Total Cost as Suitable Indicator in Realization of More Sustainable Product Life Cycles Regarding Utilization of Natural Abiotic Resources

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Abstract

A total cost accounting approach was used to analyse the suitability of copper and aluminium as winding material for transformers with respect to sustainability, using available data from the Ecoinvent database. It could be concluded that repeated recycling of metal is a necessary requisite to obtain product element life cycles exhibiting a high degree of sustainability. Using cost data for energy and materials and reasonable assumptions about costs for labour and interest for the metal supplier and the product manufacturer, the copper alternative turns out to be the better choice, especially when the expected increase in the prices of energy, copper, and aluminium during life cycle is taken into account.

From the study it could be concluded that the total cost accounting approach would be a valuable tool for assessing the degree of sustainability of a product life cycle, in particular, regarding use of natural abiotic resources such as metals.

Keywords: Product life cycle, Total cost accounting, Sustainability, Life cycle analysis, Life cycle cost assessment, Product design, Material selection

1. Introduction

Sustainable growth has become the ultimate goal for all kinds of future development, at least in a political sense, but considering its meaning, the picture is quite diverse: (a) meeting the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987), (b) improving the quality of human life while living within the carrying capacity of supporting ecosystems (IUCN, 1991), and (c) economic growth that provides fairness and opportunity for all the world's peoples, not just the privileged few, without further destroying the world's finite natural resources and carrying capacity (Pronk & Haq, 1992). Definitions such as these are not particularly clear from a scientific point of view, and for concrete decisions concerning sustainability it is necessary to have more explicit measures.

From the viewpoint of how sustainable a given product or process is, Bakshi and Fiksel (2003) have proposed the following perhaps more useful definition: A sustainable product or process is the one that constrains resource consumption and waste generation to an acceptable level, makes a positive contribution to the satisfaction of humans needs, and provides enduring economic value to the business enterprise. A product can also be seen as a form of produced capital, which is an essential element in the definition of sustainable development made by the Joint UNECE/OECD/Eurostat Working Group on Statistics for Sustainable Development (2008): Sustainable development ensures non-declining per capita national wealth by replacing or conserving the sources of that wealth; that is, stocks of produced, human, social, and natural capital.

If sustainability is first considered in the sense of an acceptable or minimum consumption of natural resources and waste generation, the concept of exergy consumption (see e.g. Rant, 1956; Wall, 1977; Szargut, 1978; Szargut, Morris & Steward, 1988) might be of interest. Exergy consumption is used as a measure of deterioration in quality of energy and material in thermodynamics. It may therefore, at least hypothetically, be used for the purpose of measuring degree of sustainability of a product life cycle, or in a wider sense, the technical system it is a part of. At a certain temperature, the exergy content of the materials of the product represents the energy of the materials that can be converted into work (electrical or mechanical) relative to some defined reference state. The most relevant reference state would be the chemical elements of the material in their native minerals at concentrations corresponding to their occurrence in the earth's crust as has been proposed by Szargut (1978). The exergy content of a metal is, when using this reference state, a measure mainly of how far from chemical equilibrium the metal is in its reaction with atmospheric oxygen. The exergy content of a material in general, therefore, does not stand for a value in the economic sense of how useful the material is for a given technical application. However, the exergy content of a material represents an economic value when considering the efforts needed to produce it from its raw materials. The exergy content of metallic aluminium is consequently higher than that of metallic copper, although the latter is a more noble and also a rarer element in the earth's crust compared to the former.

Moreover, a product element made of metallic aluminium can function properly as a construction material for a sufficient service time only because its surface is chemically passive. The presence of a surface layer of oxide prevents further oxidation and deterioration of the bulk metal. The formation of the oxide layer will decrease the exergy content of the aluminium construction element. However, the existence of the oxide layer is a prerequisite for the product element to be of any economic value for construction purposes.

The two examples clearly point to the difficulties in using exergy as an indicator for sustainability of a product life cycle.

