Figure 14).

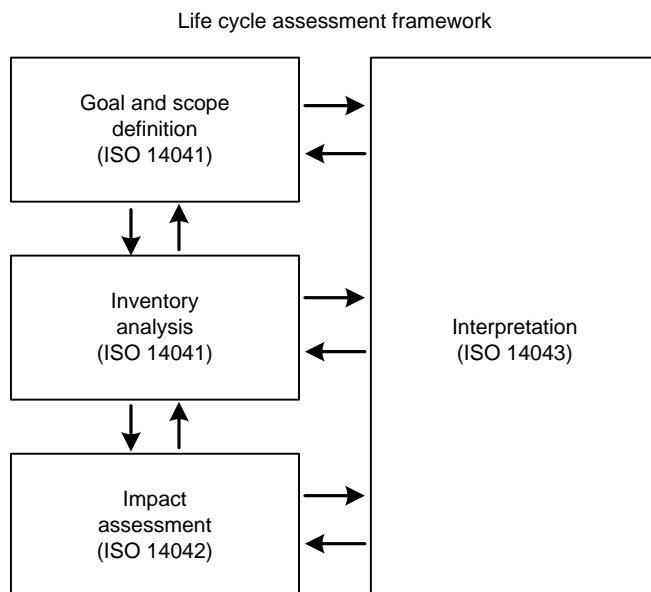


Figure 14 – Phases of an LCA (following ISO 14040: 2006)

Definition of goal and scope

The first step of an LCA is to define the scope of the analysis. A central element here is to define the functional unit; i.e. what is to be analysed. This might be a product, e.g. a car with certain properties, but often it is more appropriate to define a function, e.g. the ability to transport someone or something from one point to another. This makes it possible to compare different ways of fulfilling a function, which again can motivate system innovations, not only incremental improvements through redesign (see Figure 8).

Other issues in this phase are defining system boundaries, data quality requirement, assumptions and allocation procedures (ISO 14040: 2006).

It is also important to decide the level of data accuracy and what environmental impacts are to be included. A decision must be taken to what degree generic data for materials and processes might be used, and to what degree case specific data is needed. Case specific data increases the accuracy of the study, but at the same time increases the level of effort.

Inventory analysis

The second phase of an LCA is the life cycle inventory analysis (LCI). Here the inputs and outputs of a given system are quantified based on the defined system boundaries. In this phase no judgement of the relative significance is done; the objective is to provide quantitative data of the flows. Guidelines for how this should be done are provided in ISO 14041: 1998.

Impact assessment

To determine environmental impacts, the flows of the system are assessed with regards to their potential environmental impacts. This phase of LCA is referred to as Life Cycle Impact Assessment (LCIA). Here, different flows affecting the same environmental problem, e.g. acidification or climate change, are classified into environmental impact categories (ISO 14042: 2000). The contribution of the different flows to the same impact categories can then be characterized according to their magnitude and merged to one unit, e.g. the global warming potential of different greenhouse gasses is assessed based on the relative strength compared to CO₂. The characterization is based on scientific criteria, e.g. the IPCC model for greenhouse gasses (ISO 14042: 2000).

Interpretation

The LCIA results from the previous phase are used to identify hot-spots and areas for improvements. A controversial issue is to what degree environmental impact categories are to be further aggregated. Environmental impact categories are often referred to as midpoint categories and these can be further aggregated to endpoint indicators that are related to safeguards subjects or areas of protection (see Figure 15, Udo de Haes et al. 1999; Jolliet et al. 2004). These areas of protection are normally identified as human health, natural environment, man-made environment and natural resources. The advantage with this aggregation towards areas of protection is an increase in environmental relevance, but at the same time this increases the uncertainties (Udo de Haes et al. 1999; Jolliet et al. 2004). The areas of protection can be further aggregated into a single score. This is for instance done in the LCIA methods Eco-indicator 99 (Goedkoop and Spriensma 2001) and Environmental Priority Strategies (EPS, Steen 1999). In ISO 14040: 2006 aggregation is not supported due to lack of scientific basis, but neither is it completely rejected as an option (ISO 14044: 2006).

LCA is a comprehensive tool and the data requirements are extensive. In practical applications, all requirements can not be met fully (Graedel 1998; Wrisberg et al. 2002) and trade-offs must be done. In addition, not all possible environmental impacts are included. Impact on biodiversity is in general included, e.g. through the assessments of acidification and ecotoxicity, but the most severe impact on biodiversity is caused by changes in land use (Müller-Wenk 1998; Chapin et al. 2000; Sala et al. 2000) which normally is not included due to lack of appropriate indicators (Michelsen 2004; Milà i Canals et al. 2006). This is an important shortcoming and will be discussed in section 5.1.2 (see also Michelsen 2006c).

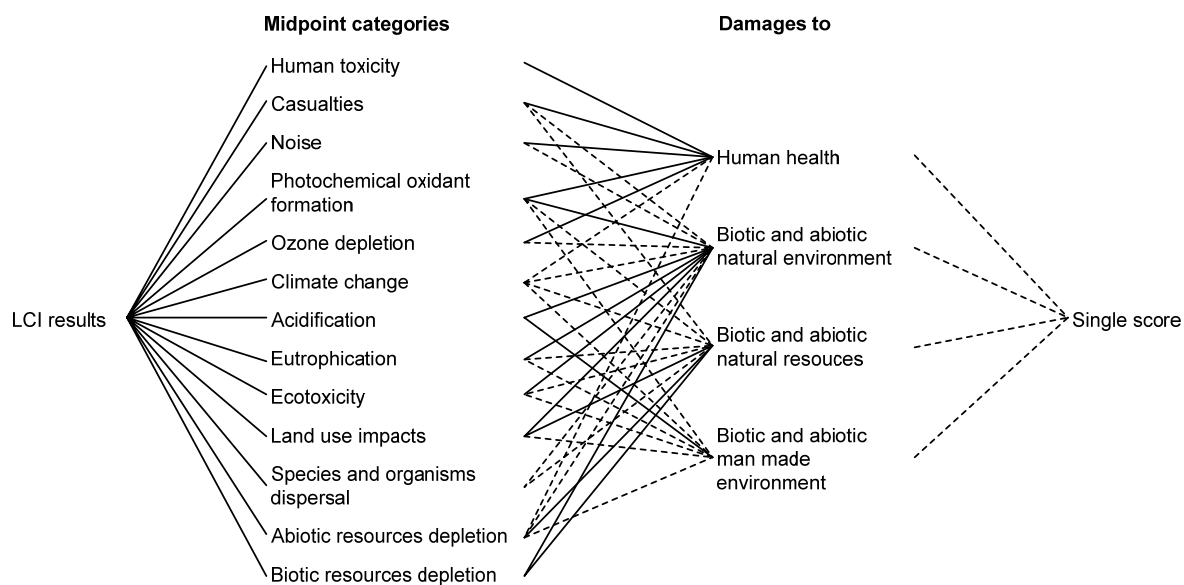


Figure 15 – General structure of LCIA. Solid arrows indicate that a quantitative model is available, dashed arrows indicate that only uncertain or qualitative relationships are known (adapted from Jolliet et al. 2004)

Methodological discussions on LCA

In addition to the debate on weighting and aggregation of LCIA results, which is mentioned within the ISO-standards, there are several other methodological choices that are not addressed in the standards but still are of major importance for the outcome of the assessments. Here, four such topics will be addressed;

1. Retrospective versus prospective LCA
2. Use of average versus marginal data
3. Use of input-output data in addition or as an alternative to process oriented data
4. A possible site dependency of data

Traditionally, the purpose with LCA has been to describe the impact of a system (cf. ISO 14040: 2006). This is often called retrospective, descriptive or attributional LCA (Milà i Canals et al. 2006). In a retrospective LCA, the goal is to determine all impacts caused by the studied system and very often averaged data, e.g. on the electricity mix, is used (Tillman 2000). The point of reference is simply zero; there is no system and the impact is compared to this.

On the other hand, prospective or consequential LCA has an already existing system as a point of reference (Milà i Canals et al. 2006). The aim is to describe how environmentally relevant flows will change in response to possible decisions and marginal data are often used (Tillman 2000). Tillman (2000) pinpoints that the different types of LCA have different applications. Retrospective LCA is most appropriate for identification of environmental impact related to market claims, to identify possibilities for improvements and mere learning about the performance of the system. Prospective LCA is more appropriate when it comes to changes in design of products and processes and how to respond to regulatory measures aiming for change.

However, the debate on average versus marginal data does not stop here. If, for instance, a company guarantees that they only use electricity from renewable sources, what about other actors in the same country or region? If one or more companies are using only renewable energy, the others must necessarily use less renewable energy than average. Further, if a company starts a new process that uses electricity and thus increases the demand for electricity, more of the marginal energy source will be

produced. This will in many cases not be renewable energy, but still, if a retrospective analysis is conducted, the average electricity mix will most likely be used. An example of this problem is demonstrated in Michelsen et al. (2006b). If there is an increased demand for timber, it is likely that the new demand will be met by cutting in less accessible areas. These logs would thus have an environmental performance approaching the worst case assessed which has a significantly higher environmental impact than the average.

In LCA the focus has traditionally been on the different processes within the system where the inputs and outputs from the different processes have been identified and assessed (see Figure 16). The problem with this approach is that in practice it is impossible to include all processes contributing to a product system and some sort of cut-off criteria must be set. The reason for cut-offs is to leave out insignificant inputs and outputs. In Figure 16 this is exemplified with the system boundaries and processes left outside. There are rules on how cut-offs are to be selected (ISO 14040: 2006; ISO 14041: 1998), but these still can not be set on a scientific basis (Suh et al. 2004). A major problem is that the contribution of a process is not known before it is assessed, and when it is assessed, it could as well be included. The result is large difficulties in setting system boundaries for two different systems in an equivalent way. This hampers the comparison of different systems. In addition, flows that do not contribute to the energy and material content of the final product are in general not included, even though they might be essential for the product life cycle (Suh et al. 2004). Inputs from the service sector, e.g. advertising, and impact from production facilities could be examples.

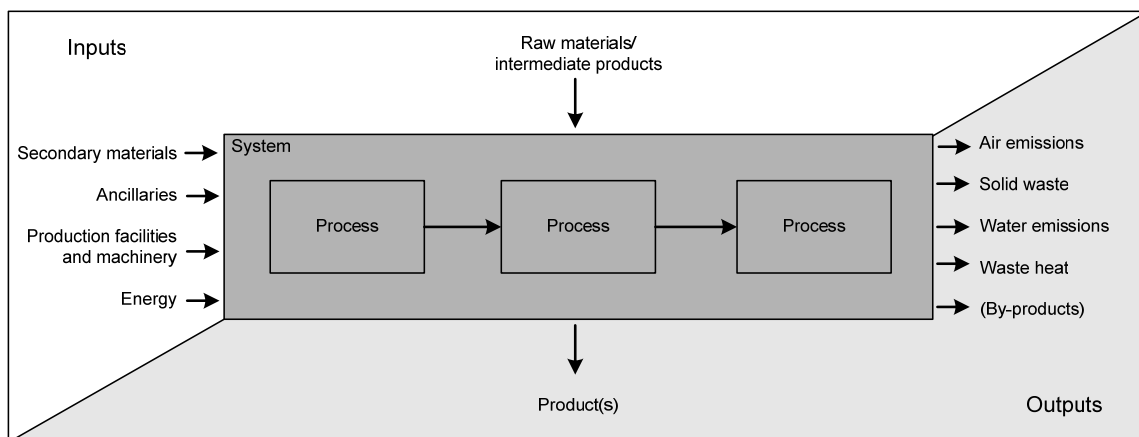


Figure 16 – System borders and inventory of material and energy flows related to the system

As a response to this, input-output (IO) analyses are introduced. As distinct from process focused LCA that has a bottom-up approach, input-output analyses have basically a top-down approach. The starting point is monetary flows and sector specific environmental impact information. This is known as environmental input-output data and is most often revealed from national statistics (see Suh et al. 2004). IO data are also encumbered with shortcoming; they are very generic and in most cases do not distinguish between different technologies, they are often some years old and hence have not incorporated technological improvements, and they are sensitive to fluctuations in prices (Suh et al. 2004).

A solution to address these shortcomings has been to develop hybrid models, incorporating the best both from process oriented LCA and IO oriented LCA where the processes in a foreground system are assessed in detail, while environmental IO data is used for a background system. It is believed that this gives a more correct picture of the actual impact from a system, and in most cases such hybrid LCAs give significantly higher assessed impact (Suh et al. 2004). An example on this approach is shown in Michelsen et al. (2006b). Following the schematic outline in Figure 16, the consequences for the assessments are that some of the inputs that are found outside of the system in

the process-oriented approach are moved into the system through IO data. In contradiction to pure IO analyses, IO-data here are used complementary to process data and follow the bottom-up approach traditionally found in LCA.

LCA is a function oriented system, and as stated in section 3.1.1, these systems are often regarded as site-independent (Wrisberg et al. 2002). However, during the last years there has been a tendency toward a higher degree of site-dependency in LCA (Finnveden and Nilsson 2005). The reason is that the impact of an emission is not independent of where the emission takes place. Emissions of NO_x will for instance have higher impact on the environment if it is released over land than over the ocean far from the shore, and the impact on human health will be higher if the emission takes place in dense populated areas than areas with lower population (Fet et al. 2000; Finnveden and Nilsson 2005). Emissions of greenhouse gasses and ozone depleting substances are in fact the only impact categories shown in Figure 15 that are completely unaffected by where the impacts take place and represent exceptions from what is generally found.

This topic is of increasing importance when land use is considered (Michelsen 2006c; Milà i Canals et al. 2006). It is however important to stress that site dependency is not equivalent to site specific; what is relevant is the characteristics of the area, not the exact geographical location (Finnveden and Nilsson 2005; Milà i Canals et al. 2006).

3.2.2 Assessment of value performance

Value performance is a far vaguer concept than environmental performance (e.g. Cox et al. 2001). The term 'value' has a variety of meanings, but in economics value designates the worth that a person attaches to a good or a service (Fabrycky and Blanchard 1991). The value of a product is thus in principle different from the price of the product. This is clearly demonstrated through processes that add costs (and thus in most cases also increases the selling price) to a product without increasing its value, e.g. most cases of storage (Christopher 1998). Processes that add costs might even decrease the value, e.g. if a new computer or mobile phone is stored and thus get outdated before it is passed to the ultimate consumer. For this reason, Christopher (1998) states that a product does not have any value at all before it reaches the customer with the requested quality and within the requested time limit.

When comparable products or functions are considered, there is a relationship between the value of the product and the costs related to the product, following the equation

$$\text{value} = \frac{\text{function}}{\text{cost}} \quad (4)$$

(Monczka et al. 2005). It is thus possible to use cost and added cost as a proxy measure for value as long as the processes or products compared have the same function. This is of course not optimal, but it makes it possible to assess a context dependent concept such as value (cf. Fabrycky and Blanchard 1991; Christopher 1998).

Life cycle costs

In the same manner as for environmental performance, value performance measured as costs can be assessed over the life cycle of a product. Life Cycle Cost (LCC) is defined as '*all costs associated with the system as applied to the defined life cycle*' (Blanchard and Fabrycky 1998). LCC analysis (or life cycle costing) has the same system focus as life cycle assessment (see Figure 13) and is often recognized as the economic counterpart of LCA (e.g. Klöpffer 2003; Rebitzer and Hunkeler 2003).

LCC is primarily developed for guidance in procurement purposes (Asiedu and Gu 1998) with the purpose to reduce the life cycle costs of a product or service system (Fabrycky and Blanchard 1991; Durairaj et al. 2002). Most LCC analyses are performed from the

customers' point of view to demonstrate the life cycle costs of an option, often in comparison to other options (Saling et al. 2002; Hunkeler et al. 2004; Schmidt et al. 2004). However, the difference in information needs between customers and top managers in manufacturing companies are in most cases minor (Kleijn et al. 2002) and in function-oriented business systems where the customer buys a function and not a product, there is no difference in the interests at all (Tukker 2004).

LCC is thus not only useful for the customers but for the producers as well. It is still important to be aware of the possible conflicting targets of different stakeholders. As shown in equation 4 there is a relationship between value and cost, but if the cost of a product increases due to an increased selling price, this might be beneficial for the producer in means of higher margins, but will at the same time cause higher costs for the customer and thus be disadvantageous.

Even if LCC analyses are commonly used, there is no uniform understanding of the term and no standardized framework commonly used in business (Rebitzer 2002). One of the most important issues is the inclusion or exclusion of externalities. Figure 17 shows schematically the different costs that might be included; costs for the manufacturer, costs for the user and costs for the society. Rebitzer and Hunkeler (2003) separates the total costs for the system into internal costs that concern all the costs and revenues within the economic system (shaded area in the figure) and external costs that are outside the economic system, but still inside the social and environmental system.

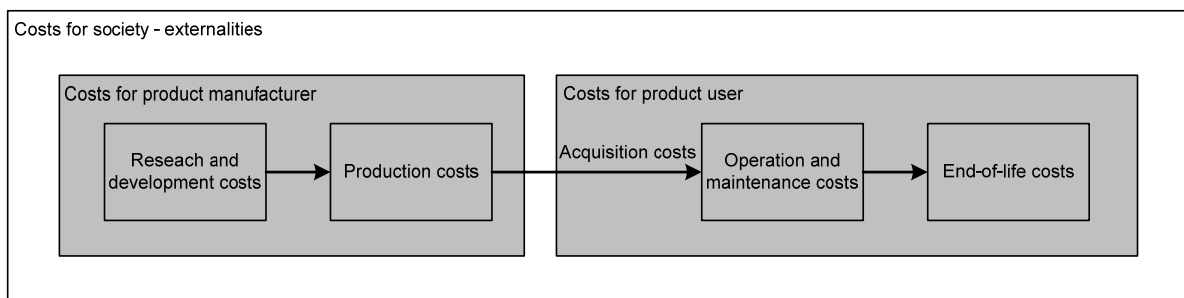


Figure 17 – Total internal (shaded area) and external costs of a product system

There are large differences in the visibility of costs, even internal costs. Shapiro (2001) uses the terms 'potentially hidden costs' and 'less tangible costs' for costs not always easy disclosed. This might be costs due to site preparation and closure, training, liabilities (e.g. pollution penalties), future regulatory compliance costs, organizational image, etc. To include these is a question of the precision of the methods used (Asiedu and Gu 1998), and not as fundamental a question as that of externalities. As earlier pointed out (see section 2.3), more than 70% of all costs of a product system are pre-determined by activities and decisions taken during the design phase.

When externalities are included, LCC analyses are as much a tool for environmental assessment as value assessment (Durairaj et al. 2002; Wrisberg et al. 2002). In this thesis externalities are not included in LCC and the definition provided by the International Electrotechnical Commission (1996) is used as basis. Life cycle costs are defined here as the cumulative costs of a product over its life cycle, given as the sum of acquisition costs, ownership costs (operation and maintenance) and end-of-life treatment costs (cf. Figure 17) following the equation

$$LCC = \text{cost}_{\text{acquisition}} + \text{cost}_{\text{operation and maintenance}} + \text{cost}_{\text{end-of-life treatment}} \quad (5)$$

Externalities are not included since the purpose of eco-efficiency is to relate value performance to environmental performance. If environmental externalities are included in the value performance, this would, when environmental impact is over a certain level,

have made the value performance and environmental performance systematically dependent on each other. The environmental fraction of the value assessment would in such situations become significant and e.g. Figure 6 would thus have become a plot where environmental performance was related to itself. In Michelsen et al. (2006a) externalities were not included and no clear correlation was found between environmental performance and costs. It thus makes sense to compare these two as independent variables such that both must improve to achieve a strong sustainable improvement (cf. Schaltegger and Burritt 2000).

However, the definition of LCC does include environmental taxes since these are internalised costs actually paid by the actors within the system. In many situations it would not be possible to identify the size of these costs. For example, it will in most cases not be possible for a purchaser to know how much of the acquisition costs are a result of environmental taxes on fuel consumption in activities upstream.

This interpretation of LCC is very much in accordance with Rebitzer and Hunkeler (2003) who define LCC *'as an assessment of all costs associated with the life cycle of a product that are directly covered by the any one or more of the actors in the product life cycle (supplier, producer, user/consumer, EOL-actor), with complimentary inclusion of externalities that are anticipated to be internalized in the decision-relevant future'*.

According to Shapiro (2001) there are reasons to believe that more externalities will be internalised as a result of increasing environmental regulations and companies will need to become aware of these. This will of course increase the correlation between environmental impact and costs. However, in eco-efficiency assessments this is already taken care of since the environmental performance is fully integrated. For the time being there is also almost no correlation between the size of an environmental tax and the environmental damage this is supposed repair (Labouze et al. 2003) which indicates that environmental legitimated taxes should be treated just as any other taxes.

It is worth pinpointing that this use of LCC is more or less equivalent with how terms like life cycle price (e.g. Labouze et al. 2003) and total cost of ownership (e.g. Ellram and Siferd 1998) are defined.

Disaggregation of costs

Life cycle costs can be disaggregated and assessed at different system levels in the same manner as environmental impact in LCA (e.g. Michelsen et al. 2006ab). It is thus possible to compare the relationship between added costs and environmental impact in different segments of an extended supply chain. In addition, life cycle costs can be further subdivided into what is recognized as a cost breakdown (see e.g. Fabrycky and Blanchard 1991; International Electrotechnical Commission 1996). Of particular interest is the value added (VA) in the different segments of the extended supply chain. This can be assessed in two different ways (Sturm et al. 2003):

$$\text{Value added} = \text{revenue} - \text{cost of goods and services purchased} \quad (6)$$

$$\text{Value added} = \text{salaries} + \text{depreciation} + \text{amortisation} + \text{interest paid} + \text{taxes} + \text{dividends} + \text{retained profit} \quad (7)$$

An interesting aspect with value added is that a high VA indicates that a high level of skills and expertise is applied to the production (Azapagic and Perdan 2000). VA also represents the contribution of an activity to GDP (Azapagic and Perdan 2000). VA is readily available on the company level since VA is the basis for VAT taxation.

However, it is argued that this should be brought one step further to net value added (NVA) where also the charge for capital invested is subtracted from VA since this comes closer than other measures of financial performance to capture the true economic profit

(Azapagic and Perdan 2000)⁴. In a similar way as VA Sturm et al. (2003) express this as follows:

$$\text{Net value added} = \text{revenue} - \text{cost of goods and services purchased} - \text{depreciation on tangible assets} \quad (8)$$

$$\text{Net value added} = \text{salaries} + \text{amortisation on intangible assets} + \text{interest paid} + \text{taxes} + \text{dividends} + \text{retained profit} \quad (9)$$

As mentioned above, LCC is often performed from the customers' point of view and to some degree this satisfies also the information need for top management in companies. Nevertheless, it is often the profit made that is of real interest for top management. When eco-efficiency is assessed for internal purposes in a company, it is sometimes assessed as the ratio between environmental impact and profit (Ilinitch and Schaltegger 1995; Schaltegger and Sturm 1998; Verfaillie and Bidwell 2000). Profit can then be understood in different ways, in particular as NVA following equation 8 (in Verfaillie and Bidwell 2000), as retained profit only or as the sum of retained profit and dividends (cf. equation 7). It is not always made clear how profit actually is understood when this is presented (cf. Ilinitch and Schaltegger 1995; Schaltegger and Sturm 1998).

Moreover, assessing value added and profit in a supply chain is not an easy task since the necessary information is not readily available (e.g. Cox et al. 2001). VA on company level is available through financial statements, but VA and profitability for different processes and products are primarily regarded as confidential. It is not possible to estimate this for a process if the company is not willing to share the information. When value performance is assessed in a supply chain, it is thus sometimes necessary to use added costs as a proxy for value added since this is the only available information (see Michelsen et al. 2006a). Table 2 summarizes advantages and disadvantages of using costs and life cycle costs as measures on value performance in an extended supply chain.

Table 2 – Advantages and disadvantages with life cycle costs as a measure for value performance in an extended supply chain

Advantages	Disadvantages
- measurable, data available	- hidden costs difficult to identify
- related to value	- not equal to value
- can be broken down in cost elements	- no standardized assessment methodology
- can be measured in different segments and aggregated over the product life cycle	- stakeholders might have conflicting interests
- equivalent system borders as in LCA	
- externalities can be included if desired	

Alternatives to monetary units in value assessments

There are of course alternatives to monetary units when value is to be assessed. The most often used alternative is to measure the quantities of goods and services produced or provided to customers (e.g. Verfaillie and Bidwell 2000).

Produced volume is in many situations a useful measure on value performance, and for companies performing eco-efficiency assessment at the company or factory level, this is a useful indicator (Verfaillie and Bidwell 2000; see Table 1 and Figure 7). Companies could then assess the performance as produced volume related to emission factors. However, at other system levels this is a less suitable measure of value performance. It is usually not possible to compare activities in different subsystems of a system since different components are produced (e.g. different subsystems in Figure 9 and Figure 10), and it is not possible to disaggregate the measure in the same manner as can be done

⁴ Azapagic and Perdan (2000) use the term 'Economic value added'

with monetary indicators (cf. equations 5-9). It is also worth noticing that neither produced volume or any other physical measure is mentioned as an alternative to economic performance indicators for value assessments in the sustainability reporting guidelines provided by the Global Reporting Initiative (2002).

3.3 Procedural tools

Procedural tools are essential to implement management systems and establish procedures for continuous assessments of environmental and value performance as described above. The use and development of procedural tools have been outside the scope of this work. However, two governmentally introduced concepts are central for the life cycle of the products and thus the extended supply chains; Integrated Product Policy (IPP) and Extended Producer Responsibility (EPR).

IPP has been supported by the EU primarily (The European Commission 2001, 2003; von Geibler et al. 2004). According to The European Commission (2003), IPP aims at reducing resource use and the environmental impact of products. IPP is not a specific tool but a concept incorporating a whole variety of tools that can be used to achieve the objective. These include measures such as economic instruments, substance bans, voluntary agreements, environmental labelling and product design guidelines⁵.

EPR can be defined as *'a policy instrument to promote total life cycle environmental improvements of product systems by extending the responsibilities of the manufacturer of the product to various parts of the entire life cycle of the product, and especially to the take-back, recycling and final disposal of the product'* (Lindhqvist 2000). Thus, within EPR the responsibility is placed solely on the producers, regardless of who is actually causing the environmental impact. The reasoning is based on the assumption that the producers have the best resources to reduce the overall environmental impact from the products throughout their life cycle (Røine 2006). A major point with EPR is to transfer the responsibility at the post consumer stage of the products away from municipalities and tax payers and onto the producers (OECD 2001).

Røine (2006) suggests that IPP and ERP are related concepts where IPP as a policy strategy is the bridge between EPR as a policy principle and the EPR policy instruments, such as material bans, take-back responsibilities, covenants and deposit-refund schemes.

⁵ <http://ec.europa.eu/environment/ipp/integratedpp.htm>

4 Introduction to case studies

Small and medium sized enterprises (SMEs) are often neglected when it comes to studies on how and to what extent industry impact on the environment (von Geibler et al. 2004) even if the overall contribution from SMEs is considerable (Ammenberg and Hjelm 2003). As pointed out in section 2.2 they often lack resources to conduct measurements and knowledge about the environmental consequences of their actions and how to assess their environmental impact. In addition, they tend to look upon their own contribution as negligible, even without quantifying their impacts (Ammenberg and Hjelm 2003).

In the case studies performed for this thesis most of the companies involved are SMEs. Since such companies in general lack the necessary resources for prioritizing environmental assessments and improvements, a side benefit of the research was that these companies actually had access to such resources for a period. During the research programme they had enhanced possibilities to initiate and consolidate their own work on environmental issues (Fet et al. 2006a).

4.1 The furniture case

The furniture industry in Norway is dominated by small and medium sized manufacturers. The manufacturers are dispersed throughout the country, but there is a higher concentration in the western regions. Several suppliers are located in the same area, and there are several long time relationships between the furniture manufacturers and their suppliers.

The environmental focused research within furniture industry has been quite extensive in the last decade. In the beginning, the focus was on environmental performance indicators and environmental reporting within the companies and also for the municipality (Stordal commune) where the involved manufacturers were situated (Fet and Johansen 2000). In 2000 four furniture producers were involved in the research projects; Stordal Møbler AS, Inform|Pedro AS, Helland Møbler AS and Modi Skandinavia AS.

The focus gradually expanded, and in the research presented here the focus has primarily been on the extended supply chains for selected products (Michelsen 2006ab; Michelsen et al. 2006a). As a part of this project, several LCAs were performed (see papers for details) and the results are used to improve the products that were analysed (Fet 2002). As pointed out earlier, an important issue has been to enable the comparison of different products and different options for producing a product.

The study has been conducted in cooperation with different manufacturers and can thus be seen as several interrelated case studies. However, the work has not been without complications, in particular due to several changes of ownership and reorganizing in some of the involved companies. The main products for the analyses in Michelsen (2006a) and Michelsen et al. (2006a) were in the beginning produced by Inform|Pedro AS (Fet and Johansen 2000), but were overtaken by Hov+Dokka AS. The production was then transferred to new facilities situated in the central parts of Norway (Dokka), but the research activities could still be continued. However, a few years later, Hov+Dokka AS was again reorganized and it was no longer possible to continue the research on these products. The last paper on the furniture industry (Michelsen 2006b) focused on a product from Helland Møbler AS. Helland Møbler AS has been actively involved over the whole period.

The furniture cases can be regarded as suitable for studies on eco-efficiency in extended supply chains. The furniture manufacturers are exposed to an increasing demand for environmental information on the products, in particular from public purchasers (see in particular Michelsen 2006b), and they have in general been highly motivated for

cooperating with the research activities and providing available data. The prospects of an improved image for marketing purposes have motivated the manufacturers (Fet 2002). Their needs and requirements (cf. the two first steps in the SE-process, section 3.1.2) are thus easy to identify; they need to provide information on the environmental performance of their products and be able to quantify improvements. The challenge for these companies has been lack of access to adequate tools and knowledge to perform reliable environmental assessments and an important outcome for these companies is the development of a range of environmental product declarations (EPDs)⁶.

The cooperation between NTNU and the furniture industry has also been expanded since the case studies performed for this thesis. A new environmental life cycle inventory database for furniture production is being created (Fet and Skaar 2006; Fet et al. 2006b) and the first version of Product Category Rules (PCR) for furniture production is proposed (Fet et al. 2006b). Data from the database is used in Michelsen (2006b), and the PCRs will ease comparison between different products. This will be discussed further in section 5.2.

The case studies on furniture production have resulted in three papers included in the thesis:

- Eco-efficiency in extended supply chains: A case study of furniture production (Michelsen et al. 2006a – appendix B)
- Eco-efficiency in redesigned extended supply chains; furniture as an example (Michelsen 2006a – appendix C)
- Investigation of relationships in a supply chain in order to improve environmental performance (Michelsen 2006b – appendix D)

4.2 The timber case

In contrast to the furniture case, the timber case is a new research project initiated for this thesis. The case studies on furniture revealed large uncertainties about the environmental impact from wood components, so further investigations into these components was crucial to increase the accuracy of the environmental assessments of furniture production (see Michelsen 2006ab). This case was a prerequisite for a thorough analysis of the performance of the furniture products (cf. step 3 and 4 in the SE-process), but the forestry sector has their own challenges in providing information on their environmental performance.

The forestry sector is important in Norway. In 2004, wood and wood-based products represented 7% of the total export value from land-based activities in Norway, excluding oil and gas exports (Statistics Norway 2005b). The sector is also important for employment, especially in rural areas, and the woodworking industry is present in more than 70 percent of all municipalities in Norway (The Norwegian Ministry of Agriculture 1998). Even if spruce (*Picea abies*) is of minor importance for the furniture industry, the decision was made to focus on spruce logging since it is the most important type of wood for the forestry industry itself, representing 75% of the logged volume (see Michelsen et al. 2006b).

ALLSKOG BA was the primary cooperating company. This was their first involvement in research cooperation on environmental assessment issues, but there is no doubt that they are facing the same external pressure as the furniture industry to provide environmental information about their products. However, the pressure here originates from within the industry and not from public purchasers, as in the furniture case. Norske Skog, their largest customer, has experienced extensive pressure to document that

⁶ EPDs are available on <http://www.epd-norge.no/>

certain criteria for forestry management are fulfilled⁷, and this pressure is passed on to their customers⁸ (see also Sanness 2003). Partly in response to this pressure, a Norwegian set of PEFC⁹-standards (Living Forests 1998) were developed that could be used as environmental targets for forestry operations in an ISO 14001 certification. In a period, Norske Skog paid additional 7 NOK per m³ for timber that was certified according to this system (Sverdrup-Thygeson et al. 2004). ALLSKOG BA (at that time known as Skogeierforeninga Nord) became both ISO 14001 and ISO 9001 certified in December 2000.

