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

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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.

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6.1 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 have 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.

6.2 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 6.1 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 6.2) 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.

6.3 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 6.1 shows the main components of the chair. In addition, 3 kg cardboard is used for packaging. Figure 6.1 shows the main components and materials used in the chair.

Table 6.1 Main components of the chair used in the case study

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

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.

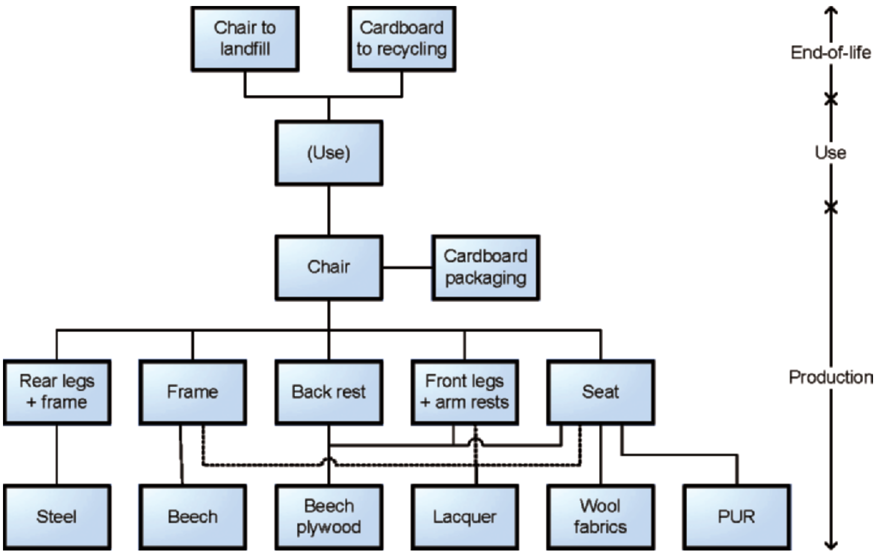

Figure 6.1 Main elements in the extended supply chain of the chair used in the case study

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:

$$\frac{(LC+PC)\times 1.15}{0.7}\times k \tag{6.1}$$

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

[1] 1€ ≈ 7.90 NOK (August 2005)

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*).

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 6.1. 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:

$$\frac{(LC + PC) \times 1.15}{0.7} \times k + (aLC + aPC) \times 1.15 \tag{6.2}$$

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 Norcargo (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 6.1, 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 6.2.

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.

6.4 Results

The changes in value and environmental performance for the different scenarios are shown in Table 6.2. The same values are presented graphically in Figure 6.2.

Table 6.2 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	30	B*: 64
C – exclusion of PUR	- 240	- 144	-
D – dismantling and recycling	- 330	30	33
E – take-back and reuse	- 280	4	-142
BE – combination	- 330	24	-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.

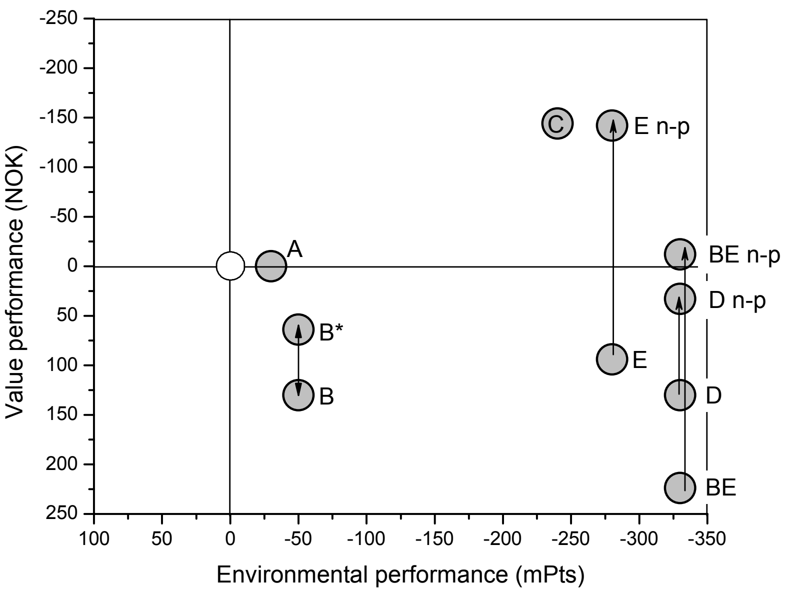

Figure 6.2 Changes in eco-efficiency in the different scenarios (see text for details)

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.

Table 6.3 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

The relative costs of the various alternatives for environmental improvement differed considerably. This is shown in Table 6.3, 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 6.2).

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.

6.5 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 6.2, 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 6.3). 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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