In Life Cycle Impact Assessment (LCIA), a technical system is generally considered in a broader sense than maintaining the earth's natural resources at a tolerable level. In the Environmental Priority System (EPS) method (Steen, 1996) the point of departure is the five safeguard objects: (a) stocks, which is evaluated in terms of future costs for extraction, (b) production, which is evaluated in terms of direct losses, and (c) health, (d) biodiversity, and (e) aesthetics. In the Ecoindicator 99 method (Goedkoop and Spriensma, 1999), the environmental impact is assessed in terms of damage to human health, ecosystem quality, and resources. The result is expressed in terms of an environmental impact indicator, which relates to the yearly environmental load or damage by one average European inhabitant. In both methods attempts are made to estimate cost, that is, willingness to pay in the EPS method, and surplus energy for future extraction of natural resources required in the Ecoindicator 99 method.

The sustainable process method, as used in the EPS method, assumes that in the very long-term perspective raw materials must be extracted from minerals present in average earth-crust rock. In the Ecoindicator 99 method the damage to natural resources is measured in terms of surplus energy. This is defined as the difference between the energy needed to extract a resource now and at some point in the future. However, this approach does not take into account economic values related to present and future prices of natural resources needed for production of useful materials. Forecasting future prices for natural resources is a very complex matter, and large fluctuations in price level with time may occur for a variety of reasons. In the long term, the prices of natural resources will most likely increase substantially, relative to today's prices. Of course, this would have a large impact on future product and product life cycle design, and also affect considerably product cost-effectiveness and profitability, as well as product sustainability.

Besides the environmental quality aspect, the service or function the product represents must be considered from a sustainability point of view. As stated by Bakshi and Fiksel (2003), the product and its functional service must provide a positive contribution to the satisfaction of humans needs. A sustainable product shall also provide enduring economic value to the business enterprise pursuing the product. Another way of expressing this is that the product or the product service it represents can only exist on the market if there is a willingness or ability by customers to pay for it. This relates to a price, which must cover all the expenses associated with the product or the product service for the techno-economic system providing it. But, the price must also be set so that it gives a competitive profit back to all partners of the techno-economic system. How would this balance work in a future scenario, when the supply of natural resources is not high enough to meet demand? It would give rise to a substantial increase in the price of natural resources and, consequently, also the price of future products and product services. Many customers would no longer have the ability to pay for such product services. To counteract such a development, the acceptable level for resource consumption and waste regeneration has to be lowered. Future product systems must also be made cyclic to an even larger extent than required with today's natural resource prices. Consequently, to arrive at a sustainable situation taking into account also those aspects would mean that the acceptable levels of resource consumption and waste regeneration, of human satisfaction, and of consumer price and profitability coincide.

From this brief review it is clear that a thorough study is needed to be able to compare different means to measure the degree of sustainability of a product life cycle. What fundamental limitations exist in the realization of sustainable product life cycles with regard to use of natural abiotic resources? How can the degree of sustainability of a product life cycle best be measured? What methods are available to take into account simultaneously the functional and environmental quality aspects, and also the economic dimensions of the concept of sustainability.

The total cost accounting approach, which has been employed for the purpose of product design in some recent studies (Carlsson *et al*., 2007; Carlsson, 2007; Carlsson, 2009), takes into account all those aspects. The total cost accounting approach would thus be a better indicator for sustainability according to the definition by Bakshi and Fiksel (2003) than the other indicators previously reviewed.

2. Total Cost As Indicator for Sustainability of a Product Life Cycle

The total cost accounting approach uses the end-user or consumer perspective and the ecological long-term view as a basis for compiling the contributions from all the various factors that might be important to the life cycle of a functional unit of a product. The point of departure is not a particular design alternative of the functional unit and its life cycle, but its intended function over time. When adopting the total cost, it is, however, not the absolute value of the total cost that is of main interest, but the difference in the total cost between two design alternatives of the functional unit of the product considered.