There is no doubt that the environmental pressure has increased during the last years, but there are contradictory findings concerning the willingness of the market to pay for improved environmental performance. In a study in the UK, it was revealed that as much as every fifth consumer was willing to pay additional for eco-labelled wood products (Veisten and Solberg 2004). However, in similar studies, Rametsteiner and Simula (2003) and Sanness (2003) found almost no willingness on the part of consumers to pay more. Notwithstanding, Norske Skog assumes that environmental performance will become a competitive factor in the future¹⁰ and the forestry industry will probably continue with the focus on certification of timber to avoid losing market positions (cf. Rametsteiner and Simula 2003).

Even if the single forest owner is responsible for how the forestry is performed and to whom the timber is sold, ALLSKOG BA is to a large degree controlling the activities within the supply chain from planning of forestry activities to delivery of logs to factories (see Michelsen et al. 2006b). They also have interests in some of the saw mills. This makes their study a particular interesting case as a supplement to the furniture case, since a breakdown of cost data is available for a larger segment of the supply chain. It is thus possible to compare results for total costs versus value added.

The case study on logging of timber has resulted in two papers included in the thesis:

- Environmental impact and added value in forestry operations in Norway (Michelsen et al. 2006b – appendix E)
- The importance of land use impact on biodiversity in an assessment of environmental performance of wood products (Michelsen 2006c – appendix F)

⁷ See newspaper articles: Adressavisen September 29 2003: Regnskog i miljøpapir; Adresseavisen September 30 2003: Han får Norske Skog til å skjelve/Før motorsagene startes; Adresseavisen October 10 2003: Stanser kjøp av tømmer fra regnskog

⁸ Adresseavisen October 21 2003: Skogeierforeninga fraråder hogst

⁹ Programme for the Endorsement of Forest Certification, <http://www.pefc.org/>

¹⁰ Statement from Sverre Thoresen, environmental corporate advisor in Norske Skog, in an interview in the magazine 'Norsk Skogbruk' 5/2006

5 Eco-efficiency in extended supply chains; findings and implications

This chapter elaborates and discusses the research questions presented in chapter 1. The research and results from the case studies are primarily presented in the papers (Appendices B-F) but will be discussed in more detail here where all findings are related to each other. The main conclusions and recommendations on methodological development are presented as a short summary in section 5.1.5. The connections and mutual dependency between methodological, regulatory and organizational issues are discussed in section 5.4.

5.1 Methodological questions

The identified needs provide important constraints for methodological choices. In the selected case studies (chapter 4) it has been important to be able to both compare different products and processes and to identify which parts of the extended supply chains are of particular importance for the value and environmental performance of the products.

When different options (e.g. different extended supply chains) are compared there is one overriding rule; the analyses must be done in a consistent way. The system boundaries must be set equivalent for all alternatives and the performance indicators selected must make comparisons possible.

When different methodologies, such as LCA and LCC, are combined, it is often argued that the system boundaries must be set identically (Klöpffer 2003; Rebitzer and Hunkeler 2003; Schmidt 2003; Sturm et al. 2003). The argument has primarily been that in both cases the system boundaries have to include the entire life cycle to better ensure sustainable development. However, a company can very well have a target for eco-efficiency that says that the company should make as much profit as possible while providing products with as small an environmental impact as possible. Their eco-efficiency ratio (cf. equation 2) would then be expressed as

$$\text{eco - efficiency} = \frac{\text{internal profit (for company)}}{\text{environmental impact (over product life cycle)}} \quad (10)$$

The system boundaries for the assessment of value performance and of environmental performance would obviously be different here, but the ratio makes sense given the company's scope. It is thus important to stress that aspects compared to each other must be measured in a consistent way, but the value performance and the environmental performance in an assessment of eco-efficiency are not compared to each other, they are *related* to each other, and therefore need not necessarily be assessed within the same system boundaries.

This is an important point due to the characteristics of LCA and LCC and the possibilities to combine these two in the first place. As discussed in section 3.2.2, value performance will often be measured as life cycle costs. The costs included are all direct costs covered by any actor in the product life cycle. This means that costs due to advertising, R&D, etc. are included. In LCA, on the other hand, environmental impacts caused by activities not directly contributing to the energy and material content of the final product are in general omitted (see section 3.2.1). Thus, if LCA is performed with a 'traditional' process oriented focus, cut-offs will be performed (cf. section 3.2.1) and the outcome is system boundaries that diverge from what is found in LCC. One example mentioned earlier is advertising and R&D. Costs related to these activities will be included in LCC since the manufacturer has to pay for these costs and include them in the price of the product (see Figure 17). However, they do not contribute to the final energy or material content of the

product and are thus in most cases omitted in LCA (Suh et al. 2004). The system boundaries in LCA and LCC for most products will consequently not be identical, but as long as LCC and LCA are performed in a consistent way for all products and processes that are compared, this is not a problem and the system boundaries for LCA and LCC can be defined independently.

It is nevertheless possible to obtain equal system boundaries for LCA and LCC assessments. If LCA is performed with the use of IO-data complementary to included process data for the processes not included (cf. section 3.2.1 and Michelsen et al. 2006b) the system boundaries will be identical to the boundaries in LCC. Alternatively the costs for all processes not included in the LCA can be omitted.

5.1.1 The extended supply chain as system unit

As pointed out in section 2.3, there is a clear tendency towards a life cycle and supply chain focus in environmental management. Still, the dominating use of eco-efficiency has been for single sites measures (e.g. Verfaillie and Bidwell 2000) and guidelines are focusing on the eco-efficiency ratio (equation 2, cf. Sturm et al. 2003). It is thus tempting to claim that measures on eco-efficiency have been somewhat retardant when it comes to expansion of system boundaries.

The necessity of expanding system boundaries is obvious when it comes to products. The consequence of the needs and requirements identified in the cases (chapter 4) is that the performance specifications must include the entire life cycle of the product and the extended supply chain must thus be the system unit in the analyses.

The pressure to provide information on environmental performance is unevenly distributed, and primarily the pressure is directed towards the actor providing the product for the market (e.g. Hall 2000, and Michelsen et al. 2006b for a case specific example). In the furniture case (Michelsen et al. 2006a) it is obvious that a focus on the end producer alone would be highly inadequate. Figure 18 shows the distribution of the relative contribution from suppliers, the end producer and dismantling for the four most important environmental aspects. Of the four aspects included, the end producer has a direct significant impact to only one of them; emission of photochemicals, primarily due to varnishing. For the three others, the emissions originate from the suppliers and/or from the end-of-life treatment. Similar results are shown also in other studies (e.g. Clift and Wright 2000; Labouze et al. 2003) and the key issue is the ability to identify where the largest potential for improvements within the ESC are found.

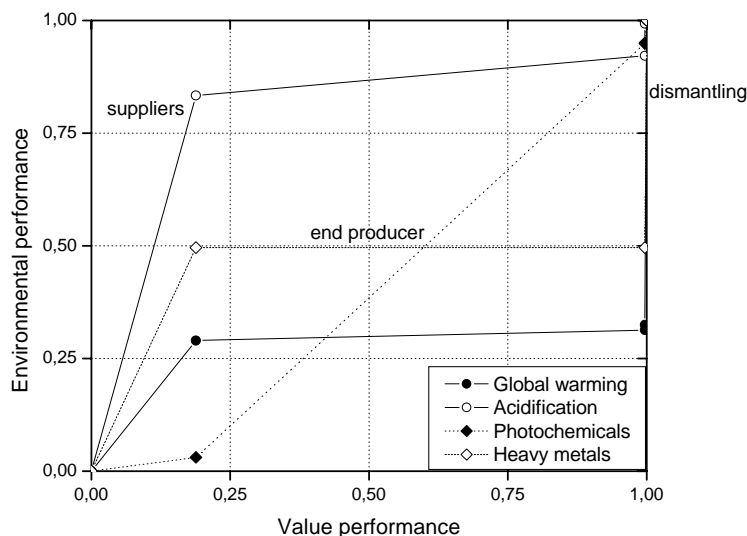


Figure 18 – Relative contribution of value performance and environmental performance from suppliers, end producer and dismantling of a chair (Michelsen et al. 2006a)

Even if the system and the system boundaries are set, a system can be observed from different viewpoints. It is important to construct the system model in accordance with the goal or identified needs (cf. Blanchard 1991), and in section 3.1.1 two different options were described. Despite the low focus on the processes involved, it was decided to view the product system as a compilation of materials and components (Figure 10) instead of a compilation of actors and processes (Figure 9). The advantage of this system perspective is the possibility to use information from different system levels to create a database for materials used in the products. In the furniture case this is already done and has been one important achievement of the research cooperation between NTNU and the furniture industry (Fet and Skaar 2006; Fet et al. 2006b). This is very much in accordance with how LCA-databases are created and is commonly used within industry (e.g. Schmidt et al. 2004). The methodology used in the timber case (Michelsen 2006c; Michelsen et al. 2006b) can be used on other types of wood and contributes to an improvement of these data elements in the furniture database.

It would of course also be possible to use a process oriented system focus (cf. Figure 9) for database creation. However, the next step in the furniture case is to use the database in product development (Fet et al. 2006b). A designer is normally more aware of the materials to be used and the amount, than what processes are necessary to complete the life cycle of the product. A process oriented database would be less useful regarding the analytical needs of the companies.

The conclusion is thus that a database for product assessment and promotion of improvement (cf. Fet et al. 2006b) should be focused on the materials. Improvements can be realized by substitution of materials and reductions in the quantity of harmful materials that can not be omitted entirely. This does however not ensure improvements of the materials as such. Here, the companies involved are the key factors (cf. Michelsen 2006b). This will be discussed in section 5.3.

5.1.2 Assessment of environmental performance

The presently available methods for assessing environmental performance were presented and discussed in section 3.2.1. Three topics will be further discussed here, namely; selection of environmental performance indicators, land use impact on biodiversity, and aggregation of environmental performance information.

Indicators of environmental performance

There is no simple answer to the question of which indicators should be used to assess environmental performance. This is closely related to the intention behind the assessment, which is case specific.

One factor is the amount of information and aggregation of the information. The amount of information is determined, for example, by the number of environmental impact categories included; aggregation is discussed below. Wrisberg et al. (2002) point out that there is in fact a clash of interest between those who ask for environmental information (the demand side) and those who provide environmental information (the supply side). According to Wrisberg et al. (2002) there is a tendency that the demand side request information as simple as possible, while the supply side wants to provide information as detailed as possible. There is thus a challenge to provide information that is reliable and exhaustive and still clear and understandable.

In the furniture case, it was decided to use the recommendations from WBCSD (in Verfaillie and Bidwell 2000) as a starting point (see Michelsen et al. 2006a). This provided an easy recognizable list of indicators as a core and permitted additional aspects that were shown to be important for the particular product to be added. In the furniture case, it was thus decided to use in total 9 indicators for environmental

performance (Table 3). The first seven originate from the WBCSD-recommendations, while the last two are included due to their importance for furniture production.

Table 3 – Suggested environmental performance indicators for furniture production (Michelsen et al. 2006a)

Environmental performance indicator	Unit
Energy consumption	MJ
Materials consumption	kg materials
Ozone depleting substance emissions	kg R11-equiv.
Water consumption	kg water
Greenhouse gas emissions	kg CO ₂ -equiv.
Acidification emissions to air	kg SO ₂ -equiv.
Total waste	kg waste
Emissions of photochemical oxidising substances	kg ethen-equiv.
Emissions of heavy metals	kg Pb-equiv.

This list of indicators is later proposed for use in the PCR for furniture production in Norway, with the modification that water consumption is replaced with eutrophication (Fet et al. 2006b).

In section 3.2.1 the use of IO-data as supplement to process data was discussed. The findings in Michelsen et al. (2006b) confirm findings from others (Suh et al. 2004) that this results in higher assessed environmental impact. As long as different products are compared and as long as the assessments are performed in a consistent way, this is in fact of minor importance since it is often the relative differences of the alternatives that are important. However, if the inclusion of IO-data causes a shift in the relative contribution from the different impact categories, this might be important if the LCI-results are aggregated. The results here are too limited to draw any conclusions, but this is obviously a question that needs to be further addressed.

Also the question about average versus marginal data was discussed in section 3.2.1. In Michelsen et al. (2006b) the relevance of this is demonstrated. If wood consumption in the furniture industry increases and this increases the overall demand for wood, it should be considered if average values are a relevant representation of the environmental impact. If the increased demand leads to logging in less accessible areas, the environmental impact of this timber will probably be closer to the worst case presented in Michelsen et al. (2006b), than the average. The ISO standards give no answer to this question, and no recommendation will be given here. For the time being the conclusion is that this has to be decided within the industry (see section 5.2.2) and the decision must be communicated in a consistent and transparent way.

Land use impact on biodiversity

LCA has mainly been used to study consumption of raw materials and energy, emissions of pollutants and generation of waste, but there is no doubt that other environmental problems also need attention. This is also underlined in ISO 14040: 2006 where it is stated that *all* environmental attributes or aspects must be considered. Land use and land use changes are undoubtedly one example and this is also explicitly mentioned in ISO 14044: 2006. The main reasons given for loss of biodiversity is changes in land use and a consequential unavoidable loss of habitats (Pimm et al. 1995; Müller-Wenk 1998; Chapin et al. 2000; Sala et al. 2000). Loss of biodiversity is one of the largest environmental problems, if not the largest (Diaz and Cabido 2001). Still, there is no agreed upon method for how loss of biodiversity due to land use is to be included in LCA (Milà i Canals et al. 2006).

The land use impact on biodiversity is in particular important when raw materials originate from land extensive activities. Forestry as the origin for wood based products is a striking example. This is thus an important aspect in the selected cases (chapter 4) and

a first proposal of a methodology is presented and applied in the timber case (see Michelsen 2006c). The basic idea is that biodiversity should be measured indirectly, applying the equation

$$Q=ES \times EV \times CMB \quad (11)$$

where Q is the quality of the area in terms of biodiversity, ES is the scarcity of the ecosystem, EV is the vulnerability of the ecosystems, and CMB is the conditions for maintained biodiversity within the ecosystem where the land use occurs (see Michelsen 2006c for details).

The methodology is immature and needs further development, but the first results indicate that the impact from land use is extremely important. In the timber case, land use impacts are more than 1800 times the impact from acidification if weighting factors from Eco-indicator 99 (Goedkoop and Spriensma 2001, see below) are used.

Weighting and aggregation environmental performance information

The debate on possible aggregation of environmental information in LCIA was presented in section 3.2.1. The criticism against aggregation and weighting is based on the fact that this process is subjective and value-based (Schmidt and Sullivan 2002; ISO 14040: 2006). Also, in ISO 14042: 2000 it is stated that '*weighting shall not be used for comparative assertions disclosed to the public*'. However, the idea of LCA as an objective, value-free assessment tool is heavily dismissed (e.g. Udo de Haes et al. 1999; Hertwich et al. 2000; Finnveden et al. 2002; Steen 2006). Steen (2006) presents an overview of the different value choices that are part of an LCA. The list starts with the basic elements, such as deciding to perform an LCA in the first place, defining the goal, setting the system boundaries etc. To some degree this is recognized in ISO 14044: 2006, but the consequence is not taken in ISO 14040: 2006 where it is argued that the reason that weighting should not be performed is the required value choices.

A possible weighting is hence just one of several subjective steps in an LCA. Hertwich et al. (2000) suggest that LCA should not merely be a decision support tool that only aggregates information, but also a tool that passes judgement regarding the importance of the different environmental impacts. This is in fact, often not fully recognized, a necessity for being able to determine which environmental impact categories to focus on. The selection of the suggested indicators in Table 3 is a result of such a process. The alternative is to include all environmental categories, but this is even more controversial since there is no common agreement on what is to be included in this list. Should for instance noise, smell, and radiation be included?

The relevance of the need to be able to prioritize between environmental impact categories is clearly demonstrated in the furniture case. In Michelsen (2006b) one of the suppliers is ranked as the second most important, responsible for 14.2% of the total upstream environmental impact when Eco-indicator 99 is used as LCIA method. However, the supplier turns out to be of minor importance if other LCIA methods are used.

For the furniture company the crucial question is if the environmental impact from this supplier is significant or not, and the only way of answering that question is to allow weighting of the results. The answer following the weighting is by no means unambiguous, but it is nevertheless important. The underlying reason for the uncertainty in this case is primarily the decision to include or exclude the land use impact on biodiversity. This is obviously a value laden question as well. As stated above, the land use impact on biodiversity is very important and if products and processes responsible for extensive land use are to be compared with other products and processes, the assessment of this aspect is important. This is for instance the case in comparisons between biofuel versus fossil fuel, wood versus concrete as building material, and wind or

hydro power versus petroleum based power. In all these cases the question is primarily to compare land use versus emissions of greenhouse gasses.

The trade-off within furniture production is more or less the same. Here the question is if wood components should be used instead of e.g. metals or plastics and vice versa.

To some degree, the whole debate on weighting and aggregation is somewhat artificial. First, as shown, LCA as an assessment method is laded with subjective (value) choices. Second, the original data does not disappear when they are aggregated. The recommendation here is therefore that the aggregated data should be presented, but all assumptions and underlying data must be readily available so it is possible to reveal that the conclusions could be different if different assumptions were taken and to ensure that the assumptions are consistent with the values of the decision makers and stakeholders (cf. Finnveden et al. 2002).

However, if products from different producers are to be compared, it is crucial that the environmental impact is assessed in a consistent way for the different manufacturers. This will be further discussed in section 5.2.2. Selection of weighting procedure should be a part of this since it is obvious that the different weighting procedures from time to time generate different results (Dreyer et al. 2003; Michelsen 2006b). However, weighting is commonly used (e.g. Hanssen 1999a; Finnveden et al. 2002) and it seems contra constructive to counteract the development of these methods. The recommendation from this work is to improve existing methods rather than counteracting this process.

5.1.3 Assessment of value performance

Advantages and disadvantages of using life cycle costs as a measure of value performance were discussed in section 3.2.2. It was indicated that one reason to focus on life cycle costs is that data on LCC normally is readily available as distinct from data on value added and profit in the different segments of the extended supply chain. Even if it can be argued that LCC in many cases is the only possibility since it is the only available measure on value performance, it is also shown to give valuable information about the products (see Table 2).

In section 3.2.2 it was pointed out that most LCC are performed primarily from the customers' point of view, which is not very different from the manufacturers' point of view. In Norway an additional reason for focusing on LCC is the Public Procurement Act that states that all official bodies have a legal obligation to take economic as well as environmental life cycle considerations when new acquisitions are planned (The Norwegian Ministry of Government Administration and Reform 1999).

In this work LCC is defined as the sum of acquisition costs, ownership costs and end-of-life treatment costs (cf. equation 5) and the usefulness of this measure on value performance is demonstrated in the furniture case (Michelsen 2006a; Michelsen et al. 2006a). In particular, in Michelsen (2006a) it is shown how changed assumptions can influence the LCC of a product and how this can be used to provide important information in front of decisions.

In Michelsen et al. (2006a) value performance is presented as $1/LCC$. This is in accordance with how Monczka et al. (2005) define value where the function simply is defined as 1 (cf. equation 4 - one unit of a function, in this case, a conference room chair with 20 years durability). From a customer's point of view this is adequate information in an acquisition process, and this also provides the manufacturer with the necessary information to meet the legal claims given in the Public Procurement Act.

It is a problem that there is no common understanding of LCC and no standardized framework for assessing LCC (Rebitzer 2002). However, there is work in progress within

SETAC¹¹, and a publication on definitions of LCC has been announced (Rebitzer and Hunkeler 2004).

In Michelsen et al. (2006b) value added is used as a measure of value performance together with total costs. As shown, the relative importance of the different processes might change significantly (see also Figure 22). The range of application is obviously quite different. From a customer's point of view, the cost breakdown is often less important; it is the final cost that is the big issue. Other stakeholders might however have different information needs. A high VA indicates for instance that a relatively high proportion of the costs are salaries and/or profit (cf. equation 7), and thus money remains in the local community where the operation takes place. Thus, VA gives more accurate information about the distribution of the income than costs alone. However, as pointed out in section 3.2.2 information on VA on process level is often not available and the situation for NVA (equation 8 and 9) is even worse. Costs are thus often the only reliable measure, but a break down of costs should always be achieved whenever possible.

In section 2.5.1 two distinct areas for eco-efficiency analyses were presented. If eco-efficiency is used to measure performance and improvements over time, it is obvious that monetary measures must be indexed. If not, changes in eco-efficiency performance will occur merely as a consequence of inflation. Following equation 4 inflation will decrease the value of a product if not indexed, while the opposite is the case during deflation.

5.1.4 Presenting eco-efficiency performance

In this thesis two primarily ways of presenting eco-efficiency performance are used, namely portfolio matrixes (xy-diagrams) for comparing products, and graphs for showing the relative contribution and distribution of value performance versus environmental performance in the different segments of the ESCs.

Portfolio matrixes

The use of portfolio matrixes for presenting eco-efficiency of products as described in section 2.5.1 has developed gradually. The case study on furniture, where such xy-diagrams are used, has further emphasised the advantages of their use (Michelsen 2006a, Michelsen et al. 2006a).

First, the matrixes visualize both the environmental performance and the value performance simultaneously. The merger of value and environmental performance into one single indicator (cf. equation 2) has been criticised since it in many cases obscures conflicting interests with respect to environmental and value performance (Azapagic and Perdan 2000; Lafferty and Hovden 2002). It is for instance possible to identify alternatives with a high eco-efficiency score that might not be economically viable. Such results will be revealed when both environmental and value performance are presented.

Second, the matrixes represent an easily understandable presentation of the results for non-specialists. The trade-off between financial and environmental interests is readily observable. They also meet the criteria in the Public Procurement Act in Norway (The Norwegian Ministry of Government Administration and Reform 1999) since they present exactly what is requested of environmental and life cycle cost considerations before new acquisitions are planned. For these reasons portfolio matrixes should be included in EPDs where more than one product is presented¹². This could be a valuable enhancement of such 'group-EPDs' before their format is finalized.

¹¹ Society of Environmental Toxicology and Chemistry, <http://www.setac.org/>

¹² see <http://www.nho.no/files/NEPD018hellNO.pdf> for an example

The last advantage mentioned here, is the possibility of using relative values. This makes it possible to omit data for processes present in all products that are compared, thereby, both reducing the data requirement for the analyses as well as the uncertainty of the results (see Michelsen 2006a). The advantages of relative values are also stressed by others (Lye et al. 2001; Saling et al. 2002; Schmidt et al. 2004).

As already pointed out in section 5.1.2, there is a debate on the level of aggregation of environmental information. Here it was argued that there are sound arguments for aggregating environmental information. Figure 19 shows a portfolio matrix where 6 chairs are compared, using a single score for environmental performance and 1/LCC for value performance (see Michelsen et al. 2006a for details).

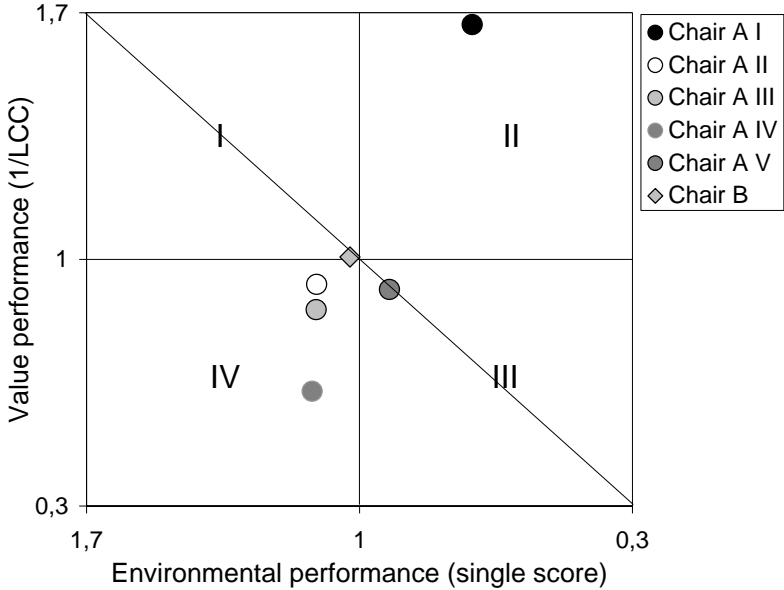


Figure 19 – Relative eco-efficiency for 6 different products using an aggregated single score for environmental impact (from Michelsen et al. 2006a)

The criticism against aggregation has primarily been focused on the reduced transparency of the results. As argued in section 5.1.2, this is somewhat peculiar since the original data does not disappear during the aggregation, and should be presented together with the aggregated data. This leaves the decision maker the option to follow the conclusions arising from the selected aggregation procedure, or to use the non-aggregated data to follow a different emphasise of the included environmental aspects (cf. Hertwich et al. 2000). Figure 20 and Figure 21 represent two different ways of presenting the environmental information behind Figure 19. In Figure 20 the different environmental aspects are presented one at the time; the performance for the selected environmental aspects is presented simultaneously in an eco-compass in Figure 21 (cf. Brezet and van Hemel 1997; Lye et al. 2001, see Michelsen et al. 2006a for discussion on data quality). The choice of one of these is primarily a matter of individual preference.

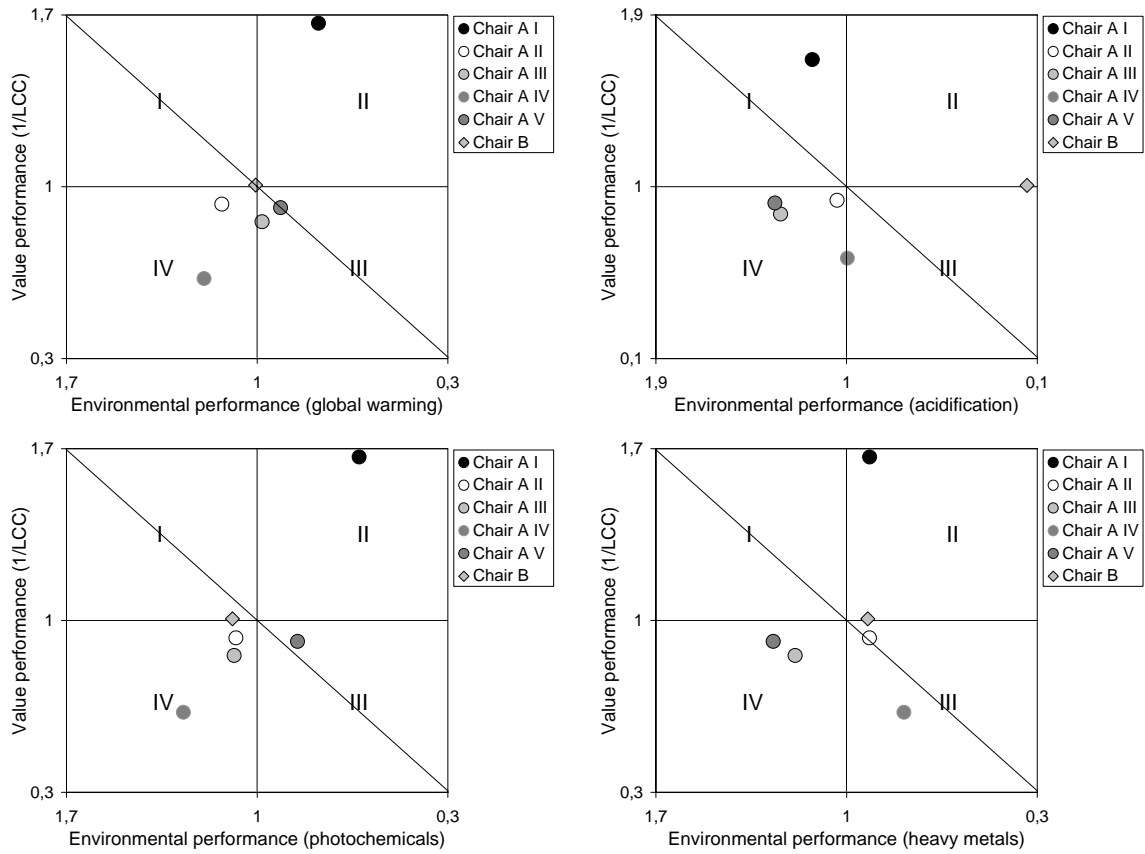


Figure 20 - Relative eco-efficiency for 6 different products using different measures for environmental impact (data from Michelsen et al. 2006a)

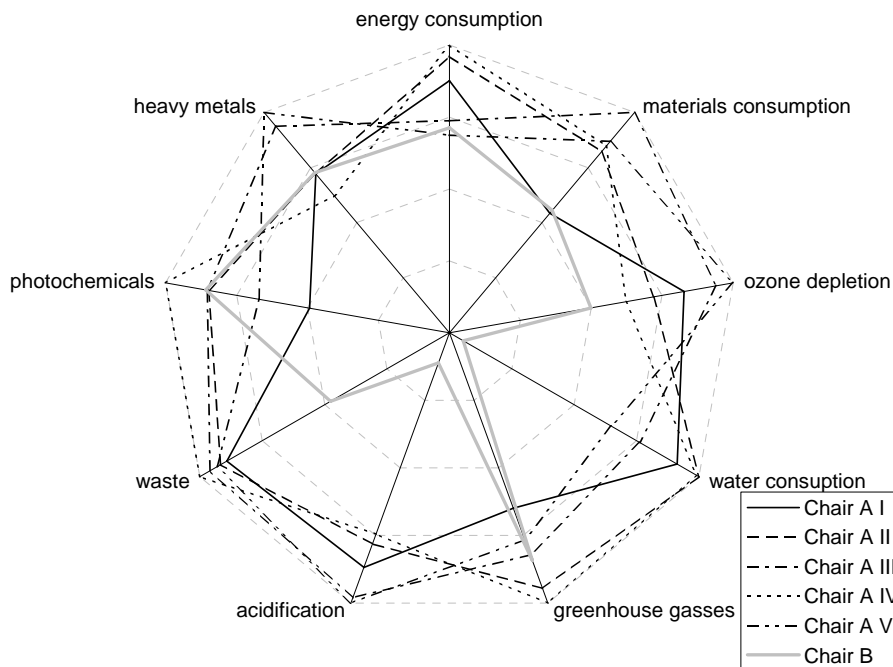


Figure 21 – Relative impact on different environmental impact categories for 6 different products (data from Michelsen et al. 2006a)

In Michelsen (2006a) it was shown that the use of portfolio matrixes is not only useful for existing products, but also for possible alterations of existing products.

Distribution of environmental and value performance within the ESC

As distinct from portfolio matrixes that are used to compare different products or different options for a product (e.g. Michelsen 2006a), contribution graphs are used to show the relative contribution for different segments within one supply chain. Figure 18 and Figure 22 show two examples. In Figure 18 the processes at the end producer and upstream and downstream activities are separated. This is a rather rough subdivision, but still it clearly visualizes the significance of the impact caused directly by the end producer.

Figure 22 is on a much finer scale where all processes necessary for cutting a log and bringing it to a factory gate are included. This represents only a part of the ESC. The application for these two figures is the same; to identify areas with a high contribution to the overall impact. Identification of these processes is important if the overall performance is to be improved. For example, if the emission of greenhouse gasses caused by the product presented in Figure 18 is to be improved, it does not make sense to focus on internal activities at the end producer but rather the end-of-life treatment. The difference in the two graphs shown in Figure 22 is that in the graph to the left the environmental performance is related to the value added directly by the specific process, while in the other, environmental performance is related to the total costs of the process (see Michelsen et al. 2006b for details).