If one design alternative of the functional unit is chosen as reference, the model to be adopted can be described as follows:

For a fixed service time, τ_s, the difference in total cost (C_{RT}) associated with maintaining a specific function defined for the functional unit is estimated from

$$C_{RT} = C_{RP} + C_{RNIP} + C_{RO\&M} + C_{RF} + C_{REoL} + C_{RE} + C_{RD} \quad (1)$$

where C_{RP} = difference in production cost between the two design alternatives; C_{RNIP} = difference in cost associated with initial non-ideal function or performance between the two design alternatives; $C_{RO\&M}$ = difference in operation and maintenance cost between the two design alternatives; C_{RF} = difference in cost of probable failures and damage between the two design alternatives; C_{REoL} = difference in end-of-life cost between the two design alternatives; C_{RE} = difference in environmental cost associated with probable ecological damage between the two design alternatives, and C_{RD} = difference in development cost between the two design alternatives.

Specifying function is the starting point for setting up performance requirements of the functional unit, based on end-user and product considerations. From the performance requirements the two different design alternatives to be compared are defined in terms of a variety of properties and their cost characteristics are evaluated.

However, detailed guidance on how the different cost terms contributing to the total cost of a specific design alternative for a functional unit are assessed, falls outside the scope of the present paper, and the reader is referred to the previous work by Carlsson *et al.* (2007) and Carlsson (2007, 2009)

3. General Analysis of a Simple Case of Repeated Material Recycling in Design of Two Product Life Cycles

For the purpose of comparing and discussing different means to measure the degree of sustainability of a product life cycle, the suitability of two design alternatives of a metallic electric conductor was studied. The conductors were considered for use as winding in transformers; in one case the conductor was made of pure metallic copper, and in the other case of pure metallic aluminium. Comparison was also made under assumption that the resistance losses in the metallic conductor strips should be the same. Thus, to compensate for the higher electrical resistivity of metallic aluminium compared to metallic copper, the cross-sectional area of the aluminium strips was assumed to be 1.64 times larger compared to that of the metallic copper strips. This meant that 1 kg of metallic copper corresponded to 0.5 kg of aluminium, taking into account the different densities of the two conductor strips. The two kinds of conductor strips were assumed to be insulated in the same way, although the transformer with aluminium would be larger compared to the transformer with copper winding.

The product or product element life cycle studied is schematically shown in Figure 1. After the end-of-life phase of the product element manufactured with primary metal, it was assumed that this product element would be replaced by a second generation of product element. This was manufactured, to the extent possible, from recycled metal from the first product element cycle. The product element life cycle, therefore, was composed of the following steps:

Step 1: Manufacturing of pure metal A from ore containing A

Step 2: Manufacturing of the product element composed of pure metal A

Step 3: Product element composed of pure metal A during its service life

Step 4: Recycling of metal A after use in product element

Step 5: Addition of compensatory metal for producing second generation of product element

By designing the product element life cycle this way, the amount of metal that could be conserved for further use by recycling could be distinguished. It was then possible to analyse in more detail the amount of metal that was lost by non-ideal processes involved in the product element life cycle.

In terms of the total cost model previously described, steps 1 and 2 constitute the production phase and are to be related to production cost and cost of initial non-ideal performance. The service phase described by step 3 is to be related to operation and maintenance costs and costs for probable failures. Both step 4 and step 5 belong to the end-of-life phase. Cost of probable environmental damage can be split up into three parts related to the production phase, service phase, and end-of-life phase.

In the analysis, environmental data originated by Althaus and given in the Ecoinvent database of the SimaPro 7 LCA software tool (2009), were used. The data representing production of primary metal, production of secondary metal, and sheet rolling of copper and aluminium are as shown in Table E1 and Table E2 in the Enclosure. Information on metal prices, also shown in the tables, was obtained mainly from the London Metal Exchange (2009).

3.1 Framework for Economic Analysis of Copper Conductor and Aluminium Conductor Product Element Life Cycles

As the point of departure for a simple economic analysis, the metallic conductor strips were assumed to be manufactured by sheet rolling from rods of pure metal. All economic and environmental data (see Table E1 and Table E2 in the Enclosure) were related to the production of 1 kg metallic copper conductor strips. Thus, in the case of the corresponding metallic aluminium conductor strips, all data were related to 0.5 kg aluminium. In terms of year 2009 cost data, the cost of metallic copper strips installed in the transformer per unit mass of copper was thereafter set at \$13.80 (All amounts in US dollars). In this cost \$7.20 constituted the cost of metallic copper and the cost of energy estimated as described in Table E3 (see the Enclosure). The remaining \$6.60 was assumed to cover expenses for labour, interest, and other costs. The service the transformer fulfilled during its life cycle and which could be allocated to the metallic copper strips, was assumed to represent a value of \$20.70 per unit mass of copper.

This service value was also assumed to be the same for the aluminium winding strips in the other kind of transformer. But, the price of the metallic conductor strips was here set at \$10.90 per 0.5 kg aluminium or unit mass of copper. The sum of the cost of the metallic aluminium and the energy needed to produce the strips was in this case estimated to be \$1.05, as described in Table E4 (see the Enclosure).

Thus, the prices for the installed conductor strips differed by about 20% between the two kinds of transformers. In the case of the copper-based transformer winding, the cost of material for the conductor strips for the product element manufacturer was 52% of the production cost. In the case of the aluminium-based transformer, this cost was 9% of the production cost. However, the assumptions made about the economics of the intended application for the two kinds of metals were just made arbitrarily. The main purpose of the analysis was to illustrate more qualitatively than quantitatively the difference in the product characteristics between the two kinds of transformers.

In Table E3 in the Enclosure, the characteristics for the different processes constituting the metallic copper-based winding element life cycle are presented, and in Table E4 the corresponding data for the metallic aluminium-based winding life cycle are shown.

3.2 Comparison Based on Material and Energy Balances

When analysing the pattern of material flows contained in product element life cycles, the situation is quite complex, as is schematically shown in Figure 2. Metal may be produced and exchanged between different parties during the product element life cycle; metal may be conserved and even lost. Material losses of copper and aluminium to the surroundings arise from non-ideal performance of manufacturing processes for primary metal. Losses also occur due to lack of durability of metals during service and end-of-life phase of product element, and of non-ideal performance of recycling metals. All such processes thus contribute to reducing the extent of sustainability of product element life cycles.

It should be pointed out that in the two cases considered, it was assumed, that scrap metal from the manufacturing of the product element was taken care of in the production of second-generation metal. Concerning the environmental data used for sheet rolling, it is stated in the Ecoinvent database (2009) that all process steps that can be attributed to semi-fabrication (sawing, scalping, hot rolling, cold rolling, solution heat treatment, finishing, and packaging) are included. However, the data do not include the material that is rolled; only the amount of metal lost in waste is balanced as primary metal input. It is doubtful, therefore, that all metal lost in waste in the process could be considered as scrap available for production of secondary metal, as has been done in present analysis. However, this assumption would have a minor influence on the main conclusions that can be drawn from the present study.

For the copper conductor case, the overall metallic mass efficiency for the first cycle is 77%, and for the second cycle, 88%. For the aluminium conductor case, the corresponding numbers are 81% and 97%, respectively. For the copper conductor case, the metal mass losses causing lack of sustainability arise from the production of primary metal from raw material to an extent of 12%; for the aluminium conductor, this rate is 16%. However, in the present analysis, the difference in durability and related performance for recycling makes the aluminium conductor the better choice in terms of metallic mass efficiency, relative to the copper conductor. The amount of metal oxide in the scrap seems the most probable reason for this difference. This would be a consequence of the fact that the resistance to atmospheric corrosion with respect to metal mass loss is higher for aluminium compared to copper; see, for example, the international standard ISO 9223.

The cumulative energy demand characteristics for the product element life cycles are shown in Figure 4. For the copper conductor, total cumulative energy demand for the first cycle is 87 MJ, and for the second cycle, it is 50 MJ. For the aluminium conductor, corresponding demand is 138 MJ and 21 MJ, respectively. From an energy point of view, the energy need for production of primary aluminium makes aluminium a second choice to copper, if only the first product element life cycle is considered. If, on the other hand, only the second product life cycle is taken into account, the opposite is the case.