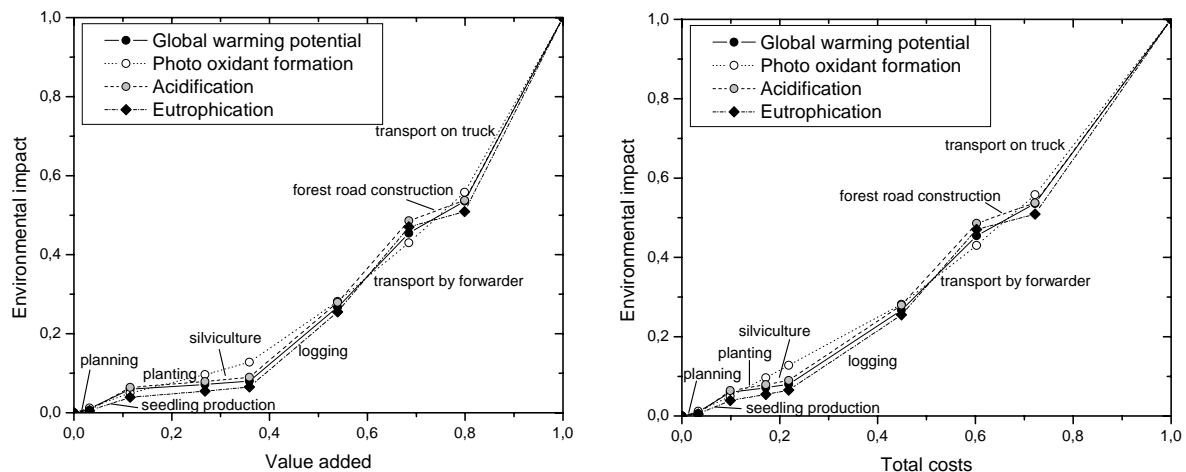


Figure 22 – Relative contribution of environmental impact and value performance assessed as value added and total costs for different processes in a production chain (from Michelsen et al. 2006b)

5.1.5 Summary of main findings and conclusions

The findings, recommendations and conclusions on methodological development are presented throughout the previous sections, but here are the main findings summarized:

- Defining system boundaries:
 - the system boundaries must include the life cycle of the products – the extended supply chain provides a useful system description for this purpose
 - system boundaries need not be set identically for assessments of environmental performance and value performance and can be defined independently
- Assessing environmental performance:
 - selection of environmental performance indicators must be done on a case to case basis and no standardized list is suitable to all assessments
 - however, selection of indicators should be based on a easy recognizable list as a core set, and expanded if necessary (cf. Table 3)
 - land use impact is important for products originating from land extensive activities and should be included as an indicator in these assessments

- industry sectors must take responsibility for deciding what kind of data should be used in environmental assessments (e.g. IO-data, marginal versus average data) to make comparisons possible
- the arguments against weighting and aggregation of environmental impact categories are weak and aggregation should be performed, provided that industry sectors can agree on weighting procedures and original data are made readily available
- Assessing value performance:
 - LCC (and 1/LCC) is in many cases the only available measure on value performance of products, but also provides adequate information on the overall performance of products
 - if possible, cost breakdowns should be performed to reveal the distribution of VA and/or NVA in the ESC together with LCC
 - externalities should not be included in cost assessments performed as part of eco-efficiency assessments
- Presenting eco-efficiency results:
 - these cases confirm the usefulness of xy-diagrams for presenting eco-efficiency performance for different ESCs/products and contribution graphs to show the relative contribution of the different segments

5.2 Regulatory questions

Regulations have the potential to change the eco-efficiency performance requirements of products and new regulations can motivate alterations within ESCs. Here two distinct types of regulations will be discussed, namely public (juridical) regulations and regulations imposed by the industry sector themselves (cf. Kaplinsky 2000).

5.2.1 Public regulations

Bleischwitz et al. (2004) identify three areas where public regulation can impose changes in the environmental performance of the material cycle; (1) taxation on raw materials, and licenses to operate and agreements on exploitation of raw materials, (2) integrated product policy (IPP) and quotas for recycled inputs, and (3) regulation on landfills, taxes on emission from incineration and technical standards for end-of-life treatment (Figure 23). For the overall environmental performance of the life cycle of a product, regulations and taxes on emissions from production and consumption are possible as well.

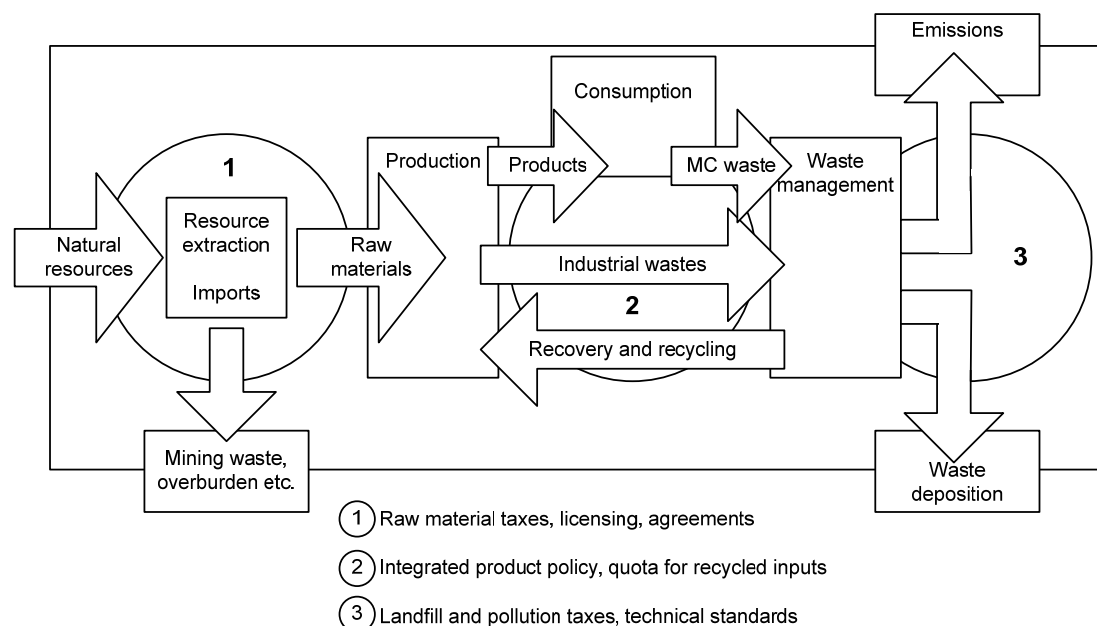


Figure 23 – Possible areas for regulations of material cycles (from Bleischwitz et al. 2004)

Several authors have claimed that authorities should set targets for environmental improvements, while the industry should be given the opportunity to decide how the targets should be fulfilled to ensure maximum opportunity for innovations (e.g. Porter and van der Linde 1995; Bleischwitz 2003). Measures on eco-efficiency can be useful here. The environmental dimension can be used to assess if environmental improvements are met (e.g. 10% reduction of greenhouse gasses). Cropper and Oates (1992) focus on the two-step process necessary for reaching the environmental goals; first the targets for environmental quality must be set, and then a regulatory system must be designed and put in place to monitor the achievement of this target.

More importantly in the context of eco-efficiency is the possibility of simultaneously presenting the environmental and financial consequences of the different alternatives to reveal if some of the environmental improvements are less realistic due to high costs.

In Michelsen (2006a) 6 different scenarios for improving an existing ESC are assessed. The product here is the chair named 'Chair A IV' in the previous section (Figure 19, Figure 20, Figure 21, see also Michelsen et al. 2006a). A brief summary of the scenarios is given in Table 4, for more details see Michelsen (2006a). The results are shown in Figure 24.

Table 4 – Summary of scenarios for environmental improvements of an extended supply chain

Scenario	Alterations of the extended supply chain
A	Reducing the amount of polyurethane in the product
B	Replacing polyurethane with an innovative compound called 'maderon'
C	Omitting polyurethane
D	Introducing dismantling and recycling activities
E	Introducing take-back with reuse of steel components
BE	A combination of B and E

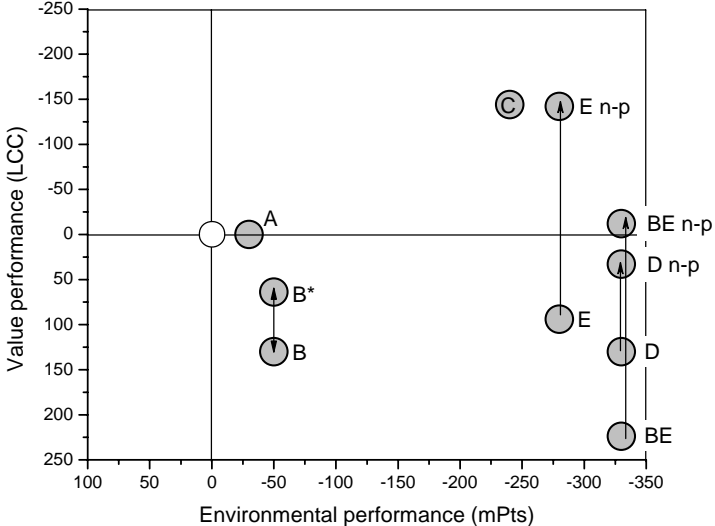


Figure 24 – Changes in eco-efficiency following different scenarios for improvements (from Michelsen 2006a)

As seen from the figure, the largest improvements take place where the end-of-life treatment is altered. Two different cost alternatives are calculated for these scenarios; one where all extra costs are calculated like any other costs and one where the costs due to dismantling and reuse are calculated as non-profit activities where no margins are

included (denoted n-p in Figure 24, see Michelsen 2006a for assumptions and calculation details).

The results show that all alternatives give higher life cycle costs if dismantling and recycling costs are included as any other costs. The exception is alternative C that causes lower seating comfort, and alternative A that only gives a minor improvement of the environmental performance.

However, regulations can change the financial preconditions for different options and motivate alterations. In this case, it is obvious that some sort of alteration of the end-of-life treatment is preferable if environmental performance is to be improved significantly. The possible effect of the following regulatory options will here be discussed;

- 1 - Include a tax on emissions
- 2 - Increase the tax on landfill
- 3 - Put a tax on raw material consumption
- 4 - Introduce IPP

The total emission of greenhouse gasses from the life cycle of the product is 35.7 kg CO₂-equiv. (Michelsen et al. 2006a). Given a CO₂-tax of 300 NOK/tonne which is about the highest used in any industry sector in Norway at present¹³, this will cause an increased cost of slightly above NOK 10 per product. In addition, with the proposed scenario alterations, about 40% of the emissions remain in the improved ESC (data not shown). It is most likely that such a tax would cause only a slight increase in the LCC.

Similarly, if the landfill taxes are doubled, the increased costs per product would be slightly above NOK 10. It is also hard to imagine a tax on raw material consumption that is high enough to impose changes, given the fact that scenario E caused 94 NOK in additional costs per product and the increase caused by scenario D is even higher. An additional challenge with such taxations is the difference in regulations in different countries. A possible outcome of a new tax imposed to improve the environmental performance might be that the production is moved to other countries with lower environmental standards (cf. Clift and Wright 2000). The above mentioned options thus appear unlikely to motivate improvements since a small increase in taxation most likely will result in increased LCC, while a high increase of taxation most likely will result in a move of the production to a country without such taxation.

Thus, in this case only one realistic option is left for the authorities, namely introducing some sort of extended producer responsibility. If take-back legislation is introduced for furniture, the industry itself could look into alternatives like D and E. This possibility has already been presented (The Norwegian Ministry of Environment 1999) and the results in Figure 24 show this would give significant environmental improvements. This would also make sense according to basic principles in industrial ecology since this would transform present linear material flows towards closed loops (cf. Figure 5), which is in fact the basic intention for all take-back programmes (Røine 2006).

When it comes to consequences for the life cycle costs of the products, the analyses show that take-back legislation does not necessarily cause increased costs. Non-profit organisations are established in other industries in Norway taking care of used products and materials (Røine and Lee 2006) and there is no obvious reason why this could not be done for furniture industry as well. Also, experiences from other sectors show that the costs are often reduced when such regulations are introduced, partly due to technological innovations (Finster et al. 2002; Røine and Lee 2006) and partly to improved performance in reverse logistics (Clendenin 1997; Clift and Wright 2000).

¹³ http://www.miljostatus.no/templates/PageWithRightListing____2334.aspx

An alternative not discussed in Michelsen (2006a) is the possibility of a second hand market. Handfield et al. (1997) show how this was the outcome of take-back solutions on furniture in USA. Given the functional unit in the analyses, every year of use beyond the assumed 20 years reduces the environmental impact significantly. One additional year beyond the assumed 20 will give a 5% decrease in environmental impact for every year the chair has been used. This is thus one of the most effective ways of increasing the environmental performance (Collins et al. 2005). However, this should be regarded as a supplement for directed end-of-life treatment and not an alternative. Thus, design for recyclability is an important outcome of prospects for take-back responsibilities (cf. Finster et al. 2002).

5.2.2 Industry imposed regulations

Sometimes overlooked, industry imposed regulations can play an important role in the performance of products. As shown in section 5.1 (see also Michelsen et al. 2006ab and Figure 25), environmental performance is unevenly distributed in the extended supply chains. The main impact often originates from actors other than those who are exposed to the pressure for improvements (Hall 2000), and industry imposed regulations might thus be important to disperse the environmental focus to the actors who make the largest contribution to the environmental impact.

Here as well, there are two distinct types of regulations. First, there are regulations imposed within an industry sector. Product Category Rules (PCR) are important since these represent an agreement within the industry sector on how environmental performance is to be assessed (cf. Fet et al. 2006b). This does not automatically result in improvements, but it makes comparisons possible and thus makes it possible for purchasers to choose the product with lowest impact and thus motivate improvements. These regulations can be seen as horizontal regulations.

The second set are regulations found within a supply chain (Kaplinsky 2000). Manufacturers might set standards for suppliers regarding environmental quality, they might monitor the performance of their suppliers and they might also help them to achieve the standards. The introduction of ISO 14001 certification and PEFC standards in the timber case presented in section 4.2 is an illustrating example. Here, environmental performance acts as an order qualifier (cf. Handfield et al. 2005) and if the requirements are not met by the suppliers, they loose this business opportunity. These regulations can be seen as vertical regulations.

To make setting and monitoring regulatory targets efficient, a harmonization of environmental performance assessments is preferable. When different companies and alternatives are to be compared, the performance must be assessed in an equivalent way. This was discussed in section 5.1. Some 'general' guidelines are also available, such as the Sustainability Reporting Guidelines developed by Global Reporting Initiative (2002) and the ISO-standard on environmental performance evaluation (ISO 14031: 1999).

5.3 Organizational questions

The possibilities for a manufacturer to introduce internal regulations in a supply chain depend on the channel power of the companies. This will be discussed in the next section.

5.3.1 Structure and power within a supply chain

It is previously stated that the pressure to provide environmental information and document improvements is unevenly distributed within the supply chain. Figure 25 gives a schematic overview. The pressure from authorities, neighbors, NGOs etc. can be directed towards all actors in the supply chain. However, even if the environmental impact tends to be higher in the early stages of the supply chain, the pressure here

might in fact be low or entirely absent since e.g. extraction of raw materials often is situated in countries with lower environmental standards (cf. Clift and Wright 2000). The pressure from consumers is quite different since this primarily is directed towards the retailer and the end producer (Hall 2000). This pressure is becoming more and more important (see section 2.2).

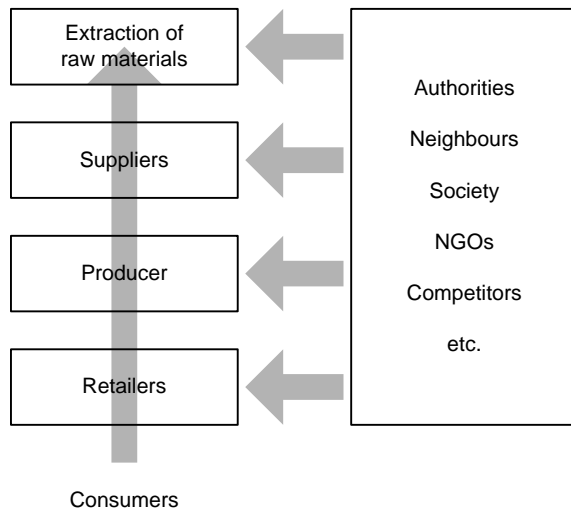


Figure 25 – Distribution of pressure to provide environmental information and improve environmental performance in a supply chain (modified from Hall 2000)

If the end producers are to be able to provide information on the performance of the life cycle of their products and make improvements, they have to know about both the upstream and downstream activities and they have to be able to influence the processes. Organizational issues are thus essential also when it comes to assessing and improving the eco-efficiency of ESCs.

It is documented that most companies' network horizons are narrow and the knowledge on the upstream activities limited (Håkansson and Johanson 1992; Lambert and Cooper 2000; Holmen and Pedersen 2003). This means that even if they require information on environmental performance from the actors they know, this will in most cases be a limited part of the supply chain. This is also demonstrated in the furniture case where the end producer with a few exceptions does not know anything about the suppliers of their suppliers, and they have no explicit knowledge on extraction of raw materials used in their products (Michelsen 2006b). However, in most cases it is neither possible nor practical to acquire too much knowledge about a large part of the supply chain (Håkansson and Snehota 1995) and the end producer should rather disperse the demands for environmental performance out to their suppliers.

The key issue is thus the ability a producer has to influence other actors in the supply chain. It is unlikely that a company, and in particular SMEs, will be able to manage the entire supply chain (Lambert and Cooper 2000) and it is thus important to prioritize. In Michelsen (2006b) a methodology for identifying the most relevant suppliers is presented. The suppliers are here investigated with respect to their contribution to the overall environmental impact of the final product and the assumed potential for improvements. In the case study used to test the methodology, an initial number of 14 suppliers were reduced to 3-5 suppliers (Michelsen 2006b). This should be a manageable number of suppliers.

The second question is how the environmental impact of the production of the components these suppliers provide can be reduced. Two alternatives exist; the relevant suppliers must improve or they must be replaced.

Finding replacements depends on the availability of alternative suppliers with a better or potentially better performance. The complexity of the supply market must thus be analysed (cf. Kraljic 1983). However, the known analytical tools for assessing the supply market are unfortunately not applicable for the furniture case. For most of the components there are only a few possible suppliers available which should indicate a high complexity. On the other hand, the end producer or a subsidiary company is able to manufacture most of the components themselves with only minor investments. This indicates a low complexity. However, the end producer has no intention of changing the suppliers (see Michelsen 2006b) and since the possibilities for analysing the market also are limited, the replacement option is not investigated further.

Hall (2000) has documented that lower profile suppliers lack incentives to improve their environmental performance, and if a buying company is not able to motivate their suppliers to improve, they must be able to force them. This requires a channel leader with sufficient channel power. Channel power is understood here as the ability of one channel member to control the decisions of another (El-Ansary and Stern 1972).

In Michelsen (2006b) it was assumed that a company responsible for more than 5% of the total sales of a supplier has possibilities to influence the behaviour of that supplier (see Michelsen 2006b for details). In the furniture case the conclusion then was that the end producer had possibilities to influence only two of its suppliers based on the size of the turnover (see Figure 27).

However, a company might have possibilities to influence its suppliers even though the share of total sales is low. If the companies within the supply chain have the same understanding of the market, the suppliers will probably have the same interest in focusing on environmental performance (cf. Pagell and Krause 2002). It is also possible that suppliers would try to avoid losing a contract due to unsatisfied environmental performance since this could give a bad reputation, and thus try to satisfy the customer, even one of minor importance (cf. Forman and Jørgensen 2004).

In the timber case, it is obvious that one actor (Norske Skog) had enough channel power to force its suppliers (including ALLSKOG) to introduce environmental management with specified requirements for environmental performance for forestry operations (Michelsen et al. 2006b). Figure 26 shows that the dependency on the largest customers also is increasing. In 2002 12 customers were responsible for 90 % of the sales measured in m³. In 2005 the number had decreased to only 7.

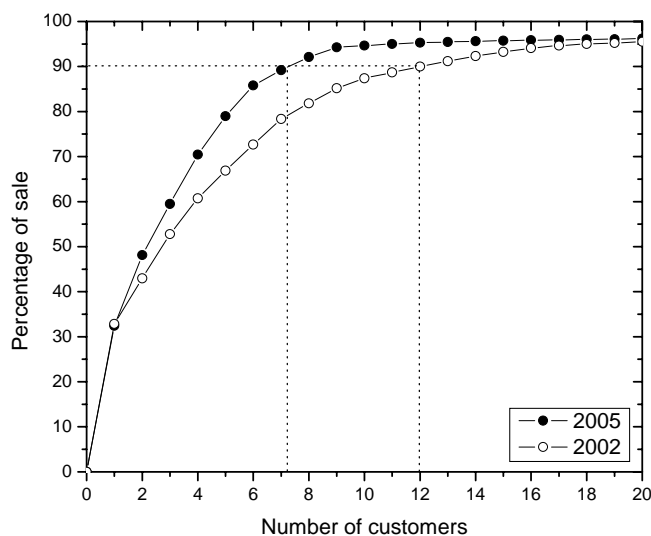


Figure 26 – Relative importance of ALLSKOG’s customers based on sales in m³

The situation in the furniture case is quite the opposite. However, as discussed in Michelsen (2006b) a channel leader does not have to be a company, it could well be a consortium. The formation of a purchasing consortia can be based on a common need, e.g. for environmental information (cf. Telle and Virolainen 2005; Michelsen 2006b) and is thus an option for the furniture industry.

In Figure 27 the potential outcome of a purchasing consortium within the furniture industry is shown. Where the single manufacturer has enough channel power to influence two suppliers, the furniture industry as such has enough power to influence 11 of the 14 suppliers, including all suppliers that are identified as important with regard to their environmental performance. The numbers in the figure represent the rank the suppliers have according to their relevance for the overall environmental performance (see Michelsen 2006b for details).

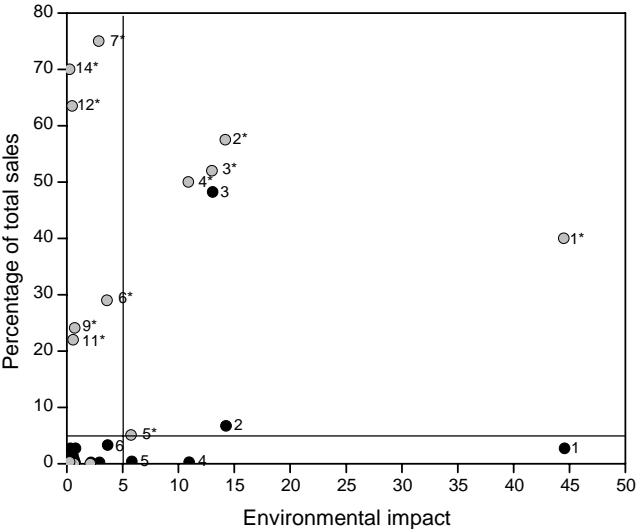


Figure 27 – Upstream environmental impact from suppliers and the possibilities to influence them (the company’s possibilities marked with closed circles, the furniture industry as a whole with open circles – from Michelsen 2006b)

An apparent conclusion is that as the research cooperation between NTNU and the furniture industry continues (cf. Fet et al. 2006b), the possibilities for establishing a consortium should be examined since the manufacturers have common needs. In the beginning this could be limited to the need to obtain more accurate information on environmental performance from their mutual suppliers, but later this could also include an effort to increase the performance as such.

There are numerous examples of manufacturers helping their suppliers to improve their environmental performance (Taylor 1992; Handfield et al. 1997; Tukker 2004; Handfield et al. 2005). However, in these cases the manufacturers are large companies helping their small and medium sized suppliers. In the furniture case this is not the situation since the manufacturers are SMEs themselves and most likely lack the necessary resources to help their suppliers. It should be further investigated if this is also an opportunity for the furniture industry to take the lead. As shown in Figure 27, 6 of the suppliers are delivering more than 50% of the total production (in monetary terms) to the furniture industry, and additionally 4 are delivering more than 20%. This raises the question of what is actually a furniture company. Some of these suppliers should probably be included in the above mentioned project since these are likely to be responsible for a much higher proportion of the environmental impact of furniture production than the end producers themselves (cf. Figure 18). As discussed in section

2.3, in-sourcing of environmentally significant processes might also be a strategy since this secures a complete control over the environmental impact from these processes (cf. Handfield et al. 1997) or a strategy involving increased ownership in the suppliers. This will obviously increase the possibilities of influencing the performance of the processes. Long time relationships with the suppliers might also to some degree have the same effect since a mutual dependency and understanding will be developed (cf. Lamming and Hampson 1996; Christopher 1998; Håkansson and Waluszewski 2002).

Cooperation within industry is also shown to be important when take-back is introduced (Røine and Lee 2006). The establishment of non-profit organizations that run the take-back systems have been an important success factor in other industry branches (Røine and Lee 2006) and could probably be an option within furniture industry as well. As shown in section 5.2.1 take-back of furniture could give significant improvements in the environmental performance of furniture, and if this is organized as non-profit activities, even the LCC of the products might be reduced.

5.3.2 *Introducing a functional economy*

A question not dealt with in this work, is the possibilities of introducing a functional economy. There are already several examples where the function and not the product is sold (e.g. Clendenin 1997; Tukker 2004), and this could be a possibility for furniture as well.

A functional economy has several advantages. First the user and the manufacturer of the products will have the same focus on reducing the life cycle costs (cf. Tukker 2004). The potentially conflicting interests discussed in section 3.2.2 will be removed and all actors involved in the supply chain will have the same goal. This opens the way for improved cooperation. A positive side effect for the user is the certainty of the level of the costs of a service since all cost factors in equation 5 will be merged into one single factor.

An increased focus on providing services will also result in more innovations for improved durability (Bleischwitz 2003). As pointed out in section 5.2.1 this is one of the most efficient ways of improving the environmental performance of a product (cf. Collins et al. 2005). The outcome is often increased use of spare parts and an increased reuse of components. The different life cycles indicated in Figure 10 thus get different length and the period of use for a product is no longer restricted by the component with the lowest durability.

5.4 *How to achieve significant improvements in eco-efficiency – final remarks*

As pointed out in chapter 2, eco-efficiency is not the same as sustainability. Eco-efficiency only incorporates environmental and economic concerns while social issues are left outside. In addition, eco-efficiency only deals with relative environmental impact. It is possible to improve the eco-efficiency and at the same time increase the total environmental impact. Figge and Hahn (2004) state that a sustainable measure must consider the efficiency and the effectiveness of all three dimensions of sustainability simultaneously and measures on eco-efficiency are only a part of this.

Nevertheless, if used correctly, measures on eco-efficiency should enable a move towards sustainability. The process of moving towards sustainability is in fact a sustainable development according to Clift (2000).

In this thesis three issues have been treated; methodological, regulatory and organizational. Earlier in this chapter it is shown that these can not be treated independently but are closely interlinked if the potentials for improvements are to be realized.

The first issue is methodological ability. If improvements are to be achieved, it is necessary to be able to analyse both the present situation and the magnitude of the improvements. Just by introducing assessments of eco-efficiency improvements will occur (cf. Porter and van der Linde 1995) and win-win situations will be revealed (Bleischwitz 2003). The expanding focus from processes to the life cycle of the products will bring along life cycle thinking that consequently will cause a shift of paradigm such that improvements are sought along the entire life cycle (Rebitzer 2002).

Detailed assessments make it possible to see where the improvements can be performed. This is followed closely by an investigation into organizational issues to see if it is *possible* to carry out the improvements within existing or altered ESCs. Some improvements might be carried out by the end producer directly, but as pointed out in the previous section, it is often necessary for the end producer to be able to influence the performance in other parts of the extended supply chain.

Related to the furniture case some possibilities for the furniture industry in Norway were discussed and the presumably most important option is to join forces within the industry sector to secure sufficient channel power. A critical point is if the furniture manufacturers in Norway primarily regard each other as competitors, or if they regard each other as allies in the competition against producers in other parts of the world, especially low-cost countries. If they regard each other primarily as allies, there should be no real obstacles to form a purchasing consortium as described in section 5.3.1 and use this to improve their ability to influence the supply chains. Some of the furniture manufacturers have already shown at least a common understanding of the challenges on environmental performance and already cooperate in obtaining information on environmental performance (cf. Fet et al. 2006b). Several authors have underlined that the development of a purchasing strategy is the first step towards a supply chain strategy (e.g. Pagell and Krause 2002).

The magnitude of improvements following such assessments is limited in most cases. Following Brezet (1997), this could include product improvements and product redesign, which occasionally can reach factor 5 improvements (cf. Figure 8). In the case example on redesign of a chair the potential improvements were considerably less (Michelsen 2006a), but here only some very short term alternatives were assessed so the actual potential might still be higher.

To achieve higher levels of improvements, policy instruments must be introduced. Regulatory policies are important (cf. Bleischwitz et al. 2004) and there are numerous examples documenting environmental improvements as a consequence of new regulations (Reijnders 1998; Shapiro 2001; OECD 2006).

Ruud (2002) claims that regulatory measures are needed to promote more sustainable changes in production and consumption patterns. One example is reuse and recycling of components. Clift (2003) has shown that with the present production regimes it is unlikely that companies will initiate take-back and recycling voluntarily since this is unprofitable in most cases. Thus, take-back must be initiated through authority regulations.

However, when take-back first is introduced, this might generate new technological innovations (Røine and Lee 2006) and take-back systems might well be the first step towards a functional economy. Here, the probabilities for function and system innovations are higher and consequently also the potential for improvements (cf. Brezet 1997; Tukker 2004).

The degree of improvements is depending on the system boundaries. O'Rourke et al. (1996) state that any system can be efficient as long as it is defined to be small enough. The converse may also apply; sometimes it is necessary to define the systems large

enough to improve the performance. In the timber case (Michelsen et al. 2006b) improvements on factor 4 or higher are hard to achieve unless large quantities of environmentally friendly engine fuel is made available or if only the logs with the lowest environmental impact are taken out (cf. the best case in Michelsen et al. 2006b). Increased logging might increase the environmental impact per produced unit. Here the system is defined as the delivery of logs at the gate of a factory. The situation might be totally different if timber were compared to alternative materials, e.g. concrete, and due to the narrow system boundaries in this assessment it is not possible to see if the timber logging is a significant part of the problem and should be improved, or if it is a part of the solution and should be expanded to provide more wood to substitute other materials.

It is difficult to see that industry, at least in a short term, is able to take responsibility for the total environmental impact. Measures on eco-efficiency are useful for industry, while the total impact (by some identified as the eco-effectiveness, cf. section 2.5.2) should be handled by the authorities. It should be underlined that it is not possible to state that a product or an option is *eco-efficient*. Eco-efficiency is a relative measure, thus it is only possible to say that a product or an option has a higher or lower eco-efficiency than alternative products or options.

It is necessary to identify different targets depending on the system level. Authorities should set targets for absolute environmental impact and on basis of these targets for the eco-efficiency for products could be deduced. This is in accordance with the two-step process pinpointed by Cropper and Oates (1992).

The results from this thesis show that the criticism against the use of eco-efficiency for larger system improvements is, at least to some extent, unjustified (cf. section 2.5.2). It is however necessary to be aware of the possible applications of measures and targets for eco-efficiency performance described above and not use assessments on eco-efficiency in the belief that this covers the entire field of sustainability.

Several authors (e.g. Kleijn et al. 2002; Hagelaar et al. 2004) have stressed that the higher the targets for improvements are, the higher are the needs for integration and information within the supply chains. This brings the topic back to organizational aspects and the importance of developing long term relationships.

6 Conclusions

The topic in this thesis is eco-efficiency in extended supply chains with a focus on methodological development and the related regulatory and organizational implications. This tripartition is reflected in the research subjects identified for P2005 Industrial Ecology (Brattebø and Hanssen 2000), in the specification of research questions for eco-effective supply chains (Fet and Johansen 2000), and in the research questions identified for the thesis. As discussed in section 5.4, these three issues are not independent of each other, but are closely interlinked.

The development of an applicable methodology for assessing eco-efficiency for extended supply chains has been a major part of the thesis. A discussion and recommendations on system boundaries and measures of value performance and environmental performance is presented. The recommendations are based on previously existing methodologies, but are further clarified and refined. The usefulness is in particular demonstrated in two of the papers (Michelsen 2006a; Michelsen et al. 2006a).

There are several unsolved issues concerning the LCA-methodology that are presented in section 3.2.1. Some recommendations are given on aggregation and inclusion of land use impacts on biodiversity, but the other questions have been outside the scope of this thesis. An important contribution of this thesis is a recommendation on how to include land use impacts on biodiversity given in Michelsen (2006c), however, this topic is still in an early stage of development and in need of further research.

In addition, there are still challenges in the assessments of value performance. The relationship between added costs and (net) value added should be further explored. Life cycle costs have been shown as a useful measurement on value performance, but information on NVA and VA could give additional insight. In Michelsen et al. (2006b) it is shown that this could be of importance, but the cases in this thesis have been too limited to investigate this in depth. This could be used for instance in an assessment of where in the supply chain the profit is created. This information could be useful for investigating the connections between created profit and environmental impact in different regions in globalised supply chains (cf. Kaplinsky 2000; Clift 2003).