3.3 Comparison Based on Total Cost and Added Value Analysis

The total costs for the two metallic conductor design alternatives were calculated by applying equation (1) and the data presented in Table E3 and Table E4 in the Enclosure.

When determining the dimensions of the metallic conductor strips, the difference in electric conductivity between copper and aluminium was taken into account, so that heat losses during use of the metallic conductors would be the same. As a consequence, the cost term for initial non-ideal performance, C_{NIP}, was excluded in the analysis, because it has the same value for the two design alternatives. For the cost of operation and maintenance, $C_{O\&M}$, the same assumption was made.

When evaluating the cost of probable failures, C_F, it was first assumed that the life-limiting factor for the two alternatives was the durability of the insulation of the transformers. As a consequence, service life would be the same for both. However, using the Ecoinvent data, it was not possible to distinguish between degradation of the metal strips during use, degradation of the scrap metal after use, and non-ideal behaviour of the recycling process in terms of material loss. Therefore, only the sum of the contributions from those processes could be estimated. In the present analysis, this sum is shown as cost of probable failures, C_F. Cost of development, C_{RD}, was not considered of particular interest for this study, and was therefore excluded.

In estimating the cost of probable environmental damage, C_E, the main difficulty was selecting the most relevant indicator and translating and expressing this indicator in cost terms. For that purpose, the Ecoindicator 99 and the EPS indicator previously mentioned were considered, but, also the cumulative exergy indicator, CExD (Bösch *et .al.*, 2007), excluding contribution from renewable water.

For translating the Ecoindicator 99 value into cost, the following assumptions were made. In the Ecoindicator 99 method the damage to natural resources is measured in terms of surplus energy. This is defined as the difference in the energy needed to extract a resource now and at some point in the future. Based upon the suggestion made by Müller-Wenk (1998), the future surplus energy is calculated at $Q \cdot N$. The quantity Q represents the total amount that has been extracted by humans before 1990, and N represents the number of times this amount has been extracted. In the SimaPro 7 software, data for damage to resources are given in energy terms for the case $N = 5$ (Goedkoop & Spriensma, 1999), and this reference point was considered reasonable for the purpose of the present analysis. To convert this energy value into cost, the electric energy price valid for the year 2009 was used.

To convert the contribution from the other damage categories of Ecoindicator 99 into cost, an analogy was made to the previously described case of damage to natural resources. For a single score, the environmental impact value derived by the Ecoindicator 99 method is given in points, Pt. Thus, the ratio between the value in Pt and the corresponding surplus energy value was first calculated. This factor was then used to convert the Pt values for the other damage categories, human health and ecosystem quality into energy units. Then they were transformed into cost units, using the same procedure as described for the damage category resources.

The results illustrated in Figure 5 indicate that metallic copper is a better alternative to metallic aluminium in terms of total cost of the application considered. How different cost terms contribute to the total costs is illustrated in Figure 6a for the copper conductor transformer and in Figure 6b for the aluminium conductor transformer. The category waste shown in the figures may denote costs associated with ore material lost to the surroundings (see C_P), and with metal lost due to oxidation or non-ideal recycling (see C_F). The category waste also includes environmental costs associated with damage to human health and ecosystem quality (see C_E (EI99)).

It can be concluded that to become equally favourable in terms of total cost, the manufacturer costs for labour, interest, and other costs need to be reduced by about 13% in the case of aluminium winding. This means that the costs for interest and other costs will be \$8.50 for the aluminium winding transformer manufacturer compared with \$6.60 for the copper winding transformer manufacturer.

In practice, however, it is doubtful, that this situation would be acceptable to the manufacturer of the aluminium winding transformer. The aluminium strips need a larger volume compared with the copper strips, which would increase the cost of the whole transformer. In reality, thus, the situation is far more complex than it appears from the example chosen for the present study. (See e.g. De Keulenaer, 2002).