The overall conclusion is that the methodology presented and the recommended system boundaries are useful for assessing the eco-efficiency for extended supply chains.

Regulatory and organizational implications are discussed in detail in chapter 5, and also in some of the papers (Michelsen 2006b; Michelsen et al. 2006b). Here it is important to stress that the recommendations given in chapter 5 and in particular in section 5.4 are case specific. It is not possible to draw general conclusions about how new regulations could be used to improve the eco-efficiency of products, or how new organizational structures should be developed. The recommendations are based on the case studies and are totally case specific.

Nevertheless, the selected approach and the methodologies used to reveal implications and give recommendations on regulatory and organizational aspects are shown to be applicable. In Michelsen (2006b) the number of relevant suppliers was reduced from 14 to 3-5, suggesting that it is sufficient to work with a manageable number of suppliers to control most of the environmental impact in that particular case. It was also shown in what manner the end producer could gain sufficient channel power to actually control the environmental impacts in the supply chain and the organizational implications include both vertical and horizontal structures. Even though the results as such are not transferable to other cases, the methods used for assessing the supply chain are shown to be both informative and effective.

In the same manner, results from Michelsen (2006a) and the discussion in section 5.2, shows that measures of eco-efficiency can be used to give recommendations on how regulations should be directed to impose real improvements in environmental performance. Also here it is the methodology that is of general interest, not the results per se since these are case specific.

There are still open questions on these topics as well. There is still a need for more research on the effectiveness of distributing environmental concern throughout the extended supply chain. As pointed out, purchasing strategies might act as a starting point for supply chain management, but there could be other alternatives. Kaplinsky (2000) also raises questions on the governance of the supply chain; who are the key actors taking the responsibility for the inter-firm division of processes and profitability. Do the actors with direct responsibility for the environmental impact have sufficient financial resources and knowledge that enables them to make improvements? This question is not addressed in this thesis.

A closing remark should be on the difference between eco-efficiency and sustainability. Measures of environmental and economic performance are reasonably well developed, but there should be an increased effort to associate these with measures of social performance. As already pointed out, the distribution of profit within the supply chain could be a good starting point. As mentioned in section 2.2, Handy (2002) claims that the purpose of business is to make a profit so that business can do something more or better. If this holds as a general assumption, the distribution of profit is thus a measure of the potential to take action on social and environmental issues. New challenges such as socio-economic and socio-ecological relationships must be included (cf. Dyllick and Hockerts 2002; Fet and Michelsen 2003) and the environmental supply chain focus must be expanded to include an implementation of corporate social responsibilities in global supply chains.

Finally, the question about total environmental impact, identified by some as eco-effectiveness, is not dealt with in the thesis. As pointed out in section 5.4 the recommendation here is that this should be handled by the authorities and targets for total environmental impact should then be used to set targets for eco-efficiency.

Eco-efficiency in business practices does not solve all the challenges on the path to environmental sustainability, but as Hunkeler et al. (2004) underline, it is better to implement a better practice than wait for the best practice that ensures sustainability. Hopefully this thesis is a contribution to the body of better practices.

7 References

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Appendix A

List of abbreviations

List of abbreviations

BCSD	Business Council for Sustainable Development
CERA	Cumulative Energy Requirement Analysis
CP	Cleaner Production
DfE	Design for the Environment
EA	Environmental Auditing
EAc	Environmental Accounting
E/E	Eco-efficiency
EMS	Environmental Management System
EPD	Environmental Product Declaration
EPE	Environmental Performance Evaluation
EPR	Extended Producer Responsibility
ESC	Extended Supply Chain
ESCM	Environmental Supply Chain Management
FSC	Forest Stewardship Council
GDP	Gross Domestic Product
IChemE	The Institution of Chemical Engineers
IE	Industrial Ecology
IO	Input-Output
IPP	Integrated Product Policy
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory Analysis
LCIA	Life Cycle Impact Assessment
LCS	Life Cycle Screening
LCM	Life Cycle Management
MET	Material, Energy, and Toxic-analysis
MIPS	Material Input per unit of Service
NTNU	Norwegian University of Science and Technology
NVA	Net Value Added
PCR	Product Category Rules
PEFC	Originally used for 'Pan European Forest Certification scheme', but lately changed to 'Programme for the Endorsement of Forest Certification schemes' (see www.pefc.org)
SCM	Supply Chain Management
SE	System Engineering
SETAC	The Society of Environmental Toxicology and Chemistry
SMEs	Small and Medium sized Enterprises
TMR	Total Material Requirement
VA	Value Added
WBCSD	World Business Council for Sustainable Development

Appendix B

Eco-efficiency in extended supply chains: A case study of furniture production

Eco-efficiency in extended supply chains: A case study of furniture production

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Abstract

This paper presents a methodology about how eco-efficiency in extended supply chains (ESCs) can be understood and measured. The extended supply chain includes all processes in the life cycle of a product and the eco-efficiency is measured as the relative environmental and value performance in one ESC compared to other ESCs.

The paper is based on a case study of furniture production in Norway. Nine different environmental performance indicators are identified. These are based on suggestions from the World Business Council for Sustainable Development and additional indicators that are shown to have significant impacts in the life cycle of the products. Value performance is measured as inverse life cycle costs.

The eco-efficiency for six different chair models is calculated and the relative values are shown graphically in XY-diagrams. This provides information about the relative performance of the products, which is valuable in green procurement processes.

The same method is also used for analysing changes in eco-efficiency when possible alterations in the ESC are introduced. Here, it is shown that a small and realistic change of end-of-life treatment significantly changes the eco-efficiency of a product.

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Keywords: Eco-efficiency; Environmental performance; Value performance; Extended supply chain

1. Introduction

There is growing focus on environmental reporting and use of environmental product declarations (EPDs). There is also a tendency that the environmental performance of products often is of importance when decisions about procurement take place (e.g. Dahl et al., 2002; de Bakker et al., 2002), even though this is far from unequivocal (e.g. Vogtländer et al., 2002).

There are reasons to believe that this trend will continue with the increasing focus on 'green procurement' in the public sector as a catalyst. The European Commission (2001) emphasises this opportunity in a green paper on Integrated Product Policy (IPP) and since 1999 in Norway, like some other countries in Europe, all official bodies have a legal obligation to take both life cycle cost and environmental performance of products into consideration when new acquisitions are planned. The European Commission (2003) has also announced ambitious goals for green procurement

within 2006. In Norway, public procurement represents 19% of GDP, which is slightly above the average in the EU (OECD, 2000). Given the importance of public procurement, there is no doubt that increased focus on environmental performance in the public sector will have a great impact on business. Companies that are not able to provide information about the environmental performance and the life cycle costs of products may face difficulties in getting contracts with the public sector in the future.

Measures of eco-efficiency are steadily becoming more common in industry. These are expanding from site-specific measures to include larger systems. Many companies have also realised that it is not only the individual companies that are competitors, but also the supply chain as a unit (e.g. Christopher, 1998; Lambert and Cooper, 2000; Mentzer et al., 2000; Mont, 2002). Information from the supply chain is thus of increasing importance also when competitiveness is considered.

This paper presents the concept of eco-efficiency in extended supply chains and exemplifies this with results from furniture manufacturers in Norway.

A goal is to identify performance indicators that can be used simultaneously for an extended supply chain and for the individual companies involved. It is a challenge to find

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indicators that are easily understood by non-specialists for communication purposes, such as in EPDs. In the case study, the environmental performance is addressed by Life Cycle Assessment (LCA) and the value performance by Life Cycle Costs (LCCs).

The information demand is primarily seen from the point of view of the users of the products. Top management does, however, have an almost similar demand for information (Kleijn et al., 2002), so the indicators have internal as well as external utility.

2. Definitions and concepts

2.1. The extended supply chain

Christopher (1998) defines a supply chain to be ‘the network of organisations that are involved, through upstream and downstream linkages, in the different processes and activities that produce value in the form of products and services in the hand of the ultimate consumer.’ All supply chains are thus in principle infinite, and criteria for selection of boundaries must be set. Christopher (1998) also uses the term ‘extended supply chain’ which includes use and disposal. The term emphasises the focus on the companies involved and incorporates the life-cycle perspective. This is in accordance with the perspective in this paper and the term ‘extended supply chain’ (ESC) is thus used to describe the systems in the case study.

2.2. The eco-efficiency concept

The World Business Council for Sustainable Development (WBCSD) has been credited for inventing the term eco-efficiency in the book *Changing Course* (Schmidheiny, 1992). The purpose of eco-efficiency is to maximise value creation while having minimised the use of resources and emissions of pollutants (Verfaillie and Bidwell, 2000). Measuring eco-efficiency is important in order to measure the decoupling of economic growth and environmental pressure.

Eco-efficiency is in most cases expressed by the ratio

$$\text{Eco-efficiency} = \frac{\text{Product or service value}}{\text{Environmental influence}}$$

(Verfaillie and Bidwell, 2000). The eco-efficiency is calculated using absolute values for the product value and environmental influence.

The two most important applications for eco-efficiency are as an internal tool for measuring progress, and for internal and external communication of economic and environmental performance (see WBCSD, 2005 for examples). The use of eco-efficiency indicators solves the problem that ‘traditional’ environmental performance indicators might fluctuate as a result of changes in production volume and thus hide real changes in environmental performance.

2.3. Environmental and value performance in the extended supply chain

In this paper, the terms environmental performance and value performance are, respectively, used for the numerator and denominator in the eco-efficiency ratio.

Ideally, the performance in an ESC should be accurately measured in each segment of the chain. This is not feasible without a disproportionately large effort since some of the companies involved will have insufficient environmental accounting. Assessing the performance will sometimes be impossible since the end-of-life treatment depends on where the dismantling takes place and cannot be known in advance. Another issue is that some of the materials used, such as steel, are bought from different smelting plants as prices fluctuate.

There is a need for standardised methods and it has been found useful to use Life Cycle Assessment according to the ISO 14040-standards to assess environmental performance (International Organisation for Standardization (ISO) 2000). Due to the problems mentioned, generic values must be used to some extent. As mentioned, Life Cycle Costs are used to assess value performance.

The WBCSD (Verfaillie and Bidwell, 2000) recommends five generally applicable indicators for measuring and reporting environmental performance and two additional indicators that are assumed to become generally applicable when standardised measuring methods are developed. These indicators are presented in Table 1. The WBCSD recommends these as site-specific indicators, but in this paper the same indicators are used in ESCs as well.

This list of indicators must not be regarded as complete. The WBCSD points out that all companies have to identify which environmental aspects are most important for their activities and products and ensure that these are included. The situation in an ESC is similar. LCA is used to obtain data from the ESC and segments of the chain when necessary.

While LCA is an established method to assess environmental performance in an ESC, no established method exists for assessing value performance. Value is not an objective term and Christopher (1998) claims that a product has no value at all before it has reached the customer in the condition and within the time limits that are requested.

Even though Life Cycle Costs (LCCs) is not an established method and Schmidt (2003), for example, warns against the large uncertainties, LCC is still chosen as a measure for value performance. There is ongoing work to standardise LCC

Table 1
Environmental performance indicators recommended by WBCSD (from Verfaillie and Bidwell, 2000)

Generally applicable indicators	Energy consumption Materials consumption Water consumption Greenhouse gas emissions Ozone depleting substance emissions
Future generally applicable indicators	Acidification emissions to air Total waste

(Rebitzer and Hunkeler, 2004) and monetary indicators are easy to understand and are recommended by the WBCSD (Verfaillie and Bidwell, 2000) and the Global Reporting Initiative (Global Reporting Initiative (GRI) 2002). LCC is here defined as the cumulative costs of a product over its life cycle (IEC, 1996), i.e. it is the total cost of buying, using and getting rid of a product, which is of great interest for the user of the products.

The value performance in a segment of the supply chain is measured as net sales, i.e. the value of sales less the cost of all inputs (goods, energy and services) purchased from sub-suppliers. This gives a measure of the costs added to the product in the assessed segment.

3. Materials and methods

3.1. Case study on furniture production

The case study is an integrated part of a research project called 'Productivity 2005—Industrial Ecology' at the Norwegian University of Science and Technology (NTNU). The main project objective is 'To raise the level of expertise at NTNU, and disseminate knowledge on product, production and recycling systems through research and networking in such a way that the Norwegian manufacturing industry has access to candidates, expertise and methodology that will help companies implement more eco-effective and competitive solutions in such systems' (Brattebø and Hanssen, 2000).

Two chairs (Chair A and Chair B) are used as examples; both are primarily used in meeting rooms, waiting rooms and cafeterias. For Chair A, five different models are analysed (I, II, III, IV and V). Model IV is the most sold and is the core model in the analysis. The weight is 6.81 kg of which steel (primarily in frames) constitutes 1.92 kg and beech plywood 3.54 kg. Chair B has a total weight of 4.47 kg of which steel constitutes 2.10 kg and beech plywood 2.30 kg.

The models are available with different varnishing, different types of fabrics and finishing. The variants chosen are as similar as possible. However, Chair A I and Chair B are not upholstered with polyurethane foam and fabrics like the other models and are thus somewhat simpler. It is assumed that all models have the same function and the same durability.

A more detailed description of the case is available in Fet et al. (2003).

3.2. System view in case study

In the case study, the products and their life cycles are the objects in the analysis and constitute the systems. The different components (arm rests, feet, back, etc.) constitute the sub-systems and the system elements are the different materials in the products (see Fig. 1). Life cycles can be identified at all system levels. In the case study, the length of the life cycles are equal, but in other cases maintenance and use of spare parts could result in different lengths of the life cycles within a system.

This system view makes it possible to develop databases. In time, companies will have data on environmental and value performance for the different materials and components, which will ease the analysis of new products. A drawback is the minor focus on the processes in manufacturing, use and end-of-life treatment. However, they must be included hierarchically in such a way that the processes necessary for producing material *i* are included in this system element. All processes necessary for producing component *j* out of the different materials, are included in this sub-system.

3.3. Environmental performance

The LCA studies are done using the LCA software GaBi 3v2 (Dahlsrud et al., 2002a,b). The LCA data for models I, II, III and V of Chair A are calculated based on data from model IV.

A 'cut-off' of 5% of the material stream is used in the LCAs, which means inputs constituting less than 5% of the total mass input in a process are generally omitted. Varnish and adhesives are included due to their toxicity potential. Particularly valuable materials could be included the same way, but this was not relevant. As a result, 94.0% of the total amount of materials is included for Chair A and 98.4% for Chair B. Transport of these materials is included.

Only the use phase of the production equipment and facilities is included since this normally is the dominating phase for energy consuming equipment (i.e. Fet et al., 2000; Funazaki and Taneda, 2001). More details on the specific system boundaries are available in the reports from the LCA studies (Dahlsrud et al., 2002a,b).

It is concluded that four environmental aspects dominate the overall environmental performance (Fet et al., 2003), namely

- global warming potential
- photochemical oxidation potential
- [emissions of] heavy metals (EI95)¹
- acidification potential

It was decided to use nine environmental performance indicators to meet the recommendations from the WBCSD and ensure that the significant environmental aspects are included. The seven indicators recommended by the WBCSD are used (see Table 1), and in addition 'emissions of heavy metals' and 'emissions of photochemical oxidising substances' are included.

A preliminary weighting procedure including four of the suggested nine environmental performance indicators is used to calculate a single score for environmental performance. Absolute values are normalised according to pressure data for Western Europe (see Guinée, 2002, p. 386) since this is the main market for the products. The normalised values are weighted according to Norwegian political targets (Fet et al.,

¹ The aspect 'Emissions of heavy metals' is used as it is generated in the LCA-programme GaBi 3v2 and originates from the LCA-method 'Eco-indicator 95'.

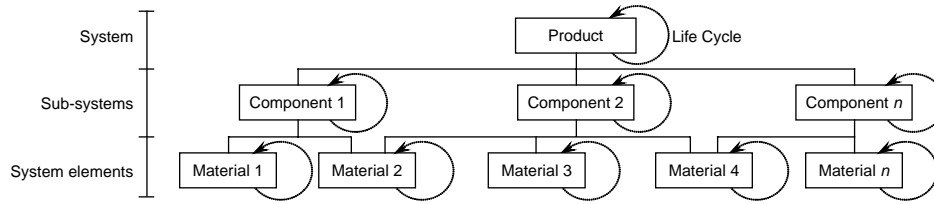


Fig. 1. Hierarchical system structure of the extended supply chains.

2000) where the weights are relative to the political targets for reduction and thus reflect a more restrictive policy. Normalisation values and weighting factors are given in Table 2.

Emissions of ozone depleting substances are not included in the selected weighting model (from Fet et al., 2000) and are hence not included in the single score. The LCA results also show this is of minor importance for the overall environmental performance in the case study (Dahlsrud et al., 2002a). Consumption of water, energy and materials and waste generation are omitted since there are no normalisation values available for these (Guinée, 2002). It is also not obvious that these can be included in a similar way since this can result in ‘double counting’—it is for instance not the energy consumption as such that is the problem, rather the environmental effects of energy production (e.g. emissions of greenhouse gasses which are already included).

The included indicators give information about the four environmental aspects that contribute significantly to the overall environmental performance. The aggregated environmental performance for product *p* is then calculated using the formula

$$\text{environmental performance}_p = \sum_{i=1}^n \frac{\text{absolute value indicator } i_p \times \text{weight}_i}{\text{normalisation value}_i}$$

where *n* is the number of performance indicators included.

3.4. Value performance

In the case study, the LCC of a product is defined as the price of the product (defined as recommended retail price minus taxes), the average costs in the use phase (cleaning, repair etc.) and the average disposal costs. Since the product with the lowest LCC is regarded as the most valuable, 1/LCC is used as a value performance indicator. The denomination is 1/NOK.²

3.5. Calculating eco-efficiency for the supply chain

In order to measure the eco-efficiency in a segment of the ESC (e.g. a single company), it might be useful to use the eco-efficiency ratio and calculate absolute values. Each of the suggested environmental performance indicators can be used combined with the added costs in the particular segment of the supply chain. This is how eco-efficiency normally is measured within a company today. Such indicators are primarily used for

internal measures and for measuring changes in internal eco-efficiency over time.

When using eco-efficiency indicators to compare products, it is best to avoid using a ratio and graphically present both environmental performance and value performance of the products relative to each other. This is in accordance with a method developed and used by BASF (Saling et al., 2002). The relative indicator value for indicator *k* for product *p* is calculated by the formula

$$\text{relative value } k_p = \frac{\text{absolute value } k_p}{\sum_{i=1}^n \text{absolute value } k_i} \times n$$

where *n* is the number of products in the analysis.

The results are presented in XY-diagrams divided in four quadrants (see Figs. 3–5). The products that are above average both in environmental and value performance are found in quadrant II. The products that are below average in both categories are found in quadrant IV. The products that are above average for value performance but below in

environmental performance are found in quadrant I, while this is the other way around in quadrant III. The distance from the plotted products to the diagonal in the figure indicates the absolute value for the eco-efficiency where the products above the line are the most eco-efficient, cf. the eco-efficiency ratio in Section 2.2.

The use of graphic presentation makes it superfluous to merge the value and environmental performance to one single indicator value, which is widely criticised (e.g. Azapagic and Perdan, 2000). Lafferty and Hovden (2002) state that there is

Table 2
Normalisation and weighting factors used in the aggregation process

Env. performance indicator	Normalisation factor	Weighting factor
Greenhouse gas emissions	4.73×10^{12}	0.99
Acidification emissions to air	2.74×10^{10}	1.35
Emissions of photochemical oxidising substances	8.24×10^9	1.50
Emissions of heavy metals ^a	7.57×10^{12}	10.00

^a Guinée (2002) does not use the impact category ‘heavy metals’. The value in Pb-equiv. given from GaBi is thus transferred to ‘human toxicity potential’ in accordance with the values given by Guinée (2002, p. 192); 1 kg Pb = 29 kg 1,4-DCB equiv.

² Norwegian kroner, 100 NOK ≈ 8€.

Table 3
Absolute values for environmental and value performance

Indicator		Chair A I	Chair A II	Chair A III	Chair A IV	Chair A V	Chair B	Unit
Environmental performance	Energy consumption	1109	1216	933	1265	869	901	MJ
	Materials consumption ^a	4.36	6.61	8.04	6.81	6.98	4.47	kg materials
	Ozone depleting substance emissions	1.24×10^{-6}	1.08×10^{-6}	1.41×10^{-6}	9.34×10^{-7}	1.50×10^{-6}	7.48×10^{-7}	kg R11-equiv.
	Water consumption	393	430	328	426	279	24	kg water
	Greenhouse gas emissions (100y)	23.2	33.8	29.3	35.7	27.3	30.1	kg CO ₂ -equiv.
	Acidification emissions to air	6.93×10^{-2}	6.23×10^{-2}	7.82×10^{-2}	5.94×10^{-2}	7.98×10^{-2}	8.87×10^{-3}	kg SO ₂ -equiv.
	Total waste	12.91	13.27	13.89	14.47	13.47	6.90	kg waste
	Emissions of photochemical oxidising substances	0.09	0.16	0.16	0.16	0.13	0.16	kg ethen-equiv.
	Emissions of heavy metals (EI95)	3.00×10^{-4}	3.00×10^{-4}	3.89×10^{-4}	2.59×10^{-4}	4.16×10^{-4}	3.02×10^{-4}	kg Pb-equiv.
	Aggregated indicator value	2.54×10^{-11}	3.96×10^{-11}	3.97×10^{-11}	4.01×10^{-11}	3.30×10^{-11}	3.66×10^{-11}	–
Value perf.	Life cycle cost	1093	1958	2123	2903	1989	1805	NOK
	1/life cycle cost	9.15×10^{-4}	5.11×10^{-4}	4.71×10^{-4}	3.44×10^{-4}	5.03×10^{-4}	5.54×10^{-4}	1/NOK

^a It has not been possible to get data on net material consumption in the supply chain. In the case studies only the total weight of products is used as a substitute.

often a real conflict between environmental and economic concerns and this should not be hidden in an eco-efficiency ratio.

4. Results

The values of the environmental and the value performance for the extended supply chains are given in Table 3. The relative contribution from sub-suppliers, end producer, use and dismantling for Chair A IV are shown in Fig. 2 for the four most important environmental aspects relative to the value performance.

All nine suggested indicators for environmental performance can be combined with the suggested indicator for value performance, giving a total of 10 eco-efficiency indicators when an aggregated value for environmental performance is included. Table 4 shows examples of the use of two of these indicators in absolute terms, namely the ratio between value performance and emissions of greenhouse gasses (in kg CO₂-equiv.) and emissions of heavy metals (in kg Pb-equiv.). This table also shows that Chair A I has the best performance in both aspects, even though it is only the second best with respect to emissions of heavy metals.

The same indicators are presented graphically by relative values in Figs. 3 and 4. Here, it is clearly displayed that the reason why Chair A I appears as the most eco-efficient product also with respect to emissions of heavy metals, is due to the value performance since the environmental performance as such is only slightly above average.

The correlation between price and environmental impact is tested to see if impaired environmental performance is simply a result of more complex, and hence more expensive, models. This is done with the SPSS 11.5 software. Kendall's tau-b

shows LCC is significantly correlated only to material consumption (correlation coefficient=0.733, $p=0.039$) and total waste (correlation coefficient=0.867, $p=0.015$). The results are similar if the Spearman rank correlation test is used. There is thus no clear correlation between environmental and value performance which emphasises the need to take both into account.

The environmental performance indicators are aggregated to a single score with the weighting procedure described in Section 3.3. The scores are shown in Table 3. The single scores for the different models are also transferred to relative values (Fig. 5).

As this figure shows, Chair A I appears to be the most eco-efficient model with the best environmental and value performance. At the other end, Chair A IV appear to be the

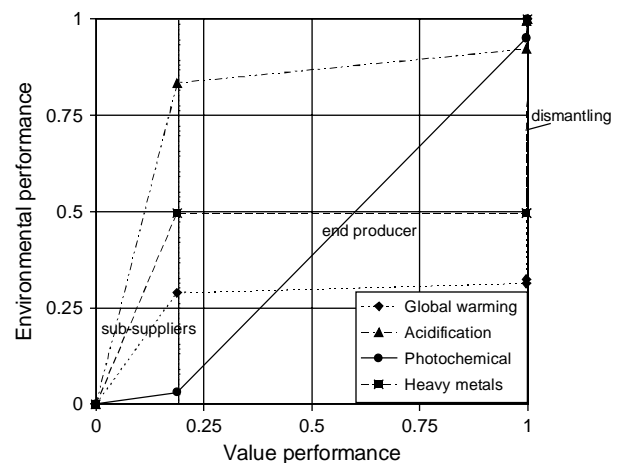


Fig. 2. Relative contribution of value performance and environmental performance from sub-suppliers, end producer, use and dismantling of Chair A IV.

Table 4
Eco-efficiency for different models

	Value perf. indicator	Environmental performance indicators		Eco-efficiency indicators	
	1/LCC (1/NOK)	Emissions of greenhouse gasses (kg CO ₂ -equiv.)	Emissions of heavy metals (kg Pb-equiv.)	(1/NOK)/kg CO ₂ -equiv.	(1/NOK)/kg Pb-equiv.
Chair A I	9.15×10^{-4}	23.2	3.00×10^{-4}	3.94×10^{-5}	3.05
Chair A II	5.11×10^{-4}	33.8	3.00×10^{-4}	1.51×10^{-5}	1.70
Chair A III	4.71×10^{-4}	29.3	3.89×10^{-4}	1.61×10^{-5}	1.21
Chair A IV	3.44×10^{-4}	35.7	2.59×10^{-4}	0.96×10^{-5}	1.33
Chair A V	5.03×10^{-4}	27.3	4.16×10^{-4}	1.84×10^{-5}	1.21
Chair B	5.54×10^{-4}	30.1	3.02×10^{-4}	1.84×10^{-5}	1.83

least eco-efficient model while there are minor differences between the four other models.

In addition to comparing existing products, the method can be used in scenario evaluation. It is possible to calculate indicator values for both future models and also for existing models where parts of the life cycle are altered, such as changed end-of-life treatment.

Fig. 6 shows the result of a scenario where the waste treatment for Chair A IV is altered. Since all indicator values are relative values, the inclusion of a new model or changes in the value(s) of one existing model, will result in new indicator values for all models. Instead of disposal in landfill, which is the normal end-of-life treatment, it is assumed that wood is incinerated for energy recovery. It is also assumed that this does not influence the value performance. As the figure shows, this improves the environmental performance significantly (here given by kg CO₂-equiv.) and the model shifts from being the worst to the best with respect to the environmental performance. However, Chair A I still appears to be the most eco-efficient model due to its better value performance.

5. Summary and discussion

This paper demonstrates how the eco-efficiency concept can be used for ESCs to compare both existing products and new ones. The method can be summarised in five steps:

- identify the systems and set system borders for existing or planned products
- select environmental performance indicator(s)
- select value performance indicator(s)
- assess performance
- display results

The models in the case study are significantly different with respect to both environmental and value performance. Understandable information about the performance is hence important to support decisions, such as when purchasing takes place or when the end producer evaluates their products. As shown in Fig. 6 there are also possibilities to explore the environmental and economic benefits of possible alterations of the ESC. Information could be presented with an aggregated value for the environmental performance as shown in Fig. 5 or with specific indicators as in Figs. 3 and 4 when this is preferable, such as when there is a particular emphasis on a particular environmental aspect.

Fig. 2 underlines the need to include the extended supply chain. For three out of four environmental aspects the main contributions to the environmental performance are not under the direct control of the end producer.

The weighting model used in the case study is immature and only four of the identified nine environmental performance indicators are included. The LCA analyses show that these are the most important ones, but this result is based on a range of other weighting procedures incorporated in GaBi 3v2. There

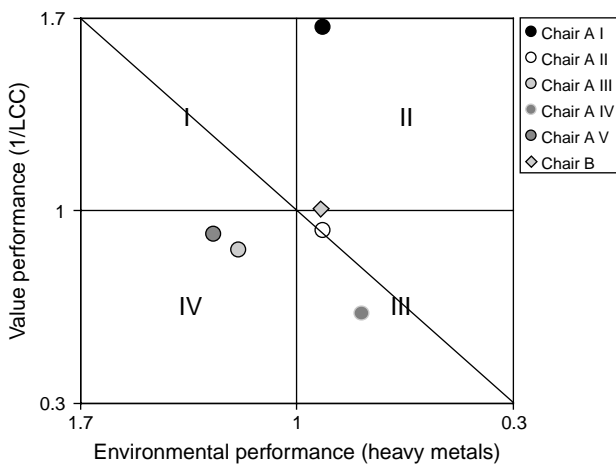


Fig. 3. Relative eco-efficiency with emissions of heavy metals as an indicator of environmental performance.

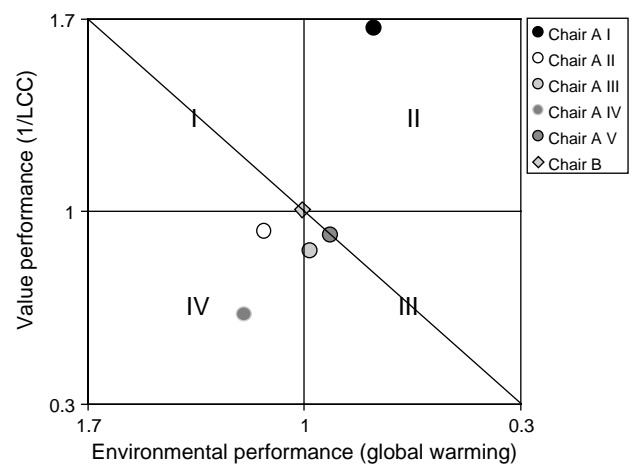


Fig. 4. Relative eco-efficiency with emissions of greenhouse gasses as an indicator of environmental performance.

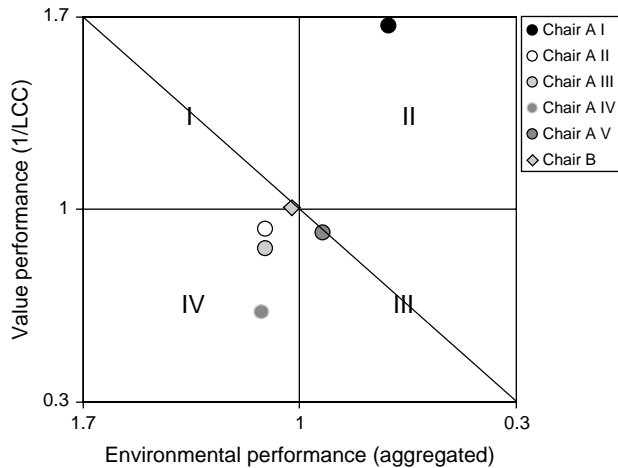


Fig. 5. Relative eco-efficiency with an aggregated value as an indicator of environmental performance.

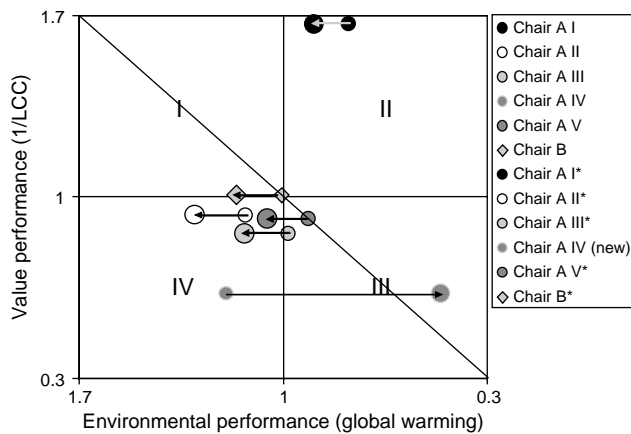


Fig. 6. Relative eco-efficiency and changes as a result of changes in the extended supply chain.

might therefore be a circular argument that indicates these four as the most important performance indicators. It is thus important to develop a weighting procedure that can be commonly accepted within the furniture industry to ensure that comparisons are made with an acceptable degree of certainty.

All comparative use of LCA data is questionable. Several subjective choices have to be made (e.g. Graedel, 1998; Hertwich et al., 2000) and in practical applications all requirements will not be fully met (Wrisberg and Udo de Haes, 2002). When it comes to LCC, the situation is not better (Schmidt, 2003) and here the work to standardise the methodology is incomplete (Rebitzer and Hunkeler, 2004). Indisputable results from comparing eco-efficiency for different extended supply chains will thus never be reached and the development of more standardised methods accepted within a business sector is necessary.

6. Conclusions

The results from the case study give information that could be useful, particularly as additional information in procurement

processes (Figs. 3–5) or in evaluating and improving existing ESC (Fig. 6). This would not be possible if eco-efficiency was used solely for companies and sites since valuable information would be lost. The chance of coming to a right decision thus increases if the presented method is applied.

This paper primarily presents results for extended supply chains. The identified indicators can, however, be used for segments as well, as shown in Fig. 2. The identified indicators therefore satisfy the need for indicators that are useable both for the extended supply chain and for the individual companies involved.

It is at present not possible to conclude that the suggested environmental performance indicators in the case study intercept all significant information concerning environmental aspects. Some of the proposed indicators may even be superfluous. It is, however, important to move towards a standard set of indicators. The indicators suggested by the WBCSD are thus used as the basis for the analysis.

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Appendix C

Eco-efficiency in redesigned extended supply chains; furniture as an example

Eco-efficiency in redesigned extended supply chains; furniture as an example

Ottar Michelsen

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Abstract

This paper shows how the eco-efficiency concept can be used to evaluate value and environmental performance when considering different scenarios for redesigning extended supply chains (ESCs). Results from a case study on furniture production in Norway are used to illustrate the concept.

An extended supply chain includes all processes necessary for production, use and end-of-life treatment of a product. The environmental performance of the products was assessed using LCA, and value performance was measured as life cycle cost. Instead of calculating absolute values using a traditional eco-efficiency ratio, relative values for different scenarios were calculated and presented graphically in an XY-diagram. This clearly visualises the alternatives that have the best environmental and value performance.

Six different scenarios were developed to assess how the performance of an existing ESC can be improved. The eco-efficiency for each scenario was compared with the present ESC. The results show that there is large and realistic potential for environmental improvements in the extended supply chain without an equivalent increase in life cycle costs.

Introduction

The growing concern for the environmental dimension of business strategy is resulting in a greater focus on environmental management (e.g. Porter and van der Linde 1995, Noci and Verganti 1999, Cramer 2000, Hall 2000, Ammenberg and Hjelm 2003, Banerjee et al. 2003, Hunkeler et al. 2004). More and more companies have also realised that this has consequences not only for the activities within the company, but for the entire supply chain (e.g. Lamming and Hampson 1996, Noci and Verganti 1999, Clift and Wright 2000).

The increased focus on environmental performance in companies has a manifold origin. Pressure from customers and legislation has often been identified as the two most important drivers (e.g. Florida 1996, Noci and Verganti 1999, Cramer 2000). Several companies are striving to stay ahead of legislation and competitors, in order to avoid more or less ad hoc interventions later on (Lamming and Hampson 1996), or to be able to influence future legislation in a way that would give them a competitive advantage (Barrett 1991, Taylor 1992). Expectations of cost savings are also an important factor, and environmentally proactive companies tend to have greater innovative power than other companies (Sharma and Vredenburg 1998, Noci and Verganti 1999).

The growing interest in environmental issues does not only influence the end producers. According to Noci and Verganti (1999) and Hall (2000), awareness and pressure from regulations and customers move upstream along the supply chain and accumulate. Environmental improvements in supply chains are thus attainable through a market-

driven process if the end producers include applying environmental performance criteria when selecting suppliers. It is therefore necessary to ask sub-suppliers to meet not only product-oriented purchasing specifications (e.g. cost and quality requirements), but also specifications for environmental performance in the production process (Hall 2000).

To comply with increased requirements from customers and authorities, it is necessary for companies to be aware of the performance of their products throughout their life cycle. One possibility is to measure eco-efficiency in the extended supply chains (ESC). Michelsen et al. (2006) have demonstrated how this approach can be used to compare different products in terms of environmental performance and costs over the life cycle of the products.

The purpose of this paper is to show that eco-efficiency can also be used to assess environmental and value performance when an ESC is redesigned in different ways. This is demonstrated by means of a case study of furniture production. Different scenarios for redesigning the present ESC of a chair have been developed and analysed to quantify the changes in environmental performance within the different scenarios, and their economic consequences.

Redesigning extended supply chains

When products are analysed to reveal possible eco-efficiency improvements, the extended supply chain should be included. Christopher (1998) defines a supply chain as 'the network of organisations that are involved, through upstream and downstream linkages, in the different processes and activities that produce value in the form of products and services in the hand of the ultimate consumer.' An extended supply chain also includes the use and disposal of the products. The term extended supply chain encompasses both the companies involved and the life cycle perspective. Clift and Wright (2000) and Clift (2003) found significant differences in the ratio between environmental impact and added value in different segments of manufacturing processes. Michelsen et al. (2006) have shown the same for furniture, and revealed that a major part of the environmental impact of the products originated not from the end producer but elsewhere in the ESC. Management of the ESC goes beyond what is normally recognised as supply chain management, as it also includes end-of-life treatment. The ESC is, in principle, infinite, and criteria must be defined for the selection of boundaries. Figure 1 shows a simplified picture of the ESC in the present case study, in which the system elements are the components of a chair.

Companies must be able to identify where improvements are possible in the ESC and what impacts these will have on environmental and economic performance. Michelsen et al. (2006) have shown how this could be done by using eco-efficiency. The environmental performance of the ESC is the aggregated environmental impact from all processes in the life cycle of the product, which is assessed using LCA. The value performance of the ESC is the life cycle costs (LCC) of the product, where LCC is defined as the cumulative costs over the life cycle from the users' point of view (cf. IEC 1996). The LCC of a product is thus the price of the product (defined as recommended retail price minus taxes), the average costs in the use phase (cleaning, repair etc.) and the average costs of end-of-life treatment. At present, there is no consensus on how LCC should be defined (Schmidt 2003), but in the present paper, it only includes the actual costs born by the user. This is motivated by the fact that all official bodies in Norway, as in some other countries in Europe, have a legal obligation to take this into consideration when new acquisitions are planned.

When measuring eco-efficiency in ESCs, all scores are compared with a point of reference. This could be an average value for all ESCs that are analysed, or the value for one particular ESC. The data are then presented graphically in XY-diagrams (see Figure 2) without merging the value and environmental performances into one single indicator,

as is often done in eco-efficiency calculations. This type of data presentation has also been used by others, e.g. in the 'Basel Eco-Controlling Concept' (Schaltegger and Sturm 1998) and at BASF (Saling et al. 2002). If the values are presented as relative values, it is possible to omit everything that is equal in all ESCs and thus simplify the analysis and reduce the uncertainties.

These graphic presentations of eco-efficiency are used to compare different ESCs. However, carrying out improvements requires a more detailed study of the segments in the ESCs. This is done by comparing environmental impact and added costs for the different segments of the ESCs.

Michelsen et al. (2006) used eco-efficiency in ESCs to compare the performance of existing products. However, the same approach can also be used to analyse scenarios in which present ESCs are redesigned to see how this affects their eco-efficiency performance. After a full assessment of a product, different scenarios can be developed, based on the following questions:

- Is it possible to change the materials or the amounts of materials used in the product?
- Is it possible to change the production processes?
- Is it possible to change the product's use?
- Is it possible to change the product's end-of-life treatment?

After potential scenarios for redesign have been identified, these are analysed like any other ESC and compared with the original product. Environmentally and economically viable new solutions are thus identified and the end producer can use this information to redesign the ESC. This does, however, presuppose that they have sufficient power in the supply chain and/or are ready to take responsibility for a larger part of the product's life cycle.

Case description

The furniture industry is no exception when it comes to the increasing interest in environmental performance. There has particularly been a focus on greater producer responsibility and the possibilities of introducing take-back legislation. In Norway, take-back of furniture was explicitly mentioned in a white paper on environmental policy (Ministry of the Environment 1999). It has also been reported that companies can gain a competitive advantage through their environmental profile (Dahl et al. 2002).

Partly as a consequence of such prospects, furniture industries in several countries have conducted studies to identify opportunities for environmental improvements and evaluate the effects of take-back legislation (e.g. Jaakko Pöyry Infra 2001, Vassbotn and Bjerke 2001, Saft et al. 2003). These studies offer some useful information about ideas prevalent in the industry sector and the findings of preliminary studies, but they were not written in English and as a consequence are poorly accessible.

A paper by Michelsen et al. (2006) compared the eco-efficiency of several chairs designed to be used in conference rooms. The chairs are made by two different manufacturers, and it was found that the flagship model from one of them had the lowest eco-efficiency of all of the models analysed. There was thus an obvious need to improve this model's performance. Therefore we decided to develop different scenarios and assess them to see if it is possible to improve the environmental performance of the chair without increasing the costs. The flagship model has a total weight of 6.81 kg. Table 1 shows the main components of the chair. In addition, 3 kg cardboard is used for packaging. Figure 1 shows the main components and materials used in the chair.

Table 1 – Main components of the chair used in the case study

Component	Weight
Steel frame	1.92 kg
Beech plywood	3.54 kg
Beech	0.44 kg
Polyurethane (PUR)	0.65 kg
Other	0.56 kg

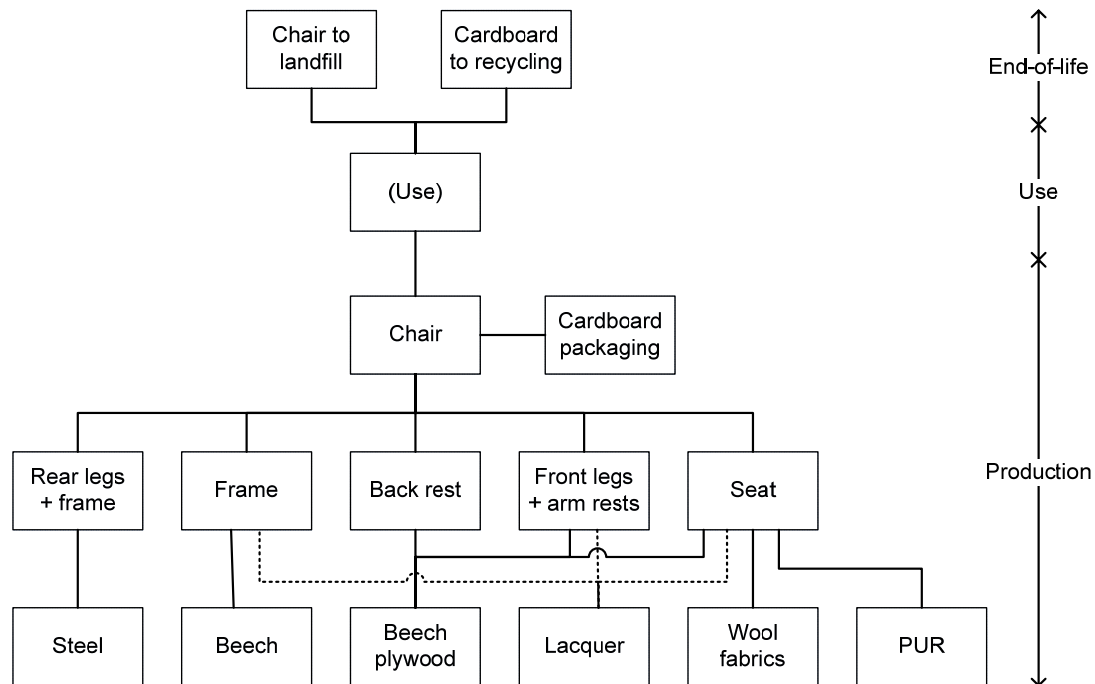


Figure 1 – Main elements in the extended supply chain of the chair used in the case study

The environmental performance of the ESC was assessed using SimaPro 5.1, selecting Eco-indicator 99 (E)/Europe EI 99 E/E as the impact assessment method. Data on raw materials production were largely based on database values. Transport and energy consumption were included, but waste handling, both by the producer and by suppliers, were included only occasionally. It was assumed that the proportion of recycled steel in the production is 23%. Raw materials for the production of lacquer and plywood adhesive were not included. Nor was the production of raw materials for wool fabrics included, due to lack of appropriate data. Cardboard packaging was assumed to be produced with 100% recycled fibres.

As regards waste handling, database values were used for landfill for all materials except wood. Emission values for wood were taken from Sandgren et al. (1996). According to Vassbotn and Bjerke (2001), landfill is the most likely waste scenario for furniture in Norway.

Land use for transport, beech production or production facilities was not included. In cases where this had been included in database values for different processes, its impact was excluded from the analysis.

In the original case, this yielded an environmental impact of 2030 mPts for the life cycle of the chair. The environmental impact was also calculated with other impact assessment methods (Eco-indicator 99 (H/H), Eco-indicator 99 (I/I), CML 2 baseline 2000 and EPS 2000) integrated in SimaPro 5.1, to check if the choice of impact assessment method had a large impact on the final results.

The life cycle cost is the sum of the price of the product, the expected costs during use and the average costs for disposal or other end-of-life treatment. The producer uses the following equation to calculate the recommended retail price:

$$\frac{(LC + PC) \times 1.15}{0.7} \times k \quad (1)$$

where LC stands for labour costs in production and PC for purchasing costs. This is multiplied by 1.15 to include indirect costs and divided by 0.7 to include the desired margin for the company. The factor k represents the costs and margins for transport and retail. The recommended retail price in 2003 was 2894 Norwegian kroner (NOK)¹.

Costs during use could be related to cleaning and repair. The present case study assumed that there are no costs related to such activities. We also assumed that the chairs are disposed of at a landfill (cf. Vassbotn and Bjerke 2001). In this case, the costs of delivery to a landfill in Oslo were used as disposal costs. At the time of writing, this was NOK 1422 per tonne (taxes not included) (Oslo kommune – Renovasjonsetaten 2004), including transport.

Six different scenarios for changes to the extended supply chain were developed. For the time being, these were limited to changes in materials used (scenarios A–C) and changes to the end-of-life treatment (D–E). Scenarios from these two groups can be combined, as exemplified by one scenario (BE). It is possible to develop scenarios that include alterations to production and assembly processes, but this was beyond the scope of the present study. It was also not considered useful to assess changes in the use of the product, since its contribution to both environmental performance and costs is insignificant (Michelsen et al. 2006).

We did not develop any scenarios that include changes to the amount of plywood, due to the lack of reliable data, especially on the land use impact of forestry. The LCA results indicate that alterations to the wood/plywood content could change the environmental performance significantly. Future work will include the impact of wood components including land use assessment, and a methodology to include land use in forestry is under development (Michelsen 2004).

Scenario A

In this scenario, the use of polyurethane is reduced by 20%. According to the producer of the chair, such a reduction should be possible without reducing the chair's comfort significantly. It is not assumed that this has any impact on the costs, since the reduction will only result in an insignificant decrease in the purchase price of the extruded foam.

Scenario B

In this scenario, polyurethane is partly replaced by an innovative material called Maderon. According to Diaz and Redondo (2002), it is possible to reduce the amount of polyether polyols by 30%, replacing them with cellulose, as well as to reduce the amount of toluene diisocyanate by 35%, replacing it by silicate, in the production of the foam. The environmental performance was estimated based on the alterations to the production phase described by Diaz and Redondo (2002).

The price of the product is not known, but the alteration to the LCC was calculated both on the assumption that the compound is twice as expensive as traditional polyurethane (scenario B) and on the assumption that it is 50% more expensive (scenario B*).

¹ 1€ ≈ 7.90 NOK (August 2005)

Scenario C

In this scenario, the upholstery is completely omitted. Both polyurethane and fabrics used on the seat are excluded. As a consequence, more lacquer is needed to get an appropriate finish on the seat. The major drawback of this scenario is that it results in reduced comfort and can hence not directly replace the original product.

Scenario D

In this scenario, the chair is dismantled after the use phase. It is assumed that the chair is transported to a dismantling facility close to the user and that this causes no extra emissions from transport and no extra transport costs compared to the present situation (transport to landfill). This could be realistic if the furniture industry had a common dismantling facility and costs and transport due to traditional waste collection were avoided.

It is assumed that the dismantling takes 5 minutes (Vassbotn and Bjerke 2001), and another 5 minutes are added to cover the time used in collection and treatment before the dismantling actually takes place. Labour costs are assumed to be at the same level as those used by the chair's manufacturer. After dismantling, it is assumed that steel is delivered for recycling and the wood for incineration in modern incineration facilities with energy recovery.

We calculated two different cost alternatives. In the first alternative (scenario D), the extra labour costs were included like any other labour cost, as shown in Equation 1. In the second alternative, it was assumed that the dismantling would be done as a non-profit activity, with no margin for the dismantler included (scenario D n-p). This was calculated using the following equation:

$$\frac{(LC + PC) \times 1.15}{0.7} \times k + (aLC + aPC) \times 1.15 \quad (2)$$

where aLC stands for the additional labour costs for the dismantling effort and aPC stands for additional purchasing costs (not relevant in this scenario). This presupposes that the work in the dismantling facility is as efficient as that at the end producer's and carries the same level of indirect costs, which again presupposes that large numbers of items are dismantled.

Scenario E

In this scenario, a take-back system is introduced. This scenario assumes that it is possible to collect 80% of the chairs after the use phase. The dismantling time and costs are similar to those in the previous scenario. The cost of the return transport was estimated based on information from Norcarg (2004), on the assumption that 10–20 chairs are transported together. After dismantling, 50% of the steel components are reused in new products, while the rest of the steel is delivered for recycling. Hence, there is an average need for 0.6 steel frames for one new chair, which reduces the purchasing costs and the environmental impact from the production of the steel frames. The rest of the waste treatment takes place according to the original situation.

In the same way as in scenario D, two different cost alternatives were calculated. The first alternative (scenario E) included the extra labour costs like any other labour cost, as shown in Equation 1, and extra transport is included as purchasing costs. In the second alternative (scenario E n-p), it was assumed that the dismantling and extra transport is done as a non-profit activity and included as in Equation 2.

Scenario BE

This scenario is a combination of scenarios B and E and is thus a scenario where both production and end-of-life treatment are altered. In calculating the LCC, it was assumed

that Maderon is twice as expensive as polyurethane. Both cost alternatives from scenario E were included.

Results

The changes in value and environmental performance for the different scenarios are shown in Table 2. The same values are presented graphically in Figure 2.

Table 2 – Changes in environmental and value performances in the scenarios

Scenario	Δ mPt	Δ NOK	Δ NOK (n-p)
A – reduction of PUR	- 30	0	-
B – use of Maderon	- 50	130	B*: 64
C – exclusion of PUR	- 240	- 144	-
D – dismantling and recycling	- 330	130	33
E – take-back and reuse	- 280	94	-142
BE – combination	- 330	224	-12

All scenarios gave an improved environmental performance, ranging from -30 mPts in scenario A to -330 mPts in scenarios D and BE. It is also clear that of these scenarios, alterations to end-of-life treatment had a greater impact on environmental performance than the proposed alterations to the materials used.

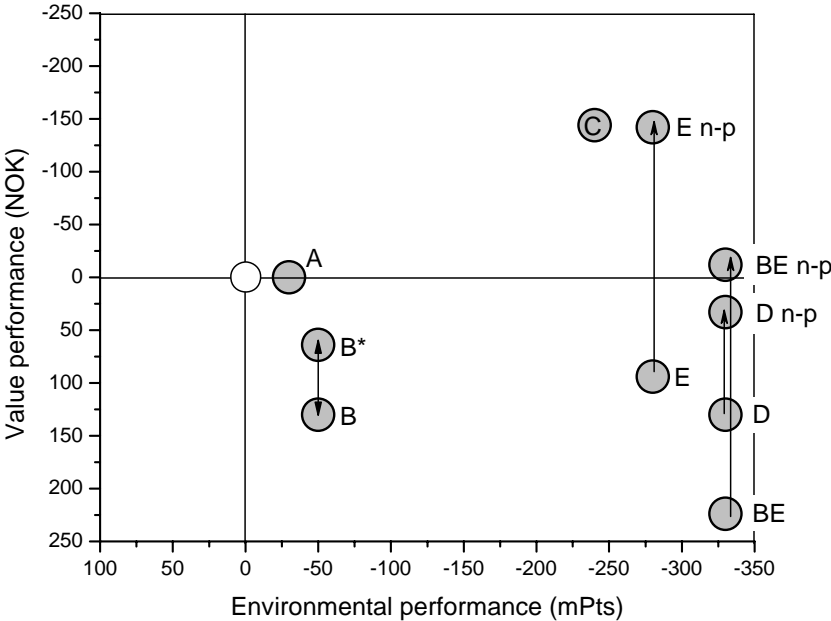


Figure 2 – Changes in eco-efficiency in the different scenarios (see text for details)

The only scenario giving an unequivocal improvement in value performance was scenario C, which unfortunately involves reduced seating comfort. However, scenarios E and BE also yielded an improved value performance when the dismantling and recycling activities were introduced as non-profit activities.

The relative costs of the various alternatives for environmental improvement differed considerably. This is shown in Table 3, where positive values indicate the cost in NOK of a reduction in mPts, while a negative value indicates cost reduction. The use of Maderon (B) was by far the most expensive way of improving the environmental performance, even when a lower cost alternative was used. Unsurprisingly, the exclusion of polyurethane and fabrics (C) was the most cost-efficient alternative to improve the environmental performance. Of the scenarios not involving reduced seating comfort, the

introduction of a take-back system (E) led to a slightly better performance than dismantling for recovery (D), and as already pointed out, a take-back system also has a potential for cost savings if the extra costs are included as non-profit activities (Equation 2).

The picture was more or less the same for the other impact assessment methods we applied. Using EPS 2000 and CML 2, the alterations appeared as greater improvements, giving an environmental impact reduction of more than 24% in scenario D. The only diverging result was that obtained by using Eco-indicator 99 (H). Here, scenarios A, B and C followed the same trend, but scenarios D and E only resulted in about half the reduction of environmental impact compared to scenario C. In addition, scenario E was now slightly better than scenario D.

Table 3 – Cost-efficiency of environmental improvements in the scenarios

Scenario	NOK/mPt
C – exclusion of PUR	-0.60
E – take-back and reuse (non-profit)	-0.51
BE – combination (non-profit)	-0.04
A – reduction of PUR	-
D – dismantling and recycling (non-profit)	0.10
E – take-back and reuse	0.34
D – dismantling and recycling	0.39
BE – combination	0.68
B – use of Maderon (lower cost alternative)	1.28
B – use of Maderon	2.60

Discussion and conclusions

Traditionally, the purpose of eco-efficiency has been to maximise value creation with minimised use of resources and emissions of pollutants (Verfaillie and Bidwell 2000). However, the combination of value and environmental performances in one single indicator has been criticised, since in many cases this obscures conflicting interests with respect to environmental and value performances (e.g. Azapagic and Perdan 2000, Lafferty and Hovden 2002). Alternative solutions with a high eco-efficiency score might simply not be economically viable. This problem is avoided when the eco-efficiency is presented as in Figure 2, since both environmental and value performances are presented as they are.

Previous studies have shown that graphic presentations in XY-diagrams are useful for comparing existing products (Schaltegger and Sturm 1998, Saling et al. 2002, Michelsen et al. 2006) and that companies can use the information to evaluate the present performance of their products. The present paper demonstrates the possibility to compare existing products with scenarios for redesigned ESCs. The case study presented above shows the value of expanding the use of eco-efficiency. The results and the way they are presented give companies valuable information in their search for opportunities to improve the ESCs and to assess in what part of the ESCs the improvements should take place.

The results and the graphic presentation are easily understandable for non-specialists. The value performance is expressed as overall costs, which is a familiar measure. No externalities are included. Environmental performance is presented as a single score, which makes it easy to understand even for those unfamiliar with LCA. The graphic presentation clearly visualises which products have the best environmental and value performances. When the graphic presentation is used for different scenarios, as in the above case study, it is also easy to see any improvements. A top-level manager or a purchaser could easily see the range of environmental improvements and the resulting costs or cost reductions.

As in all studies involving LCA, especially those involving comparisons, the quality of the data is critical. In the case study presented here, SimaPro was used to ensure a standardised approach, particularly with respect to normalisation and weighting. However, the use of different impact assessment methods reveals that this actually influences the final results, and there is thus an obvious need for standardised methods within an industry sector if the method used here is to be employed to compare products from different producers (Michelsen et al. 2006). An advantage of the case study presented here is that it used relative values, making it possible to omit data for processes present in all cases. This reduces the uncertainty of the results.

The value performance scores have large uncertainties. We have used the companies' own method of calculating costs, but it is hard to take all eventualities into consideration. The costs of dismantling facilities, for instance, greatly depend on the numbers of items that are dismantled. Costs of reverse logistics are also hardly available. Such costs might be as much as 9 times the costs of delivering the product to the consumer (Persson and Virum 1995), but in scenario E it is assumed that the transport is carried out by a transport company on a case-by-case order. It should hence be possible to reduce the costs in a real situation.

The results of the case study indicate a potential for significant improvements to the current situation, primarily by changing the end-of-life treatment for the chair. While dismantling for recycling yields the greatest environmental improvement, the additional introduction of a take-back system offers opportunities for improved value performance. A take-back system is also a more cost-efficient way of reducing the environmental impacts (Table 3). According to Clendenin (1997), Xerox has introduced such systems, for economic reasons. In the case presented here, eventual economic improvements presuppose that extra costs are included as non-profit activities. Clendenin (1997) emphasised the fact that few companies have explored the opportunities for systematic reuse of components, which might explain the apparently low profitability.

Communication with representatives from the industry reveals that there is no common opinion on this subject. There seems to be a tendency for the majority to think that take-back legislation and component reuse is unsuitable, since furniture has a relatively long life expectancy, and models are changed before components are ready for reuse. The idea of component reuse is nevertheless being seriously considered in at least one company.

The results strongly indicate that authorities should consider giving the furniture industry a statutory responsibility for end-of-life treatment. Porter and van der Linde (1995), van den Akker (2000) and Bleischwitz (2003) recommended that authorities should impose requirements for improvements, but that industry should be allowed to find out how to meet them. This is in accordance with the targets for end-of-life treatment for cars, where an EU directive (2000/53/EF) makes no distinction between reuse and recycling. An increased responsibility for the end-of-life treatment also increases the opportunities to address harmful substances. In furniture, this would particularly include brominated flame retardants (Statistics Norway 2003).

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Appendix D

Investigation of relationships in a supply chain in order to improve environmental performance

The original publication is available at www.springerlink.com

Investigation of relationships in a supply chain in order to improve environmental performance

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Accepted for publication in Clean Technologies and Environmental Policy

Abstract

This paper presents a methodology to combine environmental assessment information and knowledge on supplier relationships. The work is based on a case study of production of a chair. The methods used are shown to be effective to reduce the number of suppliers that should be managed from an environmental point of view and also obtain an overview over which suppliers that can be influenced. It is also shown how suppliers with the presumably highest potential for improvements are identified. The end-producer can thus obtain control over most of the environmental impact originating from upstream activities through a limited number of suppliers. In the case study the number of suppliers that should be managed is reduced to 3. Small and medium sized enterprises have often limited possibilities to influence the suppliers, but in this paper it is demonstrated how this problem can be overcome by cooperative purchasing within a branch of industry with a common demand for information from the suppliers.

Keywords: environmental performance, supply chain management, channel power, furniture

Introduction

The focus on the environmental performance of products is increasing. An increasing number of manufacturers are carrying out life cycle assessments (LCA) of their products and also presenting the results in environmental product declarations (EPD).

This growth can be explained partly by an increased request for information from the market, but also an increased legal pressure to provide environmental information about the performance of products. In Norway this is clearly stated in the Public Procurement Act¹ that says that environmental performance of products must be considered before public procurement take place. Further, the Environmental Information Act² states that information on environmental performance should be available. Similar legal framework exists in other countries as well.

However, the end-producers are in many cases directly responsible for only a minor part of the environmental impact caused during the life cycle of the products. From a case study of furniture production in Norway, it was revealed that the end-producer only contributed significantly to emissions of photochemical oxidising substances and none of the other environmental impact categories investigated (Michelsen et al. 2006). Thus, the end-producers need information about the performance of the other actors in the supply chains to be able to document the performance of their products. To be able to improve the performance, the end-producers must also have the possibility to influence the actors in the supply chains, or, if needed, change to suppliers with better performance.

¹ <http://odin.dep.no/fad/norsk/tema/offentlig/p10002770/024081-990048/dok-bn.html>

² <http://odin.dep.no/md/engelsk/regelverk/lover/022051-200017/dok-bn.html>

The pressure to provide environmental information and carry out improvements is unevenly distributed in a supply chain. The end-producers are in general more exposed than their suppliers (Hall 2000), and an important task is then to disperse the focus on environmental performance to other parts of the supply chain. One way of doing this is to integrate environmental and supply chain decisions (Handfield et al. 2005).

This paper is based on a case study of production of a chair. The data from this study are used to test the suitability of different methods that the end-producer can use to identify which suppliers that should be more actively managed in order to improve the environmental performance of their products. This is based on both the contribution to the overall environmental performance of the product and the end-producer's possibilities to influence the suppliers.

SMEs and supply chain knowledge

With the increasing demand for environmental information on the product level, it is obvious that manufacturers need information about the performance of their suppliers. Generic data can to some extent be used, but this is inconsistent with the requirement in the ISO standards on LCA (ISO 14040-43). It further makes it difficult to differentiate between products since they all will have an average score.

Considerable research has shown that companies increasingly rely on their suppliers for competitive success (e.g. Hahn et al. 1990; Lambert and Cooper 2000) and the increasing dependency on the environmental performance of the suppliers is an expansion of this situation. It is important to realize that when a supplier is selected, not only the requested item is delivered. The waste and emissions created during the production, and the contribution to waste and emissions during use and end-of-life treatment of the final product, are also delivered into the performance.

However, the companies network horizon is often rather narrow (Håkansson and Johanson 1992; Lambert and Cooper 2000; Holmen and Pedersen 2003). Even if they require information on environmental performance from the actors they know, this will in most cases be a limited part of the supply chain. In most cases it is neither possible nor practical to have too much knowledge about a large part of the supply chain (Håkansson and Snehota 1995) and the end-producer should rather disperse the demands on environmental performance through their suppliers. In general, the most severe environmental impact originates in the early stages of the supply chain and especially during extraction of raw materials (Clift and Wright 2000) and thus far away from the focus of the manufacturers that provide the final products for the market. It is thus not sufficient to ask for information on environmental performance of first tier suppliers, it is also necessary to get information from their suppliers and sub-suppliers.

Buyer–supplier relations play an increasingly important role in the strategies of firms, also when it comes to environmental performance (Handfield et al. 1997; Hall 2000; Handfield et al. 2005). Some companies are deliberately selecting suppliers that exceed environmental regulatory requirements and are able to disperse the focus on environmental performance to their suppliers again (Handfield et al. 2005). This gives a cascade effect in the supply chain.

However, in most cases lower profile suppliers lack incentives to improve their environmental performance (Hall 2000). If a buying company is not able to motivate their suppliers to improve, they must be able to force them. Improvements must then be initiated by a channel leader with sufficient channel power. This is defined as the ability of one channel member to control the decisions of another (El-Ansary and Stern 1972). For many SMEs this will be challenging since they themselves have low leverage power since they in many cases will be minor customers of their suppliers.

A possible solution to this dilemma is to form purchasing consortia, or in other ways take part in different variations of cooperative purchasing. Consortium purchasing is horizontal cooperation between independent organizations that pool their purchases in order to achieve various benefits (Tella and Virolainen 2005 - Figure 1). The motivation is to achieve a stronger bargaining position and this is often a successful strategy (Laing and Cotton 1997; Doucette 1997; Zentes and Swoboda 2000; Kamann et al. 2004; Tella and Virolainen 2005). Primarily, the motivation behind these consortia is financial gains, but among others, Laing and Cotton (1997) and Kamann et al. (2004) focus on the dependency on common needs to succeed with such consortia. These needs are not restricted to financial issues, but might as well be related to information of the supply market (Tella and Virolainen 2005) and is thus relevant also for environmental performance. When a business sector is exposed to increased pressure to offer environmental information on products, a common strategy on achieving information from their suppliers might then be a possible solution. Such a strategy is reported from the woodworking industry in UK (Kogg 2003).