In the evaluation of the total cost, the functional unit was limited to the first product element life cycle. However, the most important factor favouring the copper design alternative over the aluminium design alternative is selling recycled metal to the next product element life cycle, or for some other future use. Also, when considering the costs of probable environmental damage the associated environmental load cost was transferred from the first product element life cycle to the forthcoming product element life cycle. This is in agreement with common practice in life cycle analysis (LCA).

To analyse the appropriateness of such an approach from a sustainability point of view, the concept of added value, therefore, was introduced. The added value was defined as the financial value minus the associated costs of energy and material, using year 2009 cost data. The situation with respect to added value for the various actors in the first and second product element life cycles is shown in Figure 7a and Figure 7b. The situation for the product manufacturer and the consumer is the same in the two life cycles, whereas it is quite different for the ore supplier and the metal supplier.

The results in Figures 5 to 7 clearly illustrate the importance of metal recycling, and it can be concluded that, without recycling, aluminium would be a better alternative than copper. With recycling, copper is a better alternative than aluminium for the metal supplier, but this might also be true for the consumer, as indicated in Figure 7. If the consumer could sell the product at its end-of-life and be paid an amount of money that corresponds to its value in scrap metal, the situation illustrated in Figure 7 would become true. Alternatively, the consumer could sell the transformer to a scrap dealer and share with the dealer the income from selling the scrap.

From a consumer perspective, the big difference in the price of scrap metal between copper and aluminium would be of utmost importance. It would even be more important than the difference in price between a transformer with copper winding and a transformer with aluminium winding.

From the point of view of a product manufacturer, aluminium winding may seem the better choice because of its lower price. However, extra costs for the product manufacturer, due to the fact that the aluminium winding requires a bigger transformer volume relative to that with the copper winding, may change this picture, as pointed out previously.

Let us now consider the conduct of a series of product element life cycles. The first product element life cycle uses virgin metal to produce the product element. The second product element life cycle uses mainly scrap metal as raw material and to a lesser extent virgin metal. In the same way as the second product element life cycle, consecutive product element life cycles can be conducted, at least hypothetically, an infinite number of times. As a consequence, the characteristics of the average product element life cycle will approach that of the second product element life cycle in terms of added value. This is true for all the various actors in the product element life cycle and also when considering the environmental cost. The total cost characteristics shown in Figure 5, consequently, will be the most relevant to use when assessing the maximum degree of sustainability, which would be possible to obtain with present day technology. However, it is important to point out that the first product element life cycle must be designed so that metal losses to surroundings are compensated. This would be done by addition of virgin metal to the extent required to keep net metal balance at zero in the second product element cycle.

To measure the extent of sustainability of the product element life cycles, consequently, the life cycle that makes use of secondary metal would be used as a model. It is the total cost, evaluated as shown in Figure 6a and Figure 6b, that would be considered as suitable indicator for sustainability and, thus, also applicable more generally to product design.

3.4 Total Cost Comparisons Based on Other Indicators of Environmental Impact

Environmental impact indicators other than Ecoindicator 99 may be used for total cost analysis, as shown in Figure 8 and Figure 9.

The EPS system is a monetary approach to assessing environmental impact, based upon the concept of willingness to pay to restore changes in some selected safeguard objects (Steen, 1996). The system was mainly developed to assist designers and product developers in finding which one of two product concepts has the least impact on the environment. The models and data in EPS are, intended to improve environmental performance of products. The environmental impact indicator in the EPS system uses the unit (ELU), environmental load unit. But, in LCA software tools in general, there is no guidance on how to translate the ELU value into cost. In total cost accounting, therefore, it was recently suggested by Carlsson (2007) that the environmental cost associated with probable ecological damage, C_E, be expressed as

$$C_E = W_E \cdot (\text{ELU - value}) \qquad (2)$$

where W_E is a weighting factor. This would be set by the individual company itself and dictated by the company's environmental policy. If W_E is set at equal to one and EURO is used as the unit for cost, one arrives at the original idea of the EPS system. However, the idea to set one ELU equal to one EURO may be misleading; therefore, the result of an impact assessment study using the EPS system is presently expressed in ELUs only. However, when attempting to compare the different cost terms for two design alternatives of a functional unit of a product, a quantitative cost expression for C_E is needed, and therefore, equation (2) was set up.