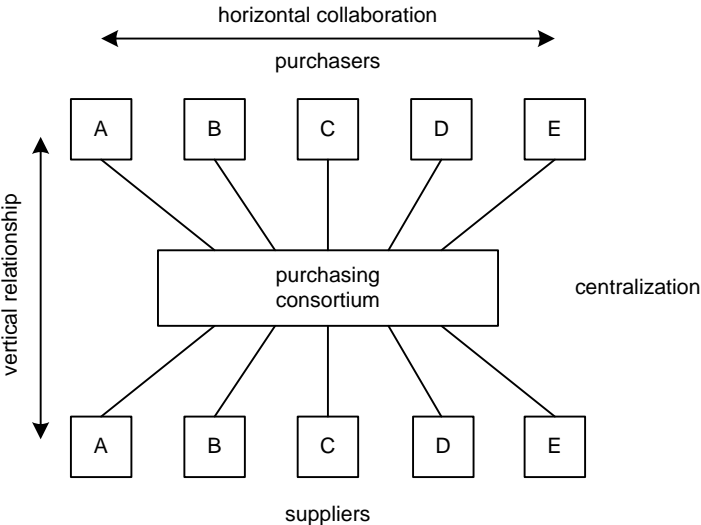


Figure 1 - Theoretical framework of cooperative purchasing (from Tella and Virolainen 2005)

It is unlikely that a company, especially SMEs, will be able to manage the entire supply chain, so some priorities must be set (Lambert and Cooper 2000). One possibility is to classify the suppliers after their contribution to the environmental impact of the products. The end producer could use this information to focus on the most important suppliers. In order to improve the environmental performance of the product two alternatives exist; the relevant suppliers must improve or they must be replaced. Improvements depend on the end-producer's ability to motivate or force them. The latter again depends on sufficient channel power. Replacements, on the other hand, are depending on existence of alternatives with better, or a potential for better, environmental performance. The complexity of the supply market must thus also be analysed.

In this paper methods inspired by Pareto-analysis and portfolio matrixes are used to analyse the supply market and effects on the environmental performance simultaneously. In addition a variant of the Overall Business Impact Assessment (OBIA) method is used to compare the environmental performance to overall performance in the economy (cf. Taylor and Postlethwaite 1996).

Case-description

The furniture industry in Norway is dominated by small and medium sized enterprises. The manufacturers are dispersed to most parts of the country, but there is a higher

concentration in the western parts. Several of the suppliers are located in the same area, and there are several long term relations between end-producers and suppliers. These relations have developed over years without deliberate planning. They have just happened (cf. Mudambi et al. 2004), but the managers in the companies have to some degree an impression of mutual dependency and therefore tend to prefer local suppliers.

Many of the furniture manufacturers are depending on public procurement. The Public Procurement Act passed in 1999, and even if the act was not instantly implemented in procurement practices, there is now an increase in environmental demands in announcements of tenders. In 2004 some sort of environmental requirement was put forward in 52% of announcements concerning furniture (Solevåg 2005). There is also evidence for increased sale due to documented environmental performance (Dahl et al. 2002).

A white paper on environmental policy was also presented in 1999, where increased producer responsibility and introduction of take-back legislation was discussed (Ministry of Environment 1999). Furniture was explicitly mentioned.

Since most of the manufacturers are SMEs, they have a challenge in meeting this information demand. Partly due to this situation, several of the manufacturers have joined a research project where one of the goals is to ease the generation of environmental information of the products. A number of analyses are already performed (Brekke and Klæboe 2001; Dahlsrud et al. 2002ab; Fet et al. 2003; Michelsen 2006; Michelsen et al. 2006). A database on environmental impact is developed and the goal is to make EPDs on 80 % of the products (Fet and Skaar 2006)³.

In a previous study it was revealed that the end-producer made a significant contribution to only one out of four environmental impact categories (Michelsen et al. 2006). Consequently, improvements must be carried out elsewhere in the supply chain. The problem is that it seems as the furniture manufacturers have little influence on their suppliers. The response rates on questionnaires sent to the suppliers have been low and also the quality on the information on environmental performance has been poor.

In this study, a chair from one furniture manufacturer (from now on called '*Furniture*') is investigated with respect to what contribution the components of the product have to the overall environmental impact and what channel power *Furniture* has in relation to the suppliers of the components.

The chair in the case study is typically used in nursing homes and institutions. Most of these are public owned in Norway and as a consequence most sales are to the public. It has a high backrest and generally high seating comfort. It has a total weight of approximately 20 kg of which steel constitutes 4.5 kg, solid wood 4.8 kg, MDF 2.6 kg, plywood and laminated wood 1.2 kg, polyurethane 3.8 kg and wool fabrics 1.2 kg. In addition 2.65 kg paperboard is used in packaging. Figure 2 shows the suppliers to the chair and which components they deliver. The figure also shows the network horizon of *Furniture*.

The value added to the product by *Furniture* is primarily based on assembly of components. Some of these processes, such as seam of fabrics, are labour intensive. In total *Furniture* spends 37.5% of the revenues on purchasing. This is less than average for manufacturing companies which is about 55% (Monczka et al. 1998). *Furniture* had in 2004 a total income on almost 83 million NOK (approximately 10.4 million €) and a margin on 6 million NOK.

³ Present EPDs are available on <http://www.epd-norge.no/>

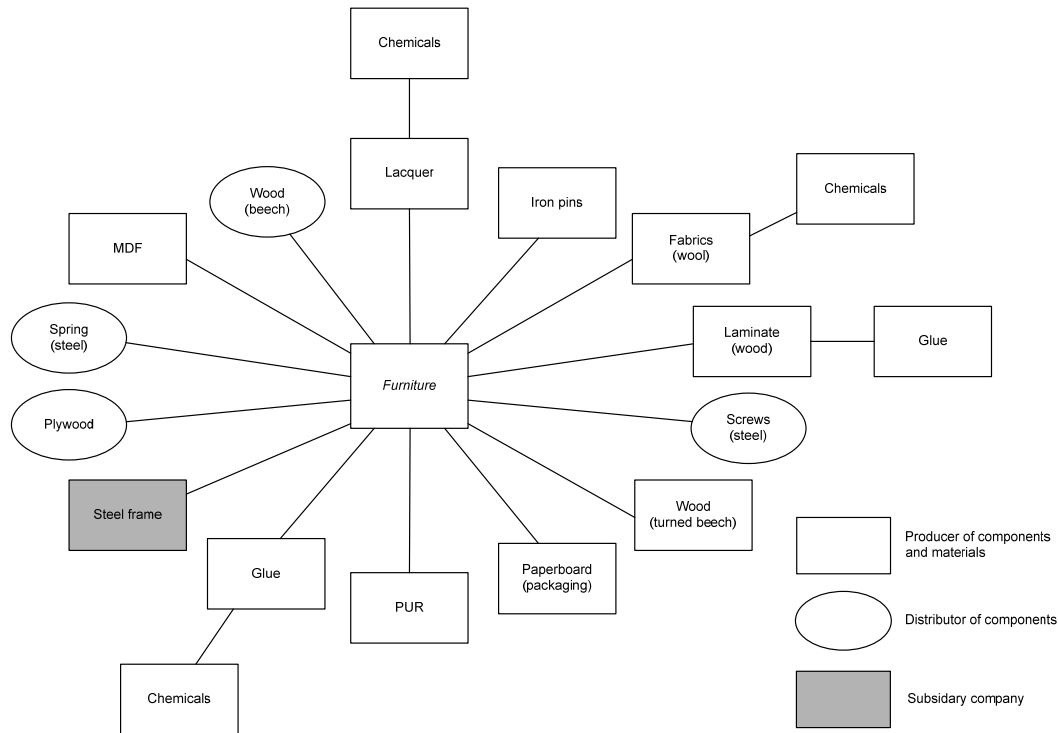


Figure 2 – Network horizon of the supply side showing the producers and distributors of the components in the case model

Methodology

A screening LCA is performed to reveal the environmental performance of the product and its components. Only data from the production phase is used in this paper. The data on production is from the environmental database of furniture production (Fet and Skaar 2006, Fet et al. 2006). Eco-indicator 99 (E/E) is used as impact assessment method.

Components contributing with more than 5% of the total environmental impact of the production phase are regarded to be of interest for further management.

To analyse the relations between *Furniture* and its suppliers primarily two approaches are used. Personnel at *Furniture* were interviewed about the relations to the suppliers and known alternative suppliers. The interviews were also used to identify the network horizon of the supply side (Figure 2). All suppliers were contacted to obtain sales figures. In addition annual reports from *Furniture* and the suppliers were investigated to verify the information and to fill in gaps.

This is low tech industries and most of the theory on complexity of the market (e.g. Kraljic 1983) can not be used. In most cases there are only a few possible suppliers available which should indicate a high complexity. On the other hand *Furniture* or a subsidiary company would be able to manufacture most of the components them selves with only minor investments. This indicates a low complexity. The relationships are thus analysed based on the relative size of the sales compared to the total sales for the suppliers. This is calculated both with respect to the share that goes to *Furniture* and to the furniture industry in Norway as a whole when possible. The latter is done since several of the furniture manufacturers are involved in the project and this would in particular be important if the furniture manufacturers decide to join forces for obtaining information on environmental performance.

Even though the influence a manufacturer has on its suppliers is depending on more than just the share of the total sales, it is assumed that a company responsible for more than 5% of the total sales of a supplier has possibilities to influence the behaviour of that supplier. A Pareto (or ABC) analysis would indicate that such customers must be maintained and requests should therefore be complied with if possible (cf. Christopher 1998, p. 60).

This information on environmental performance and share of total sales is used to make a supplier matrix where environmental impact is related to channel power.

The importance of the suppliers is also analysed using the OBIA method developed by Unilever (Taylor and Postlethwaite 1996) and developed further by Clift and Wright (2000). The main purpose with this method is to compare the ratio between environmental impact and economic activity (the eco-efficiency) between a specific activity and an average value and then reveal activities deviating most from a sustainable production. In this paper the method is used to compare the eco-efficiency of the components from a supplier *i* to the national mean in Norway using the formula

$$\text{Relative impact } (i) = \frac{[\text{environmental impact of components } (i)/\text{value of components } (i)]}{[\text{national environmental impact/GDP}]}$$

Values above 1 indicate higher environmental impact per capita than the average. In the case study emissions of greenhouse gasses are used as an environmental impact indicator. This is one of the key performance indicators in the Norwegian EPDs⁴ and is reported in a consistent way. National values on emissions and GDP are 2004-data from Statistics Norway (2006).

Results

Table 1 gives an overview of the components provided by *Furniture's* suppliers and used in the case model. The table shows the environmental impact of the components as the percentage of the total environmental impact of all upstream activities, i.e. the impact associated with production and transportation of the specific components delivered from the suppliers. The suppliers are ranked based on this impact. The cumulative environmental impact from the suppliers is also shown in Figure 3. Total impact from the included processes is 3190 mPts.

The two next columns give numbers on the importance of *Furniture* and the furniture industry in Norway as a whole for the suppliers. The values are given as the share of the total sales that goes to these two 'actors'. The last column gives a number on the relative size of the supplier compared to *Furniture*. This is based on annual sales. Values above 1 indicate that the supplier has larger annual turnover than *Furniture*.

Figure 4 shows the supplier matrix of the 14 suppliers listed in Table 1. The x-axis shows the relative environmental impact from the components provided by the different suppliers and the y-axis the percentage of total sales which is a proxy for channel power. All suppliers are plotted twice; once where the percentage of total sales to *Furniture* is used (solid circles), and once where the total sales to furniture industry in Norway are used (open circles). As the figure shows, only two suppliers (2 and 3) fulfil both the demand for more than 5% of total environmental impact and 5% of total sales when only sales to *Furniture* are used. This increases to 5 suppliers (1, 4 and 5 in addition) when the sales to the furniture industry are used instead.

⁴ see <http://www.epd-norge.no/>

Table 1 – Overview of the importance of components and suppliers in the case study

Supplier	Materials/ components	Environm. impact ^a	Percentage of sales to <i>Furniture</i> ^b	Percentage of sales to furnit. industry ^c	Size ^b
1	PUR components	44.5	2.5	~40.0	0.85
2	Solid wood (beech)	14.2	6.5	57.5	0.32
3	Steel frame, other steel components	13.0	48.0	52.0	0.10
4	Wool fabrics	10.9	0.03	~50.0	1.01
5 ^d	Paperboard (packaging)	5.8	0.2	5.1	8.61
6	MDF	3.6	3.1	29.0	0.93
7	Plywood	2.9	<0.01	75.0	0.59
8 ^e	Steel spring	2.1	<0.01	N/A	454.0
9 ^f	Laminated wood	0.7	2.5	24.1	0.36
10	Glue	0.7	<0.01	N/A	N/A
11	Lacquer	0.6	0.6	22	1.04
12	Solid wood (beech, turned)	0.5	1.2	63.5	0.29
13	Steel screws	0.3	0.1	N/A	8.33
14	Iron pins	0.3	2.5	70.0	0.05

- a) Measured as the percentage of the total environmental impact (in Ecopoints) of all upstream activities, i.e. the impact associated with production of the specific components, not only the impact caused by the first tier suppliers
- b) Based on average values for 2002 and 2003. Data obtained from annual reports
- c) Average values for 2002 and 2003. Data are estimates from suppliers
- d) Only including production sites in Norway. The company is a part of a larger concern that is 32.4 times the size of *Furniture*
- e) Values are based on total values for a multinational company. In 2001 (the only figures available) 0.005% of the total sales took place in Norway. Furniture industry had a large share of this, but details are not available
- f) Average values for 2003 and 2004 due to change of ownership and strategy in 2003. The percentage to furniture industry was 9.1 in 2002. The company became in 2003 a part of a larger concern that is 5.56 the size of *Furniture*

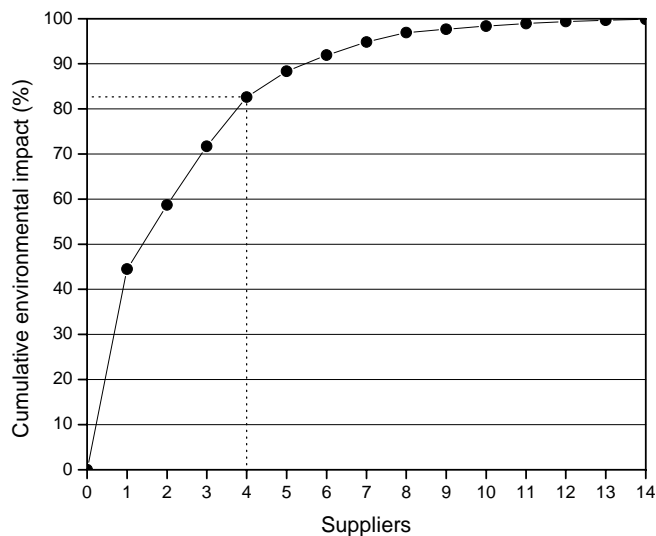


Figure 3 – Cumulative environmental impact related to the number of suppliers

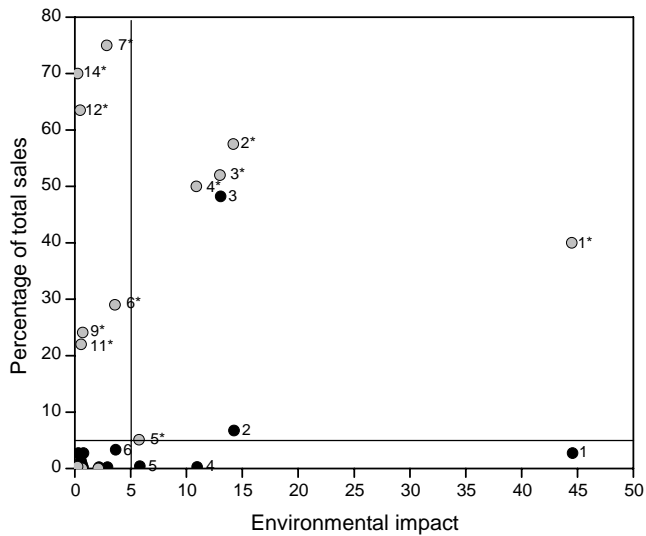


Figure 4 – Supplier matrix with focus on environmental impact and the share of total sales to Furniture (solid circles) and furniture industry in Norway (open circles)

The ratio between the environmental impact and economic activity related to the components delivered from the different suppliers is compared to a national average in Figure 5. All values above 1 indicate lower eco-efficiency than national average. Environmental impact is here measured as emissions of greenhouse gasses. The components from supplier 1 and 10 stand out as the least eco-efficient components, but also 3 and 5 have more than 50% higher environmental impact per capita than the national mean.

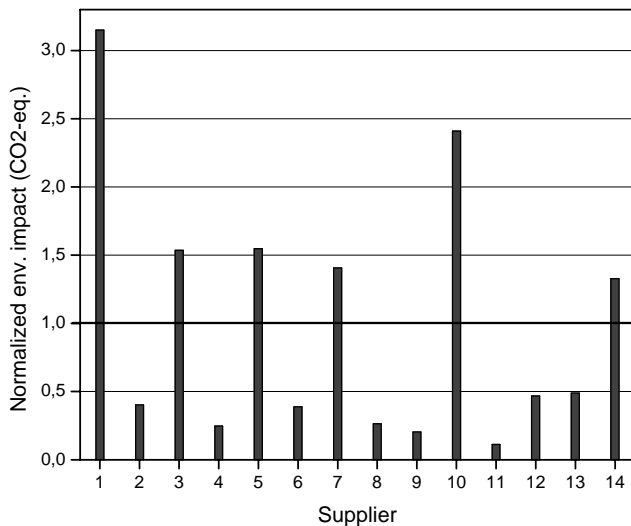


Figure 5 – OBIA-diagram showing the normalized environmental impact of the components delivered by the different suppliers

Discussion

This paper presents different methods that are combined to reveal how many supplier relationships a manufacturer must control to have sufficient insight into the origin of most of the environmental impact generated in upstream activities. The possibilities the end-producer in the case study (*Furniture*) have to influence these suppliers are also analysed.

In accordance with other studies, the network horizon of *Furniture* is rather narrow (Figure 2). *Furniture* does not have a complete knowledge of the origin of the materials used in the case model, and for some components they only know the final distributor and little about who is involved in the production. It is obvious that if these are to be influenced, *Furniture* either needs better knowledge of the actors upstream, or they are depending on their suppliers' ability to disperse the environmental focus further in the supply chain (cf. Lamming and Hampson 1996; Hall 2000).

The screening LCA shows that more than 80 percent of the total environmental impact is originating from only 4 suppliers (Figure 3). 5 suppliers are contributing with more than 5 percent each of the total impact and these are in total responsible for more than 88 percent of the impact. Most of the environmental impact can thus be controlled through a manageable number of suppliers.

However, the ranking of the suppliers (Table 1) is to some degree depending on the selected impact assessment method. EPS 2000 and CML 2 baseline 2000 are tested as alternatives to Eco-indicator 99. Both gave significantly less impact to wood based components, which in particular reduces the focus on supplier 2. Both methods allocate only 1.5 percent of total environmental impact to supplier 2. The choice of impact assessment method has only minor importance for the valuation of the other suppliers.

Two of the suppliers (2 and 3) should be possible to influence due to the leverage power *Furniture* has as a customer (Figure 4). Supplier 3 is in addition a subsidiary company. The possibilities to influence the other suppliers are less, but two options exist. One strategy would be to cooperate with other furniture manufacturers since the furniture industry is important for the other environmentally important suppliers (1, 4 and 5). The second opportunity is to rely on the suppliers' wish to be associated with 'green' products and thus regard *Furniture* as a strategic customer for their own reputation (cf. Forman and Jørgensen 2004). The latter option is so far not further investigated.

The possibilities to improve the environmental performance are most likely present for all suppliers, e.g. with focus on the environmental performance of all processes (cf. Porter and van der Linde 1995). However, in Figure 5 the eco-efficiency of the components provided by the different suppliers are compared to the national mean and thus shows where the potential for large improvements are most likely to be found. The figure shows that suppliers 1, 3, 5, 7, 10 and 14 are all above the national average for emissions per capita. A large part of the activities upstream supplier 4 (wool production) is not included in the analysis. The score for this supplier is hence too low, but no better data exists.

The results thus advocate a focus primarily on suppliers 1, 3 and 5. For supplier 4 better data is needed. It is a matter of discussion if supplier 2 is to be included. The high rank when Eco-indicator 99 (E/E) is used is caused by impact from land use. This category is not included in the other impact assessment methods and is highly debated (i.e. Michelsen 2004; Milà i Canals et al. 2006).

Opportunities for improving the performance of these suppliers should be further explored. The production of PUR (supplier 1) is possible to improve significantly (Diaz and Redondo 2002), but it is questionable if this is cost-effective (Michelsen 2006). The supplier is also participating in a research project where the possibilities to regenerate the PUR is tested, and through redesign it could be possible to reduce the amount of PUR.

The environmental assessment of steel (supplier 3) is based on average values. So far this is correct since the steel is bought on a case to case basis, but a higher focus on environmental performance and demands for a higher percentage of recycled steel would improve the performance.

The potential for improvements on paperboard packaging (supplier 5) is less obvious since recycled fibres are already utilized, but also here alternative materials or a higher focus on the processes should be explored.

A development of a purchasing strategy that includes an environmental focus could be the first step towards supply chain strategies and a more active supply chain management for a company like *Furniture* (cf. Pagell and Krause 2002). However, it is often necessary to help suppliers, especially SMEs, with implementing environmental management systems (i.e. Handfield et al. 1997). In the case of *Furniture*, the problem is that the end-producer is itself an SME. The most obvious strategy in this case is to develop the cooperation between the furniture manufacturers further and make this consortium the channel leader. Most of the suppliers in the case are to a large degree depending on the furniture industry (Table 1, Figure 4) and should also be more actively involved in the running project on environmental performance on furniture production in Norway.

Conclusions

In this paper it is shown that a combination of environmental assessment information and knowledge on supplier relationships is an effective way to both dramatically reduce the number of suppliers that should be managed from an environmental point of view and also obtain an overview over which suppliers that can be influenced. When this is combined with OBIA-metrics a picture of the probable potential for improvements of environmental performance is displayed. The establishment of a purchasing consortia or some kind of cooperative purchasing seems to be an alternative for small and medium sized manufacturers that themselves have limited leverage power.

Analyses such as those presented in this paper are a suitable starting point for more active supply chain management. The creation of databases on environmental impact, as the one on furniture production in Norway (Fet and Skaar 2006, Fet et al. 2006), has been the starting point in the case presented. This must be supplemented with information on supplier relations. In many industries this is to some degree known even if detailed information is not available.

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Appendix E

Environmental impact and added value in forestry operations in Norway

Environmental impact and added value in forestry operations in Norway

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Abstract

The forestry sector is experiencing an increasing demand for documentation on environmental performance. Previous studies have revealed large differences in environmental impact both due to location and forestry practice and reliable information on environmental performance for forestry operations in different regions should thus be obtained. This paper presents a case study of forestry operations in Norway where both environmental performance and value added in the selected operations are assessed. This is done using a hybrid LCA approach. Main results including sensitivity analysis are presented for a set of four impact categories. The production chain assessed includes the processes from planting of forest to the delivery of logs to a downstream user. The environmental impact is mainly caused by logging, transport by forwarders and transport by truck. These three operations are responsible for approximately 85% of the total environmental impact. The impact on value added and total costs are more evenly distributed. The sensitivity analysis reveals that the difference in the worst case scenario and the best case scenario is more than a four-fold in environmental impact. The single most important factor is the transport distance from pile to factory. The results show that the environmental impacts from forestry operations in boreal forests probably are underreported in earlier studies.

Keywords: LCA, input-output analysis, forestry, environmental impact, value added

Introduction

The forestry sector is important in Norway. In 2004, wood and wood-based products for more than 14 billion Norwegian kroner (NOK¹) were exported. This is about 7% of the export from land-based activities, excluding oil and gas exports (Statistics Norway 2005). The forestry sector is also important for employment, especially in rural areas, and woodworking industry is present in more than 70 percent of all municipalities in Norway (The Ministry of Agriculture 1998).

More than one third of the country is covered with forest and 74 000 km² is productive. The timber volume in the productive forests is estimated to 705 million m³ (Hobbelstad et al. 2004). The forests are also important habitats for a range of species; almost half of the species in the Norwegian Red List depend on the forests (The Directorate for Nature Management 1999). Forestry is thus important for the biodiversity.

Wood is a renewable material and the regrowth in Norway is at time being estimated to 24 million m³/year (Hobbelstad et al. 2004). The annual logging the last decade has been approximately 8 million m³/year (Statistics Norway 2006a). Still the use of wood is

¹ 100 Norwegian kroner (NOK) \approx 12.50 Euro

debated, in particular due to the impact on the forest ecosystems and habitat loss (Sanness 2003; Petersen and Solberg 2005; Puettmann and Wilson 2005). Wood-based products do not have a solely positive environmental reputation and the public's confidence in non-proven marketing statements like 'environmental friendly', e.g. due to the renewable nature of wood, is rapidly declining (Kuckartz 2000). Reliable information and data combined with a consistent methodological framework is thus required to compare forest products to other products, which primarily will be non-renewable.

Several studies have compiled life cycle inventory data on forestry operations. These reveal large differences in environmental impacts depending on location, forestry management and logging techniques (e.g. Schweinle 2000; Berg and Lindholm 2005; Johnson et al. 2005). As an example, Berg and Lindholm (2005) have shown that emissions of greenhouse gases during forestry operations are almost 40 percent larger in northern Sweden than in the south. Some of the differences between the studies might also be a result of methodological inequalities. Studies focusing on products where wood is a major component must thus be performed carefully to account for the large variations due to geography and management practice.

The purpose of this study is to assess the environmental impact from forestry operations in a given region in Norway. This is done using a hybrid LCA approach. The focus is on spruce logging, primarily *Picea abies*, which is the dominant species in Norwegian forestry, constituting about 75% of the logged volume. All forestry activities are included as well as transport to a downstream user. An average score is assessed, as well as a best case and a worst case scenario for environmental impact. The results are also compared to other LCA studies on forestry operations in order to see differences both in actual environmental impact and methodological choices.

Case description

This study is performed in cooperation with ALLSKOG BA. Until 2006 ALLSKOG was known as Skogeierforeninga Nord (SN) and was a forest owner organization. In 2005 SN was reorganized and is now a co-operative society with 9250 part owners, primarily relatively small forest owners in 5 counties in the northern parts of Norway (see Figure 1). In 2005 SN sold 759 000 m³ locally logged timber (Skogeierforeninga Nord 2005) which is 70 percent of the timber logged in this region (Statistics Norway 2006a). 52% of the timber is sold as sawn timber, while the rest is pulp wood, primarily for paper production, chips and firewood. In the end, however, more than half of the timber ends up as pulp wood since residuals from sawmills to a large extent are used in paper production (Figure 2).

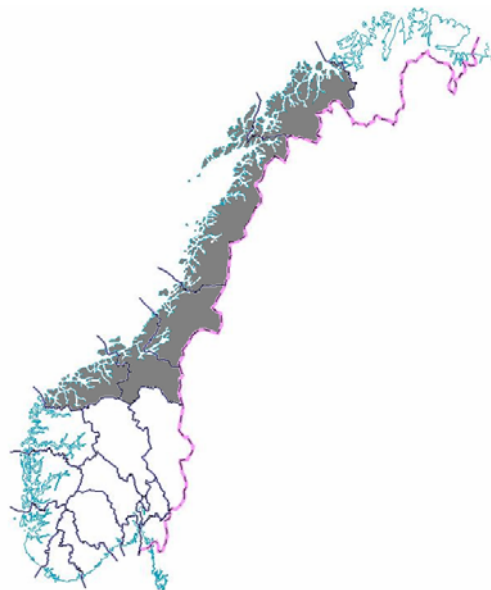


Figure 1 – Geographical location of ALLSKOG's forestry activities in Norway

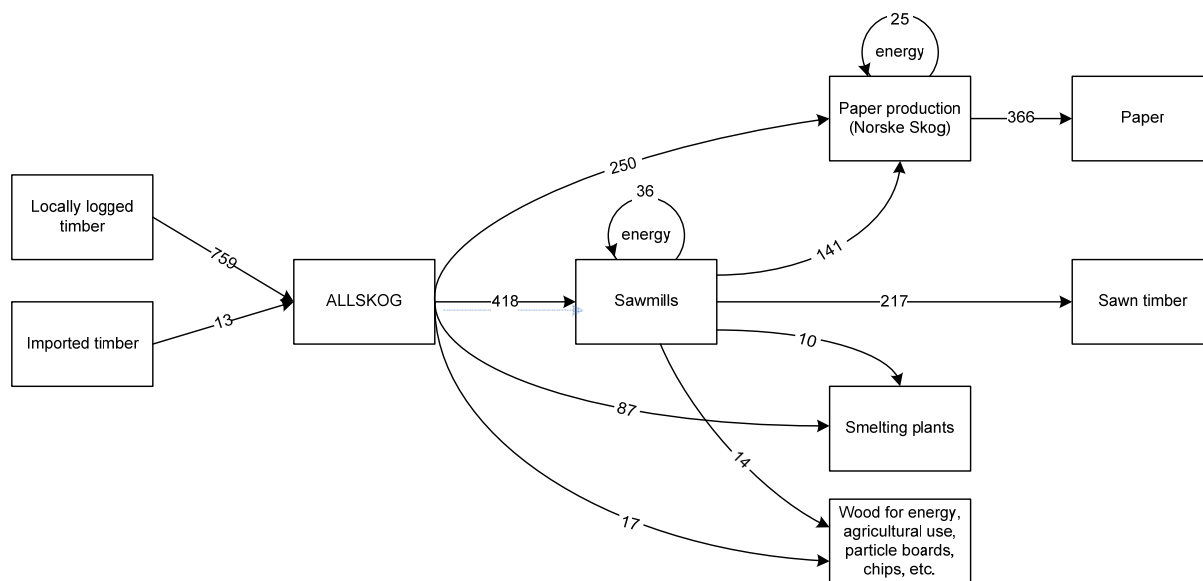


Figure 2 – Wood flow based on timber logged by ALLSKOG (2005 figures in 1000m³). Only wood from ALLSKOG and products originating from this is included

It is the forest owners that are responsible for all forestry operations, but most forest owners sub-contract some or all of the operations to ALLSKOG, such as planning, planting, silviculture, logging and sales. As an example SN had the direct responsibility for logging 557 000 m³ in 2005, which is more than half of the amount of wood logged in the region. Of this, 94% was taken out with harvesters and forwarders (Skogeierforeninga Nord 2005). Other logging techniques, such as use of chainsaw and cable yarding, are not taken into account in this study, due to their small contribution.

As already mentioned, woodworking industry is present in a majority of the municipalities in Norway, and in 2005 SN had in total 515 customers. Most of these are however small, and the 10 largest purchase almost 95% of all logged timber. This number has increased during the last years since a few of the largest customers have experienced a significant growth in through-put.

ALLSKOG has ownership interests in two sawmills in the region as well as some subsidiary companies; a wholly-owned subsidiary producing and delivering chips to smelting plants, 50% of a company for marketing and delivering of wood pellets for heating and 50% of a transport company responsible for almost all timber transport in the region. Some of the sawmills have ALLSKOG as their only supplier. ALLSKOG is therefore an important actor in the forestry and woodworking sector in the region.

The forestry sector in Norway has experienced a significant increase in the pressure to provide information on environmental performance during the last decade. In particular the paper producing company Norske Skog has been exposed on export markets with a demand to only use wood from 'sustainable forestry'. Norske Skog has passed this pressure to their suppliers and has encouraged them to introduce environmental management systems to secure documentation of their performance (cf. Sanness 2003). In a period, Norske Skog paid additional 7 NOK per m³ if the timber was from environmental certificated forestry (ISO 14001 with the Norwegian PEFC² standard 'Living Forest'³ as basis for the forestry performance - Sverdrup-Thygeson et al. 2004). As shown in Figure 2, Norske Skog is by far ALLSKOG's most important customer, purchasing almost one third of all timber. Partly as a consequence of this, SN became

² <http://www.pefc.org/>

³ http://www.levendeskog.no/Engelsk_Default.asp

certified following ISO 14001 and ISO 9001 in December 2000 (cf. Rametsteiner and Simula 2003).

However, there is a growing concern for environmental issues within the entire woodworking sector (Sverdrup-Thygeson et al. 2004) and Sverre Thoresen, environmental corporative advisor in Norske Skog, assumes that environmental performance will become a competitive factor in the future⁴. All sawmills⁵ in the region regularly receive questions on environmental performance of their products.

The questions on environmental performance are to a large degree related to impact on biodiversity in the forest (cf. Seppälä et al. 1998, Hanski and Walsh 2004). However, at present there is no agreed upon methodology for including the impact on biodiversity from land use activities in LCA (Milà i Canals et al. 2006). This will thus not be discussed in this paper but will be treated separately in a forthcoming publication (Michelsen in prep.).

The functional unit in this study is the production of one m³ round wood logs under bark delivered at the gate of a factory. The factory might be a sawmill, a pulp- or a chip producer (cf. Figure 2).

The system boundaries include planning of forestry operations, seedling production, soil scarification and planting, silviculture (mechanical cleaning of undesirable vegetation, fertilization, chemical cleaning and weed combating and drainage), harvesting (felling, pruning and cutting into logs), transport to pile at forest road, construction of forest roads and transport from pile to a factory (Figure 3).

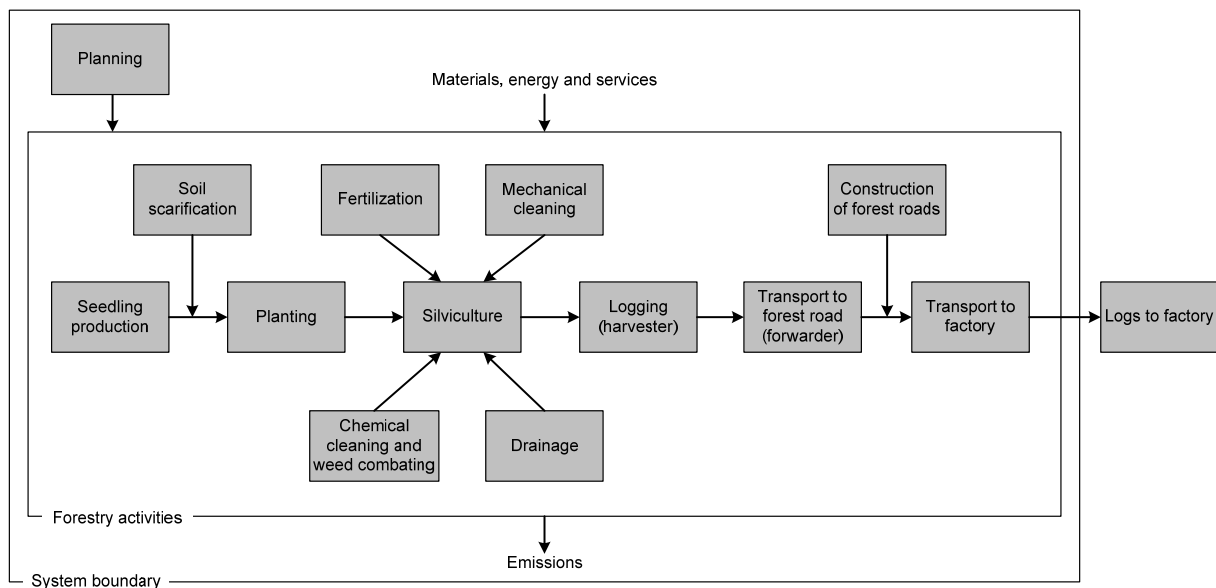


Figure 3 – System boundaries. The functional unit is 1 m³ round wood logs under bark delivered at the gate of the factory

Methodology

Previous life cycle assessments on forestry operations have focused on physical life cycle inventories (e.g. Schweinle 2000; Berg and Lindholm 2005; Johnson et al. 2005). However, the combination of physical life cycle inventories (LCI) with input-output (IO) data has received significant interest within the Industrial Ecology and LCA community over the last years. Suh et al. (2004) provides a rationale for the application of hybrid LCA approaches based on system boundaries issues in traditional LCA.

⁴ Statement in an interview in the magazine 'Norsk Skogbruk' (Norwegian Forestry) 5/2006

⁵ Only sawmills with an annual flow on more than 5000 m³ were contacted

The hybrid LCA system used in this paper is formulated by adapting the notation of input-output analysis (IOA). The Open Leontief model is applied (Leontief 1936). This is a linear model of process interdependence. Central to its understanding is the requirements-, or coefficients-, matrix A . The columns of the A matrix describe the intermediate inputs a production process requires from itself and other processes, to produce one unit of output. These inputs can be expressed in any units; it be monetary, mass or energy. In common LCA terminology this A matrix contains the inventory data on inter-process relations. Further, the remaining inventory data, the emissions end stressor intensities for each of the processes is given in the F matrix. The C matrix contains characterization factors for the various stressors. The functional unit of the system is defined by the y -vector, and I is the identity matrix. An LCA can then be expressed in a single equation yielding the vector d of category indicator results:

$$d = CF(I - A)^{-1}y$$

In the analysis an approach for establishing hybrid inventories developed by Strømman and Solli (2006) is applied. This approach allows for estimating missing inventories from input-output data utilizing knowledge of product prices. Principally input-output based data is combined with original key data and adapted to represent the input structure of the processes in question. The application of Leontief's price model (Leontief 1949) is essential in doing this.

The method of Strømman and Solli (2006) requires an identification of which sectors of the economy the various foreground processes belong to. The input structure of these sectors is used as models for the missing inputs. Further, the structures are scaled so that they together with the original key data satisfy the Leontief price model. The resulting hybrid LCA structure is then a model that is valid in both the primal and dual form. That is, it has a consistent representation of both the flow- and cost-structure. This is quite advantageous when performing eco-efficiency assessments and is the main motivation for applying this specific framework in this study.

A structural path analysis is applied to identify the most significant contributions to environmental impact. Thus, it is possible to see whether the contributions are based on specific process data or estimated data. See Strømman and Solli (2006) for more details on the applied methodology for input-output analysis and Peters and Hertwich (2006) for more details on structural path analysis.

For this study the Norwegian input-output matrix for 2000 was applied (Statistics Norway 2003). The matrix includes capital and imports and is compiled in basic prices plus trade, transport and FISIM (Financial intermediation services indirectly measured) margins. The vector of value added is supplied additionally by Statistics Norway (see Appendix A). The environmental stressor intensities of each sector include emissions of 20 components contributing to global warming, photochemical oxidation, acidification, eutrophication and human toxicity potential (HTP), see Table 2.

Assumptions and data sources

Time is a critical element in life cycle inventories of timber production. In a boreal forest the rotation period might be 100 years or more. Planting and silviculture are thus carried out long before logging and in most cases under different management principles and methodologies than what is common today. Similarly, the areas planted or left for natural regeneration will not be logged for about a century, and it is not possible to know for sure which principles that will apply at that time.

Despite this time-lag, the present level of planting and silviculture is allocated to today's level of logging since no better options are available with current knowledge. The amount of planting, silviculture, logging and forest road construction is based on annual average

data from the period 2000-2004. This is the best available data for the given region. Other assumptions and data sources are given in Appendix A.

Average values for environmental impact and value added is calculated in accordance with the functional unit. In addition a worst case and a best case scenario are assessed based on three assumptions (Table 1). First, the size of the log has major impact on the diesel consumption during logging (Kjøstelsen and Lileng 2006) and average size in a logging area is here assumed to range from 0.1 to 0.5 m³/log. Second, the distance from the logging spot to forest road is important. Here it is assumed that this ranges from 50 meter to 3000 meter. Finally, the transport distances from pile to factory registered in the region ranges from 12 to 301 km. It is also assumed that the loading factor will be somewhat higher on long distances, while there is no return cargo at the shortest distances (giving a loading factor of 50%). Estimated diesel consumptions in the processes due to these assumptions are also shown in the table.

Table 1 – Estimated diesel consumptions and assumptions taken in calculation of worst case and best case scenarios

		Best	Average	Worst
Harvester	Average size of log (m ³)	0.50	0.24	0.1
	Diesel consumption (l/m ³)	0.50	0.83	1.48
Forwarder	Average distance to forest road (m)	50	740	3000
	Diesel consumption (l/m ³)	0.44	1.03	2.97
Truck	Distance to factory (km)	12	120	301
	Loading factor (%)	50	55	60
	Diesel consumption (l/m ³)	0.42	2.73	6.56
Total	Diesel consumption (l/m ³)	1.36	4.59	11.01

For each process in the production chain the basic prices are identified. The basic price is not equal to the price the purchaser has to pay since the amount of subsidies (in particular related to planting, silviculture and forest road construction) and taxes is not included (United Nations 1999), but represents a cost that has to be covered to run the system. The resource rent paid to the forest owner is also not included. Both subsidies and the resource rent fluctuate due to shifting policy and market possibilities and it is here decided to relate the environmental impact to the actual and fixed costs. This assumption does not influence the environmental assessment. The method does, however, presuppose a cost breakdown where the value added (VA) is identified. In the forestry operations VA is primarily salaries and to some degree dividends and retained profit (cf. Sturm et al. 2003).

Results

The average scores for all impact categories and costs are shown in Table 2. The impact from soil scarification is included in the impact from planting, while silviculture is the sum of fertilization, mechanical and chemical cleaning and drainage. For silviculture more than 90% of the impact is due to mechanical cleaning (data not shown). Absolute and relative values for the different processes are shown. As the table shows, the emissions are primarily caused by logging operations, transport by forwarder and transport to factory. For emissions of greenhouse gasses, these three processes are responsible for almost 84% of all emissions, and similar numbers for acidification and eutrophication are 85% and 89% respectively. When it comes to emissions of photo oxidants and human toxic compounds, also forest road construction is of importance.

Costs, and in particular value added, are more evenly distributed in the system than environmental impact. Logging, transport by forwarder and transport on truck here account for 66 % of the total costs, and only 52 % of the value added. Also here forest road construction is of importance, and for VA also seedling production, planting and silviculture make significant contributions.

Table 2 – Absolute and relative impacts from the processes included in the product system

		Planning	Seedling production	Planting	Silviculture	Logging	Transport by forwarder	Forest road construction	Road transport	Total
Global warming potential	kg CO ₂ -eq.	0.251	1.244	0.296	0.222	4.680	4.694	2.033	11.628	25.048
	Relative contribution	1.0	5.0	1.2	0.9	18.7	18.7	8.1	46.4	100
Photochemical oxidation potential	kg C ₂ H ₂ -eq.	0.001	0.003	0.004	0.002	0.012	0.011	0.010	0.034	0.076
	Relative contribution	1.2	3.8	4.6	3.2	15.4	14.8	12.8	44.2	100
Acidification	kg SO ₂ -eq.	0.001	0.006	0.002	0.001	0.021	0.023	0.006	0.052	0.111
	Relative contribution	0.6	5.8	1.5	1.0	19.0	20.6	5.2	46.3	100
Eutrophication	kg PO ₄ -eq.	0.000	0.001	0.000	0.000	0.005	0.005	0.001	0.012	0.024
	Relative contribution	0.5	3.4	1.6	1.1	19.0	21.5	3.9	49.1	1
HTP air, cancer	kg benzene eq.	0.000	0.000	0.000	0.000	0.003	0.001	0.002	0.004	0.011
	Relative contribution	2.5	3.8	1.2	1.1	22.8	12.6	20.9	35.0	100
HTP air, noncancer	kg toluene eq.	0.223	0.370	0.142	0.118	2.162	1.269	1.753	2.728	8.765
	Relative contribution	2.5	4.2	1.6	1.3	24.7	14.5	20.0	31.1	100
Total cost	NOK	10.86	21.45	23.90	15.21	75.49	50.32	39.19	91.00	327.41
	Relative contribution	3.3	6.6	7.3	4.6	23.1	15.4	12.0	27.8	100
Value added	NOK	4.47	11.50	21.12	12.60	24.91	20.13	15.68	27.66	138.07
	Relative contribution	3.2	8.3	15.3	9.1	18.0	14.6	11.4	20.0	100

The relationship between environmental impact and total costs (basic prices) and value added are shown in Figure 4 and Figure 5 respectively. The steeper the line is, the higher is the environmental impact per cost unit.

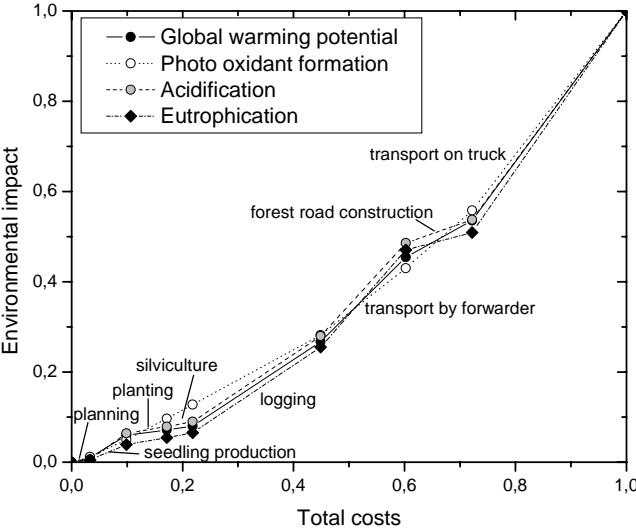


Figure 4 – The relationship between environmental impact and total costs for the included processes

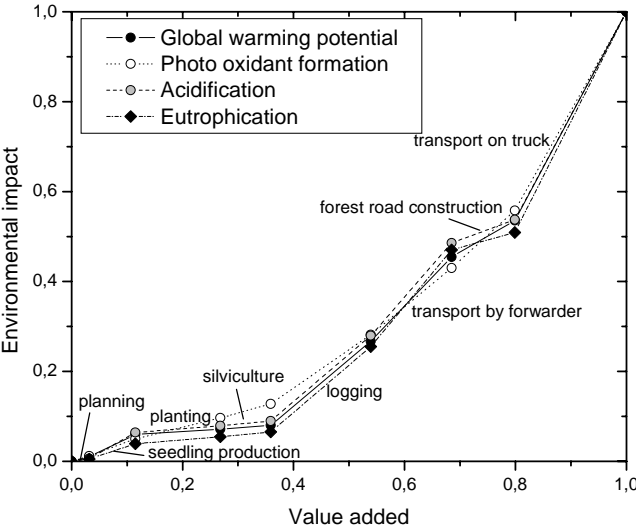


Figure 5 - The relationship between environmental impact and value added for the included processes

In Figure 6 the scores for the best case and the worst case are shown as relative values to the average. Four impact categories are included. In the best case the total costs sum up to NOK 210 and VA to NOK 99. In the worst case the numbers are NOK 564 and NOK 221 respectively.

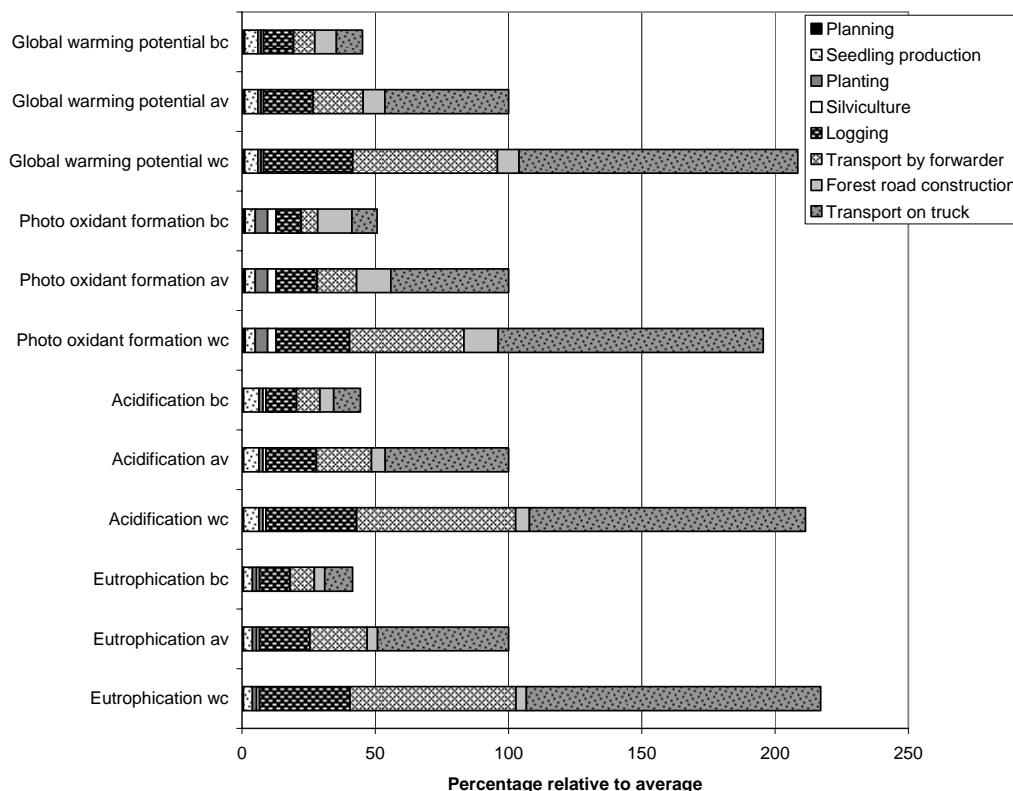


Figure 6 – Relative scores for the best case scenario and the worst case scenario compared to average values

Discussion

The most important processes when it comes to emissions are logging and transport operations. This is in accordance with previous findings (Berg and Lindholm 2005; Johnson et al. 2005). Our results do, however, show higher emissions than reported by Berg and Lindholm (2005) even though both forestry practice and climatic conditions for forestry is comparable in these studies. As an example, our results indicate an average emission on slightly above 25 kg CO₂-eq/m³ (Table 2), which is more than 40 percent higher than Berg and Lindholm (2005) found for a similar system in northern parts of Sweden.

However, the difference of comparable inputs is far less. In fact, the diesel consumption for logging operations and transport to factory is 4.69 l/m³ in northern Sweden (Berg and Lindholm 2005), while our results give an average on 4.59 l/m³ for the same operations (Table 1). Berg and Lindholm (2005) state that their results show significantly higher emissions from forestry operations than earlier reported in the Scandinavian countries, but also their emissions are probably underestimated. This is consistent with recent literature on hybrid LCA (Suh et al. 2004; Strømman et al. 2006). Since hybrid LCA studies generally has more complete upstream system descriptions than standard LCA inventories, they capture a larger share of the total impacts generated (Suh et al. 2004; Strømman et al. 2006).

Issues related to system boundary selection are relevant for the comparison of our results with those of Berg and Lindholm (2005). They have not included the life cycle of capital goods or the transport of energy carriers, and a more narrow system boundary has thus been applied. This at least partly explains the difference from our results.

We find that there are much larger variations in transport to factory than in forest operations (logging and transport to forest road). The worst case scenario has 16 times

as high fuel consumption as the best case scenario when it comes to transport on truck, while the differences in logging and transport in the forest is less than 5 times. The relative importance of these operations is comparable with what Berg and Lindholm (2005) found in Sweden. The diesel consumption here was 1.48-1.78 l/m³ for forest operations and 2.13-2.91 l/m³ for transport on truck. In addition they have included some transport on electric railway which corresponds to diesel consumption up to 0.24 l/m³. Johnson et al. (2005) report similar diesel consumption for logging operations in the Pacific northwest of USA (1.70 l/m³)⁶ while they report a significantly higher consumption due to road transportation (6.30 l/m³).

However, for the time being, the worst case scenario for forest operations is highly hypothetical. The estimated basic price (costs) of logging and transport by forwarders are in this case estimated to 280 NOK/m³ which would have made the logging unprofitable. The range for logging operations are thus even smaller than the scenarios indicates, while this is not the situation for transport on trucks since this is based on real data. The differences from site to site are thus primarily a result of transport distances from the pile to the factory.

The data used in this study is a combination of process data and data based on input-output (IO) data. Six environmental impact categories are included (see Table 2). The results for the HTP-categories are almost entirely based on IO data. This is in particular problematic for the results for forest road construction since data from the construction sector here is used and thus probably somewhat different from construction and maintenance of forest roads. Even though the data are adjusted through known economic data and a structural path analysis, the result might still be diverging from what actually is the real situation. In addition, also the economic data for forest road construction are uncertain (see Appendix A) so this should be treated very carefully. The HTP-data are thus only included in Table 2 and not in the figures.

For improvements of the system, a further investigation of the trade-off between forest road construction and transport by forwarders should be performed. Long transport distances by forwarders make a significant contribution in the worst case scenario, but at the same time the impact from forest road construction is significant. The impact from forest road construction is, as pointed out above, uncertain. Also impacts not included here, such as habitat loss due to fragmentation (Michelsen 2004) should be considered in such analyses (cf. Michelsen in prep.).

Other possibilities include looking into different logging techniques and also the trade-off between high planting frequency and costs and emissions due to this, compared to natural regeneration. Planting is relatively costly (seedling production and planting sum up to almost 14% of the total costs), but at the same time shortens the rotation period and consequently make logging in hard accessible areas less necessary.

However, the largest potential for improvements is in transport to factory, causing almost half of the environmental impact. Here, several improvements could be foreseen. The loading factor, in particular on long distances, could be significantly increased since there at present is almost no return cargo. Also, better engines and alternative fuels would make significant improvements. Other transportation options, e.g. by rail or boat, are not analysed here. A transfer to these could make improvements, but depend on heavy investments in infrastructure. Never the less, the area in which ALLSKOG operates includes the most rural areas in Norway and fuel consumption rates equal to what is found in Sweden and less than half of what is found in Pacific northwest USA, indicate that the road transport already is performed comparatively efficient.

⁶ Johnson et al. (2005) have included loading on trucks in the logging operations. In this number this is subtracted and moved to the transport operations to make the system division equivalent with the division in this study and in Berg and Lindholm (2005)

However, the most important goal with this study was to provide reliable data for forestry operations in a region in Norway. The environmental improvement potentials depend on the system boundaries. O'Rourke et al. (1996) state that any system can be efficient as long as it is defined small enough. The converse may also apply; sometimes it is necessary to define the systems large enough to reveal the potentials for improvements. In the case presented in this paper, it is obvious that the environmental performance for an average log delivered at factory gate will improve if only the easiest accessible areas are logged. Increased logging will therefore most likely increase the environmental impact from an average log since less accessible areas also must be logged. This situation might, however, be totally different if timber is compared to other materials, e.g. concrete in constructions. On the basis of this study it is not possible to see if the timber logging is a significant part of the overall environmental problem in e.g. construction industry (cf. Sanness 2003; Petersen and Solberg 2005; Puettmann and Wilson 2005) and should be improved, or if timber logging is a part of the solution and should be expanded to provide more materials to substitute other materials. At least some studies show that the latter is the best alternative (e.g. Deroubaix 2004; Lippke et al. 2004; Petersen and Solberg 2005).

Conclusions

Our result show that emission factors from forestry operations probably are underestimated due to narrow system boundaries. In addition, there are large variations in emissions from forestry operations due to different log size, different transport distances in the forest, and in particular different transport distances from the pile and to the factory. Average data on forestry operations should hence be cautiously used and avoided if possible.

Acknowledgement

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Appendix A – Data sources and assumptions

Data sources for total costs, value added and emissions in the included processes

	Basic prices	Value added	Emissions
Planning	1	2a	2a
Seedling production	1	3	4
Planting	5	1	2b
Soil scarification	6	1	2b
Silviculture			
- young forest tending	6	1	2b
- fertilization	6	2b	2b
- chemical cleaning	6	2b	2b
- drainage	6	1	2b
Logging – harvester	7	7	8
Transport – forwarder	7	7	8
Forest road construction	6	9	2c
Transport to factory	10	11	12

1. Data from ALLSKOG
2. Norwegian input-output data from year 2000 (Statistics Norway 2003). Additional data for the Norwegian emission inventory is provided as a result of a collaboration between The Norwegian Pollution Control Authority (SFT) and Statistics Norway (SN). SN has been responsible for the development of the emission models, for the collection and processing of activity data, and for the calculation of national emission levels. SFT has been responsible for developing emission factors and providing data reported by industrial plants and specific industries
 - a. sector 70-74: Real estate, renting and business activities
 - b. sector 2: Forestry, logging and related service activities
 - c. sector 45: Construction
 - d. sector 602 Land transport
3. Data from 'Skogplanter Midt-Norge' (a nursery in the region)
4. Average data from Aldentun (2002), other emissions as 2b
5. Plant density as 6, costs per unit from ALLSKOG
6. Data from Statistics Norway (2006)
7. Average Norwegian data provided by Kjøstelsen and Lileng (2006), adjusted based on average values from ALLSKOG and information from entrepreneurs
8. Data from Idemat (2001) with adjusted based on Athanassiadis (2000), other emissions as 2b
9. No available data for Norwegian average. An estimate given by Professor Reidar Skaar, Norwegian University of Life Sciences, which is slightly adjusted based on information from entrepreneurs
10. Data from Transportselskapet Nord (a partly owned subsidiary of ALLSKOG providing transport of timber in the region)
11. Data from Statistics Norway (2006b)
12. Consumption of diesel is based on a combination on Ecoinvent data (Ecoinvent Centre 2004) and estimates based on fuel costs in 10 and 11, in addition to consumption data for loading and unloading from Andersen et al. (2001). Percentage of return cargo is estimated based on information from advisor Rune Damm in The Norwegian Haulier's Association and all empty driving is allocated to transport of timber. Emission data is from Ecoinvent, combined with 2d

Appendix F

The importance of land use impact on biodiversity in an assessment of environmental performance of wood products

The Importance of Land Use Impact on Biodiversity in an Assessment of Environmental Performance of Wood Products

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Submitted for publication in The International Journal of Life Cycle Assessment

Abstract

Background, aim and scope: Land use and changes in land use have a significant impact on biodiversity. Still, there is no agreed upon methodology for how this impact should be assessed and included in LCA. This paper presents a methodology for including land use impact on biodiversity in Life Cycle Impact Assessment and provides a case example from forestry in Norway.

Materials and methods: The proposed methodology is based on a framework developed within the UNEP-SETAC Life Cycle Initiative. The methodology applies indirect measures on biodiversity based on knowledge on what key factors are important for maintaining biodiversity in a boreal forest. These are used to construct an index on Conditions for Maintained Biodiversity. In addition the intrinsic quality of an area is assessed on the basis of the Ecosystem Scarcity and Ecosystem Vulnerability. Globally available data on ecoregions are here used. In addition the spatial and temporal impact is assessed based on the annual increment of the forest.

Results: In the case study the ecoregions 'Scandinavian and Russian taiga' and 'Scandinavian costal coniferous forests' and different forestry regimes are compared. Based on the proposed methodology, the intrinsic quality of the Scandinavian costal coniferous forests is estimated to be approximately 40% higher than equivalent for the Scandinavian and Russian taiga. New and improved targets for the key factor 'areas set aside' can also reduce the impact on biodiversity from land use by approximately 20%.

Discussion: The paper presents a new methodology for how land use impacts on biodiversity can be included in LCA. The methodology is based on a proposed framework and the results from the case study show that the methodology is capable to distinguish between different forestry regimes and forestry in different ecoregions. The data used are readily available, but more research is needed to scale the proposed key factors and also include new key factors. It is at present not possible to validate the size of the differences.

Conclusions: The importance of land use impact on biodiversity is indisputable and this should be included in LCA. The proposed methodology is developed within a framework developed within the UNEP-SETAC Life Cycle Initiative and provides a methodology demonstrated to be able to distinguish between both similar activities in different ecoregions and different management practices within one ecoregion.

Recommendations and perspectives: More work is needed to establish a methodology for land use impact on biodiversity in LCIA and due to the importance this should be a prioritized task. The proposed application of indirect indicators to assess impact on biodiversity from land use changes in LCIA should be further explored, but the proposed methodology can already be applied with globally available data on ecoregions. The challenge is to develop sound key factors for the relevant ecosystems.

Keywords: biodiversity; forestry; key factors; land use impacts; land quality; LCA; LCIA

Introduction

There is no doubt that loss of biodiversity is one of the largest environmental problems, if not the largest (Diaz and Cabido 2001). The main reason given for loss of biodiversity is changes in land use and a consequential unavoidable loss of habitats (Pimm et al. 1995; Chapin et al. 1998; Müller-Wenk 1998; Chapin et al. 2000; Sala et al. 2000). Still, there is no agreed upon method how loss of biodiversity due to land use is to be included in life cycle assessments (LCA) (Milà i Canals et al. 2006a) and it is even debated if this should be done at all (Udo de Haes 2006).

The land use impact on biodiversity is in particular important when extraction of raw materials originating from land extensive activities is assessed. Forestry as the origin for wood based products is a striking example. In Europe, the forested areas have increased with more than 9 millions hectares during the last decade (UNEP 2002), but most of the natural forest vegetation is transferred to agricultural and urban areas, and most of what is left is strongly influenced by forestry and other human activities (Angelstam 1998; Larsson 2001).

Maintenance of biodiversity is an urgent issue for forestry operations (Angelstam 1998). In Norway, almost half of the species in the Norwegian Red List are forest living species (The Directorate for Nature Management 1999) and only 2.9 % of the forested area in the country can be classified as undisturbed by man (Hytteborn et al. 2005). Even though only a few species are known to be extinct, present forestry practice has given an extinction debt, i.e. species that are still present but are likely to go extinct in a not too far future due to present pressure (Angelstam 2001; Hanski and Walsh 2004). The extinction debt is not estimated in Norway, but is assumed to sum up to approximately 1000 forest living species in Finland (Hanski and Walsh 2004). As many as 50% of all species in Norway depending on dead wood are threatened (Framstad et al. 2002).

Loss of biodiversity is probably the major single environmental problem caused by the forestry sector (Seppälä et al. 1998). This aspect should thus be included in LCA of forest products to obtain a more holistic picture of the environmental impact of such products and enable comparison to other products (Lippke et al. 2005; Milà i Canals et al. 2006b). This would be in accordance with ISO 14040: 2006 that states that all aspects of natural environment must be considered. Land use is explicitly mentioned in ISO 14044: 2006.

In this paper the first outline of a methodology for assessing biodiversity aspects related to land use in forestry operations in a boreal forest is presented. One important characteristic with the proposal is the ability to distinguish both between different forestry regimes and forestry at different locations. It is believed that the presented methodology could be applicable to other ecosystems as well. This paper is also a contribution to the debate on how indicators for assessing land use impact on biodiversity should be framed (cf. Milà i Canals et al. 2006b). The first attempt to apply the methodology on a case study of logging of spruce (*Picea abies*) in Norway is presented.

1 Measures of biodiversity applicable for LCA

Biodiversity is a concept with a wide content, and in the Convention on Biological Diversity it is stated that '*Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*' (UNEP 1992). In spite of this, the most frequently used measure on biodiversity is number of species. Gaston (1996) claims there are four obvious reasons. First, species richness is thought by many to capture much of the essence of biodiversity, and many authors use the two terms more or less as synonyms. Second, species richness as term is widely understood. Third, species richness is considered in practice to be a measurable parameter in contrast to

biodiversity as stated in the definition, and fourth, much data on species richness already do exist.

It is suggested that the low focus on conservation of biodiversity in decision making, is due to the fact that biodiversity is hard to quantify (OECD 2002). The low focus on loss of biodiversity as a consequence of land use in LCA is probably due to this. Nevertheless, several attempts have been made to include land use in LCA (see Milà i Canals et al. 2006a for references), but proposed indicators are in most cases not checked with a consistent framework (Milà i Canals et al. 2006a).

Some of the proposed methodologies, such as the Biotope Method (Kyläkorpi et al. 2005), are at present too coarse to distinguish between different management regimes. Another severe problem with many of the methodologies proposed is that they are based on assumptions that probably are invalid. This relates in particular to the proposal of vascular plant diversity as an indicator for biodiversity in some of the methods, e.g. the SPEP-method (Köllner 2000) incorporated in Eco-indicator 99 (Goedkoop and Spriensma 2001). The SPEP-method is a praiseworthy proposal, but might turn out as a dead end due to the underlying assumptions.

The first problem is that vascular plant diversity is an inappropriate indicator for biodiversity. An overwhelming number of studies show no correlation between species richness in one taxonomic group and species richness in other groups (i.e. Prendergast et al. 1993; Hengeveld et al. 1995; Gaston 1996; Dobson et al. 1997; Lawton et al. 1998; Molau and Alatalo 1998; Chapin et al. 2000; Larsson 2001). Lawton et al. (1998) conclude that on average only 10-11 percent of the variation in species richness of one group can be predicted by the change in richness of another group.

Also, if ecological changes are to be measured through registration of changes in species composition, other groups of species are more useful. Just to mention a few; Molau and Alatalo (1998) have shown that bryophytes are better indicators than vascular plants for effects of global warming, Hilmo and Holien (2002) have shown that lichens are useful indicators for edge effects and fragmentation, and Bongers (1990) has shown that nematodes are useful indicators for changes in soil conditions.

A third problem is that it is not only important what species that are present, it is also important to maintain areas that enable invasions. A focus on presence or absence of different species is more or less consciously based on an assumption of static conditions in the ecosystems. This is simply not true, cf. the equilibrium theory of island biogeography (MacArthur and Wilson 1967), the metapopulation concept (e.g. Schemske et al. 1994), and the natural disturbance hypothesis (Connell 1978). In addition, there might be a tremendous time lag between the change in conditions¹ and the actual change in species composition. Saunders et al. (1991) emphasise that this time lag might be on several hundred years for long lived species, such as long-lived trees, and the result is an extinction debt (cf. Angelstam 2001; Hanski and Walsh 2004) that is difficult to assess. In some countries extinction rates due to different land use impacts are available (see Müller-Wenk 1998; Köllner 2000), but for most areas of the world this is not the situation.

The abundance of the species present is also of interest. Chapin et al. (2000) stress the importance of abundance for ecosystem functioning, and Didham et al. (1996) show that even if a species is present in an ecosystem, the ecosystem might function as if the species is absent if the abundance falls under a certain level. Hengeveld et al. (1995)

¹ The term 'condition' is here used for all environmental factors influencing the species probability to survive. Thus, it includes both what in ecological terms is recognized as conditions (abiotic environmental factors which varies in space and time, and to which organisms are differentially responsive, cf. Begon et al. 1986) and resources (all things consumed by an organism, cf. Tilman 1982).

emphasise that the number of species alone is not enough to evaluate diversity, but also i.e. evenness should be taken into account.

The underlying cause of these problems is the fact that biodiversity as defined by UNEP (1992) cannot be measured directly. Several authors within the field of biodiversity have thus started to focus on indirect measures and focus on conditions known to be important for biodiversity. Hansson (2000) states that a biodiversity indicator might as well be a structural component, a process, or some other feature of the biological system that ensures maintenance or restoration of the most important aspects of biodiversity when present. From this point of view Larsson (2001) focus on the key factors affecting biodiversity. For forests, this means to recognise that biodiversity is dependent on the structure of stands and landscapes, the forest formatting trees and the management and disturbance regimes they experience. Larsson (2001) identifies in total 17 key factors for assessing biodiversity in European forests. These are used as a basis for the proposed methodology in this paper.

2 Proposal of methodology

Three different aspects must be assessed to quantify the land use impact on biodiversity (cf. Milà i Canals et al. 2006a). First, a quality measure must be established and assessed. Second, the area affected must be recognized and third, the duration of the impact. This is shown schematic in Figure 1. Due to changes in land use at time t_1 the quality declines from Q_0 to Q_1 . At t_2 the land use stops and the area is left for relaxation, and at t_3 the quality has been restored to Q_0 . The changes in quality are given by the bold line and the total impact is given by the shaded volume. Other outcomes are possible, e.g. different quality at t_0 and t_3 , gradually changes in quality between t_1 and t_2 etc. (see Lindeijer et al. 2002).

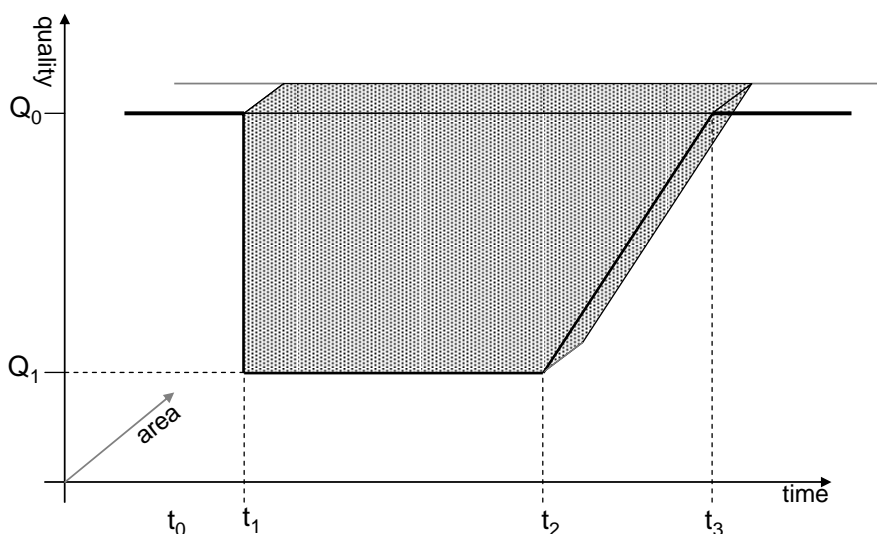


Figure 1 – Changes in land quality and total impact due to land use changes (adapted from Lindeijer et al. 2002)

2.1 The quality of an area in terms of biodiversity

In the absence of possibilities to measure biodiversity directly, it is here proposed to measure biodiversity indirectly by means of three factors:

- the Ecosystem Scarcity (*ES*)
- the Ecosystem Vulnerability (*EV*)
- the Conditions for Maintained Biodiversity (*CMB*)

Quality (Q) at a given location and time can be assessed as a product of these three factors:

$$Q = ES \times EV \times CMB \quad (1)$$

2.1.1 Ecosystem Scarcity (ES)

This indicator was introduced by Weidema and Lindeijer (2001). The rationale for using ES as an indicator is that biodiversity linked to scarce ecosystems normally would be more vulnerable than biodiversity linked to more widespread ecosystems. The populations will in general be smaller and the extinction risk due to stochastic processes higher. Weidema and Lindeijer (2001) express the indicator as the inverse value of the potential area of the structure² (A_{pot}), resulting in the equation

$$ES = \frac{1}{A_{pot}} \quad (2)$$

This indicator can be used at different levels (biome, landscape, vegetation type etc.) depending on data availability and purpose of the study. Weidema and Lindeijer (2001) use the indicator at biome level, but data on 825 ecoregions³ are now globally available. Since the analysis can be performed at different levels, the indicator score should be normalized following the equation

$$ES = 1 - \frac{A_{pot}}{A_{max}} \quad (3)$$

where A_{max} is the potential area of the most widespread structure at the relevant level. The structure with the highest scarcity then gets a score close to 1, while other structures have scores relative to this. This normalization follows a linear relationship between potential area and biodiversity quality. Other proposals are of course possible but will not be discussed here.

To be able to use this as an indicator for forestry at different sites, it is necessary to have area factors for different forest types. Comprehensive classification systems for vegetation types exists (e.g. Fremstad (1997) for Norway and Pålsson (1998) for the Nordic countries), but at present data on potential distribution are in general absent and data on ecoregions are used.

2.1.2 Ecosystem Vulnerability (EV)

Ecosystem vulnerability (EV) is introduced as an indicator to give information about the present total area pressure to an ecosystem type and relate the existing area of an ecosystem to the potential area. The rationale is that the more of an ecosystem that is lost, the more valuable is the remaining areas. This is a consequence of the species-area relationship (MacArthur and Wilson 1967). As with the previous indicator, this can be used at different levels depending on data availability and purpose of the study.

Peter et al. (1998)⁴ propose the formula

$$EV = \frac{1}{1 - \text{fraction lost}} \quad (4)$$

while Weidema and Lindeijer (2001) propose another formula given by

² The term structure is here used to indicate that this could be used at different levels; biome, landscape, ecosystem, vegetation type etc, and structure is used as a level independent term

³ As defined by Olson et al. (2001) including 867 ecoregions, while data on 825 is available on <http://www.worldwildlife.org/wildfinder/>

⁴ Originally named the 'area factor' by Peter et al. (1998)

$$EV = \left(\frac{A_{exi}}{A_{pot}} \right)^{z-1} \quad (5)$$

A_{exi} is the existing area of the structure and A_{pot} is the potential area. z varies between different ecosystems (Hengeveld et al. 1995), but are often given the value 0.25 (MacArthur and Wilson 1967).

Both these proposed formulas give the range $[1, \infty]$ and must thus be normalized. One possibility is to normalize in the same manner as with ES and give the most vulnerable structure the score 1 and other structures scores relative to this.

However, data on EV is hard to find on an appropriate level and in most cases it will be necessary to use proxy values. Information on conservation status can be used and is often readily available. Fremstad and Moen (2001) classify Norwegian vegetation types (cf. Fremstad 1997) in the same scale as is used in species red lists. World Wildlife Fund provides a three grade scale on conservation status for the ecoregions of the world⁵. In the absence of better data, this is made use of and ecoregions with the conservation status critical are given the score 1.0, ecoregions with status vulnerable are given the score 0.5 and intact ecoregions are given the score 0.1.

2.1.3 Conditions for Maintained Biodiversity (CMB)

The indicators on Ecosystem Scarcity and Ecosystem Vulnerability give information on the intrinsic biodiversity value of an area, while the indicator on Conditions for Maintained Biodiversity (*CMB*) gives information on the present conditions for the biodiversity in the area; is it intact, or is it reduced. Under some circumstances it might even be improved, which will be described below.

CMB is in fact an index composed by indicators known to be important for biodiversity in the particular structure. *CMB* must therefore be ecosystem specific since the key factors (cf. Larsson 2001) are different in different ecosystem. The number of key factors will also vary. Hence, it is here proposed to assess *CMB* as

$$CMB = 1 - \frac{\sum_{i=1}^n KF_i}{\sum_{i=1}^n KF_{i,max}} \quad (6)$$

where KF_i are the different key factors identified. Larsson (2001) suggests using a four level scale for the status of the key factor:

- 0 – no impact
- 1 – slight impact
- 2 – moderate impact
- 3 – major impact

In addition, the relative importance of the key factors for biodiversity must be determined. It is here proposed to use the same scale $[1, 3]$ and multiply the status score with this factor. As a consequence, an indicator with a slight impact have the scale $[0, 1, 2, 3]$ while an indicator with a major impact have the scale $[0, 3, 6, 9]$. $KF_{i,max}$ is then the maximum score for KF_i , giving *CMB* the range $[0, 1]$ independent of the number of included key factors. A *CMB* score on 1 indicate that the biodiversity in the area is not affected, while a score on 0 indicate that the land use is devastating for the biodiversity. Larsson (2001) has proposed a range of possible key factors for European forest and an example of how this can be used is presented below.

⁵ see <http://www.worldwildlife.org/wildfinder/> and Olson and Dinerstein (1998)

When the framework proposed by Milà i Canals et al. (2006a) is used, the quality of an area before a land use intervention (at time t_0 in Figure 1), is given as

$$Q_{t_0} = ES \times EV \times CMB_{t_0} \quad (7)$$

while the quality of the same area after the land use intervention (at t_1 in Figure 1), is given as

$$Q_{t_1} = ES \times EV \times CMB_{t_1} \quad (8)$$

It follows from this that if the area is undisturbed by human activities before the intervention, Q_{t_0} is simply the product of ES and EV . It also follows from this that the land use impact might be positive if $CMB_{t_1} > CMB_{t_0}$, e.g. as a result of restoration or improved land management.

2.2 Spatial and temporal impact

The duration of the intervention in time and space must be assessed together with the quality difference. In a forest, this is defined as the time and area necessary for regrowth of the amount of timber harvested. This means that if the annual increment is 5 m³/ha, 0.2 ha×y is needed to provide 1 m³ of logged wood.

3 Land use impact in a case study of forestry on Norway

A life cycle assessment of forestry operations in Norway is presented in Michelsen et al. (in prep.). The functional unit is 1 m³ round wood logs under bark delivered at the gate of a factory. Forestry and silviculture operations, such as seedling production, planting, soil scarification, cleaning of unwanted vegetation, logging and construction of forest roads are included. The assessment is performed in cooperation with ALLSKOG BA which represents the majority of forest owners in the western and northern parts of Norway and the analysis is valid for this area (see Michelsen et al. in prep. for details). However, only 'traditional' impact categories are included. In the present paper land use impact on biodiversity in this case is assessed following the proposed methodology.

3.1 Impacts on quality

3.1.1 Intrinsic quality score

Since the case study is of logging of spruce, the logging can take place in two different ecoregions; in region PA0608 Scandinavian and Russian taiga or in the less distributed PA0520 Scandinavian coastal conifer forests (cf. Olson and Dinerstein 1998). The distribution of PA0608 is 2 156 900 km² while the distribution of PA0520 is 19 300 km². The ecoregion with the largest distribution is PA1327 Sahara desert with 4 639 900 km². The conservation status for both PA0608 and PA0520 is critical. All data are from World Wildlife Fund's Wildfinder. These data are used to calculate $ES \times EV$ for the two ecoregions as shown in Table 1.

Table 1 – Intrinsic quality of the two relevant ecoregions

Ecoregion	Potential area (A_{pot})	Ecosystem Scarcity (ES)	Conservation status	Ecosystem Vulnerability (EV)	$ES \times EV$
PA0520	19 300	0.9958	1 – critical	1.0	0.9958
PA0608	2 156 900	0.5351	1 – critical	1.0	0.5351

3.1.2 Definition and assessment of Key Factors

Larsson (2001) identifies in total 17 key factors for biodiversity in European forests. Not all are of equal importance for boreal forests. Similar lists are proposed by others (e.g. Stokland et al. 2003). The main problem with most of them is the scaling, i.e. how much of a particular key factor is needed for the scores from 0 (no impact) to 3 (major impact) and what is the relative importance of the key factors. As mentioned, the advantage with

the proposed methodology is the possibility to start with a few key factors and subsequently prolong the list. According to Hanski and Walsh (2004) the two most important factors for decline of biodiversity in boreal forests are the reduced amount of decaying wood and loss of the most diverse forest formations. Based on this fact and combined with present data availability, three key factors are here included in a first proposal:

- amount of decaying wood
- areas set aside
- introduction of alien tree species

It is important to underline that these key factors are not independent of each other. In particular, the size of the areas set aside has consequences for the targets of the others. However, if the areas set aside should be sufficient to maintain the biodiversity within forests, it would probably be necessary to set as much as 60% of the areas aside (Framstad et al. 2002). This is not realistic, so conservation of biodiversity must be based on both areas set aside and sustainable forestry (Bengtsson et al. 2000; Framstad et al. 2002). The proposed threshold values for the other key factors must hence be seen in relation with the values for areas set aside.

The species' probability to survive changes in the amount of suitable habitats might in principle follow two response curves. First, there is a linear relationship shown as response type I in Figure 2. However, in most cases there is a non-linear relationship where the amount of suitable habitats has to exceed a threshold value for the species to be able to maintain or establish a viable population (Hanski and Walsh 2004). This is shown as response type II in Figure 2. It is believed that at least threatened species follow this response curve (Hanski and Walsh 2004). It is also possible to combine these two and use a linear relationship with a threshold value (response type III in Figure 2). Identification of the most relevant response curve is one element in defining the severity of the key factors at different impact levels.

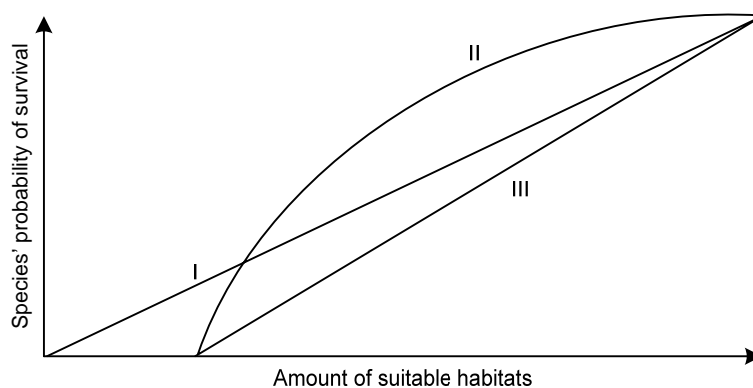


Figure 2 – Three possibilities for species' probability to survive changes in habitat quality (see text for details)

Amount of decaying wood

It is well documented that present level of dead wood in managed boreal forests is far below what is found in undisturbed boreal forests, and Siitonen (2001) estimates that the decline is as high as 90-98 percent due to forestry. In Norway the average in productive forests are at time being 8.3 m³ dead wood/ha (Hobbelstad et al. 2004).

There are different opinions on how much decaying wood that is necessary to prevent extinction of species depending on dead wood, but an estimate on 20 m³/ha in managed forests (Hanski and Walsh 2004) seems to be a minimum. Framstad et al. (2002) claim

that present level of dead wood might give a reduction on 50% of organisms depending on dead wood in Norway. A first proposal of impact on this key factor is given in Table 2.

Table 2 – Proposed scale for the key factor 'Amount of decaying wood'

Amount of decaying wood	Impact
> 20 m ³ /ha	0 – no impact
10-20 m ³ /ha	1 – slight impact
5-10 m ³ /ha	2 – moderate impact
< 5 m ³ /ha	3 – major impact

Areas set aside

Areas set aside are important since it is unlikely that the normal forest dynamics can be preserved within managed forests, such as forest fires, storm felling and browsing. It is also important to preserve the ecosystems capacity to evolve and function also under changed environmental conditions, e.g. climatic changes (Aarts and Nienhuis 1999). It is of course not only the total size of the area that matters; it is important to both have representative areas and large areas (e.g. Framstad et al. 2002). However, if we assume that areas are set aside as a result of a conservation plan, it is possible to assume that this is taken care of and hence only focus on total area as a key factor.

There are conflicting views on how much that is necessary to set aside, but in a combination with more sustainable forestry, there seems to be an agreement that about 10% should be sufficient (Framstad et al. 2002; Hanski and Walsh 2004). In Norway, about 2% of the areas are at present set aside. Half of this is done through establishment of national parks and nature reserves (Framstad et al. 2002), while the second half is a result of the PEFC⁶-standards used by almost all forest owners in Norway (e.g. Sverdrup-Thygeson et al. 2004). A first proposal of impact on this key factor is given in Table 3.

Table 3 - Proposed scale for the key factor 'Area set aside'

Areas set aside	Impact
10%	0 – no impact
6-10 %	1 – slight impact
1-6 %	2 – moderate impact
<1 %	3 – major impact

Introduction of alien tree species

Introduction of alien species are known to have a severe effect on ecosystems (eg. Clay 2003; Eppinga et al. 2006), and when the forest formatting tree is changed, the whole ecosystem is affected (Cushman et al. 1995; Larsson 2001; Stokland et al. 2003). Different tree species produce e.g. litter of different amount and quality, and provides different kinds of shelter etc. Stokland et al. (2003) distinguish between local introductions and long distance introductions. A relevant example on the first is primarily introduction of *Picea abies* in *Betula*-stands in the western and northern parts of Norway, while examples on the second are introduction of *Picea sitchensis* or *Larix spp.* in spruce (*Picea abies*) forests.

Values given in Stokland et al. (2003) show that introduced tree species in Norway constitute 2.4% of the forests. Impact values are here not well developed, but a first proposal on this key factor is given in Table 4.

⁶ Programme for the Endorsement of Forest Certification schemes, <http://www.pefc.org/>

Table 4 - Proposed scale for the key factor 'Percentage of alien tree species cover'

Percentage of alien tree species cover	Impact
0%	0 – no impact
0-10 %	1 – slight impact
10-25 %	2 – moderate impact
< 25 %	3 – major impact

Assessment of Conditions for Maintained Biodiversity

The three proposed key factors are not weighted to each other, but are assumed to have the same impact given the same score. Following equation 6 the present average value for *CMB* in conifer boreal forests in Norway is thus

$$CMB = 1 - \frac{\sum_{i=1}^n KF_i}{\sum_{i=1}^n KF_{i,max}} = 1 - \frac{2+2+1}{3+3+3} = 0.44 \quad (9)$$

Several other key factors could be considered, e.g. cutting regime, tree species composition (in particular amount of deciduous trees in boreal coniferous forests), regeneration methods (area left for natural regeneration), ditching, forest road density and amount of large trees (cf. Larsson 2001; Stokland et al. 2003). These are however not included in this first proposal.

3.2 Spatial and temporal impact

The annual increment of conifer trees in productive forests is on average 2.3 m³/ha in Norway (Stokland et al. 2003). Thus, for the production of the functional unit of 1 m³, 0.435 ha×y is needed.

3.3 Total impact of land use

Milà i Canals et al. (2006a) propose to use the dynamic reference situation for assessing quality changes. In the case study presented in this paper, it is assumed that the forest already is altered due to centuries of forestry, and the land use in the case study represent a postponement of the natural processes that eventually will bring the area back to its natural state and quality (= *ES*×*EV*).

Further, it is assumed that the relaxation time is equal to the rotation time in the forest. The total impact caused by land use can then be assessed as shown in Figure 3a. The time and area needed for one rotation period (*t_{rot}*) is as shown above 0.435 ha×y. The quality due to the forestry operations (assuming forestry in ecoregion PA0608) is given by

$$Q_{t1} = ES \times EV \times CMB_{t1} = 0.535 \times 1 \times 0.44 = 0.235 \quad (10)$$

This represents a postponement of a potential quality after relaxation, given by

$$Q_{rel} = ES \times EV = 0.535 \times 1 = 0.535 \quad (11)$$

The quality difference (ΔQ) is thus 0.3 for a duration of 0.435 ha×y, giving a total impact of land use on biodiversity expressed as 0.131 ΔQ ×ha×y.

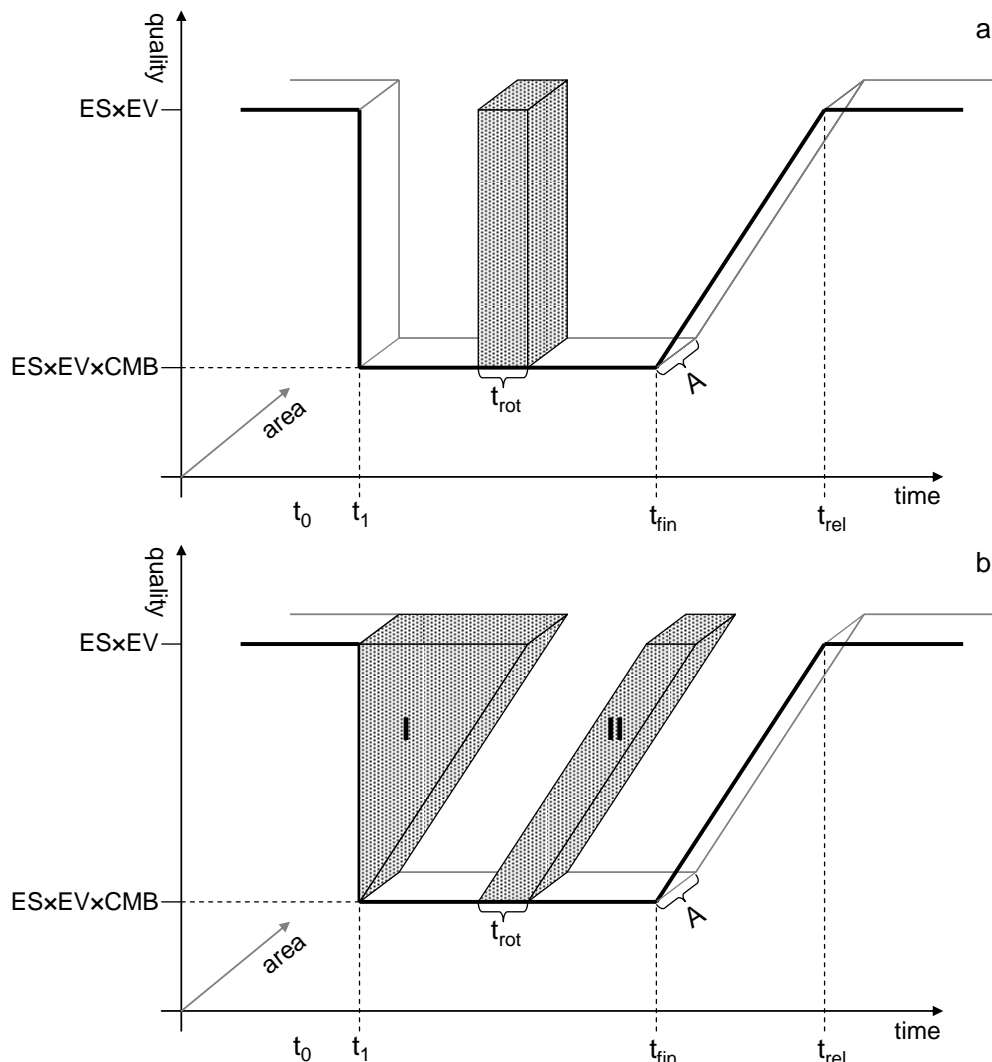


Figure 3 – A graphical interpretation of land use impact on biodiversity (see text for details)

The proposed assessment visualized in Figure 3a might be an underestimation of the actual impact. The temporal impact is assumed to be equal to the time needed for the forest to regrow (t_{rot}), which might be an underestimation of the time needed for biodiversity to recover (cf. Duffy and Meier 1992; Müller-Wenk 1998). According to Milà i Canals et al. (2006a) the dynamic reference situation should be used to assess the land use impact and the total impact from land use should therefore be calculated as 'II' in Figure 3b. It is not made any attempt here to evaluate if this difference is significant for a rather slow growing forest, but this could be of major importance, particular in other ecosystems.

In addition, no attempt to include the transformation impact is done. The transformation impact is caused by the initial transformation of an area from an undisturbed forest to a managed forest, and the total transformation impact is shown as 'I' in Figure 3b. However, this impact must be allocated to all timber logged in this area, and if there have been many rotations, this will with time become insignificant. It is not made any attempt to verify if this is the situation in this case.

3.4 Sensitivity of the methodology

In the previous example average values for forestry in ecoregion PA0608 are used. Two different cases are likely to occur as well. First, the logging can be situated in ecoregion PA0520. Here, a slightly higher annual increment can be assumed (Stokland et al. 2003)

and the value 3.0 m³/ha is used. The area and time needed to provide 1 m³ of wood is then 0.333 ha×y.

It is also possible to assume that the areas set aside are increased to slightly above 6%. This can either be a result of implementing the Swedish FSC⁷-standard (The Swedish FSC Council 2000) instead of the Norwegian PEFC-standard (Living Forests 1998), intensified demands in the PEFC-standard⁸, or as a result of increased areas of forest reserves following the minima recommendation of Framstad et al. (2002). The impact on this key factor might thus decline from 2 to 1 (cf. equation 9).

These two options can of course be combined and the results are presented in Table 5.

Table 5 – Differences in land use impact on biodiversity due to different ecoregions and changes in forestry regime

Case	<i>ES×EV</i>	<i>CMB_{t1}</i>	ΔQ	ha×y	$\Delta Q \times \text{ha} \times \text{y}$
PA0608	0.535	0.44	0.300	0.435	0.131
PA0608, 6% set aside	0.535	0.56	0.235	0.435	0.102
PA0520	0.996	0.44	0.558	0.333	0.186
PA0520, 6% set aside	0.996	0.56	0.438	0.333	0.150

3.5 Relative importance of land use on biodiversity

It is controversial to compare different impact categories in LCA and it will not here be suggested weighting factors for comparing land use to other impact categories. However, the importance of this category seems indisputable. Intuitively, this must be the situation since land use is the single most important cause for loss of biodiversity (cf. Pimm et al. 1995; Chapin et al. 1998; Müller-Wenk 1998; Chapin et al. 2000; Sala et al. 2000), which again might be the largest environmental problem (cf. Diaz and Cabido 2001). Seppälä et al. (1998) have concluded that loss of biodiversity is the major environmental problem caused by forestry.

In Eco-indicator 99 there are proposed weighting factors that enable comparison of the impact of land use to ecosystem quality to other impact categories (Goedkoop and Spriensma 2001). As an example, acidification is given the weight 1.04 PDF ym²/kg SO_x (see Goedkoop and Spriensma 2001). In the presented case, the total emissions are 0.113 kg SO_x (Michelsen et al. in prep), giving an impact of 0.118 PDF ym².

In comparison, Hanski and Walsh (2004) states that 1000 of the 20 000 forest living species in Finland are threatened by extinction due to present forestry practice, giving a PDF on 0.05. Assuming that the number for Norwegian forestry is equivalent, this number can be multiplied with the necessary space and time needed for logging 1 m³ of timber. The impact will then be 217.5 PDF ym², an impact more than 1800 times higher than the impact due to acidification in this particular case.

4 Discussion

In this paper a new methodology for how land use impacts on biodiversity can be included in LCA-studies is presented. The methodology is also used on a case study of logging in Norway.

The proposed methodology is shown to distinguish both between different forestry regimes and forestry in different ecoregions (Table 5). Logging in PA0520 Scandinavian coastal conifer forests represents approximately 40% increase in the impact compared to logging in PA0608 Scandinavian and Russian taiga according to the proposed

⁷ Forest Stewardship Council, <http://www.fsc.org/>

⁸ The PEFC-standard is under revision and this is one of the issues debated, cf. <http://www.nationen.no/naeringsliv/article2293073.ece> (article in the Norwegian newspaper 'Nationen')

methodology. The figure makes sense (cf. Framstad et al. 2002), but the size of the difference can at present not be validated. Also, the hypothetical increase of areas set aside reduces the impact with about 20%. More work is needed to verify and adjust these results, in particular on the scale of the key factors. Nevertheless, this paper represents a proposal for how such methodologies can be developed and what indicators on biodiversity that should be further investigated.

Selection and scaling of key factors are critical steps. In this paper three key factors are introduced. These are assumed to be among the most important for biodiversity in boreal forests (cf. Hanski and Walsh 2004), but there are obvious others (Larsson 2001; Stokland et al. 2003). It is probably not possible to determine the scale and mutual importance of these on a purely scientific basis with present knowledge (cf. Bennett and Adams 2004), but expert judgements are often seen as a good approximation (Seppälä et al. 1998; Scholes and Biggs 2005).

When different forestry regimes are to be compared, it must be determined what geographical range that is to be used for assessing the selected key factors. As an example, within a spruce plantation on the western coast of Norway, the entire tree cover will in most cases consist of introduced *Picea abies*. However, in the landscape as a whole, the plantations constitute a rather small proportion of the area. In this paper, average values for Norway are used, but in most cases, it will probably be more appropriate to set the scores according to the state within the borders of a decision-making unit. This might be a single forest owner, or as in the study described in Michelsen et al. (in prep.), within the borders of a forest owner association.

The temporal scale is also assessed in a simplified way in this paper. In Figure 3 the solid line can be interpreted as the quality difference over the time the forestry is performed in the area (cf. Milà i Canals et al. 2006a). The first forestry operations take place at t_1 and forestry is carried out until t_{fin} when the area is left for relaxation. The relaxation is completed at t_{rel} . The temporal impact is here assumed to be equal to the time needed for the forest to regrow (t_{rot}), but as pointed out, this is most likely an underestimation. The significance of this simplification must be evaluated and more accurate relaxation times are needed. The potential significance of the transformation impact must also be further investigated (cf. Figure 3b).

The proposed methodology assesses the changes in quality as given by the solid line in Figure 3. However, the actual quality in terms of biodiversity will in most cases change more gradually, e.g. as a result of long lived species that are able to survive for long periods after the ecosystem conditions are changed. This is referred to as an extinction debt. This methodology does thus not measure the present quality, but the future quality following the present management regime.

The intrinsic quality assessment is sensitive to the size of the ecoregions. If, for instance, a ecoregion is split in several new ecoregions, the assessed quality of the areas within them will increase significantly. It must thus be assumed that the subdivision of ecoregions done by Olson et al. (2001) is done on a consistent basis. In the future, it might also be possible to use finer scales, e.g. the vegetation types identified by Fremstad (1997) and Pålsson (1998). The consistency of the subdivision into different structures is obviously a critical part of the proposed methodology.

A problem that is not taken into consideration at this point is how seminatural vegetation should be treated. If the assessment of the two factors Ecosystem Scarcity and Ecosystem Vulnerability is applied strictly as proposed, seminatural vegetation types will be regarded as without any value since their potential area without human influence by definition is zero. It is of course possible to argue that only natural occurring vegetation should be protected and maintained, but this is not a common opinion and will undoubtedly result in extinction of a range of species adapted to these habitats through

millennia. Studies indicate that even in temperate forests there are species that are adapted to forestry and hence would become threatened if forestry stopped (Decocq et al. 2004). This problem must be addressed if this methodology is to be used also for seminatural vegetation.

5 Conclusion

The importance of land use impact on biodiversity is indisputable and this should be included in LCIA, in particular when raw materials, such as wood, originate from land extensive activities. However, there is no agreed upon methodology for how this should be done, and a debate on the topic is crucial.

The proposed methodology provides a possibility to distinguish between land use impact from forestry both related to different forestry regimes and forestry at different locations. The methodology is proposed within the framework provided by Milà i Canals et al. (2006a), which is an outcome of the UNEP-STEAC Life Cycle Initiative. The scale of the differences must however be subject to further investigations, probably based on expert judgements. More research is also needed to see if this methodology can be transferred to other land use interventions than forestry in boreal forest.

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