As is shown in Figure 8, if one ELU is set equal at to one US$ and the year 2009 energy price is used, the environmental cost C_E will be the dominating cost term, and the copper transformer design alternative will become less favourable compared to the aluminium alternative.

If the C_E value based on Ecoindicator 99 shown in Figure 5 is used as reference, W_E values may be calculated by use of equation (2). These become 0.03 for the copper design alternative and 0.06 for the aluminium design alternative. The quotient between the C_E values for the copper case and the aluminium case is equal to 43 if the EPS indicator is used. This value should be compared with the quotient value of 16 obtained when Ecoindicator 99 is used. Accordingly, the EPS system places much more emphasis on depletion of less abundant metals than on energy, as the Ecoindicator 99 does.

The cumulative exergy demand indicator CExD (Frischknecht *et al.* 2007; Bösch *et al.* 2007) was also considered for use, but, without taking into account the contribution from the category renewable water, which is normally included in this indicator. With the modified cumulative exergy indicator, which in the present study was denoted "CExD(material and energy)", the environmental cost C_E was calculated. It was found that the environmental cost values with this indicator are slightly higher than those obtained when the Ecoindicator 99 is used; cf. Figure 5 and Figure 9. The quotient between the C_E values for the copper-based transformer and the aluminium-based transformer is equal to 2.9 in the CExD(material and energy) case, compared to 16 when Ecoindicator 99 is used. The modified cumulative exergy demand indicator thus places less emphasis on depletion of less abundant metals than even Ecoindicator 99 does. The copper design alternative is slightly more favourable than the aluminium alternative also with this indicator in terms of total cost.

It should, however, be pointed out that Ecoindicator 99 contains contributions not only related to depletion of natural resources but also to damage to human health and ecosystem quality that the cumulative exergy demand indicator does not.

Consequently, there are different indicators that may be useful to express environmental impact in total cost accounting.

4. Most Suitable Indicator for Sustainability – Discussion

4.1 Minimal Depletion of Natural Resources as a Measure of Sustainability

If sustainability is restricted to minimal depletion of natural resources only, all the indicators shown in Table 1 point to the use of aluminium in preference to copper. The most important factor favouring the aluminium alternative to the copper alternative is the need to compensate for the mass loss of metal in the product element life cycle. This mass loss is 3% in the former case and 11% in the latter case.

As the concentration of copper in the earth's crust is much lower than the corresponding concentration of aluminium, the EPS indicator gives in a relative sense the highest-ranking number for aluminium relative to copper; see Table 1. The contribution from depletion of natural resources according to the Ecoindicator 99 method gives a ranking number for aluminium relative to copper, which is only 12% of that obtained when the EPS indicator is used.

The EPS approach, using the average concentration of metal in the earth's, crust has been criticized by many. For example, Müller-Wenk (1998) writes that, because there is no doubt that the necessity to switch to average earth-crust materials in actual mining is extremely far in the future, it is presumably not a concern of our society. If abiotic resources are considered to be scarce, the relevant question for a weighting model should therefore focus on the resource concentrations available in 100 or 1000 years from now. Resources with the average crustal concentration will 'never' be used for actual mining. Instead, Müller-Wenk (1998) looks at the increased energy requirement for future generations. His category indicators relate to the increased use of energy when the total extracted metals are fivefold compared to the base year 1990. In practice, this may be expected to happen in a few hundred years, according to Steen (2006). If the method suggested by Müller-Wenk is used, the ore grade only drops with factors between 1.1 and 5, which gives an entirely different picture compared with the EPS approach. Damage costs for resources would then be more than two orders of magnitude less (Steen, 2006).

Consequently, there are different ideas about which time perspective to apply. The European Commission developed a strategy with a 25-year perspective. With this time perspective in mind, the LCIA working group recommends that the work of Müller-Wenk (1998) be deemed as current best practice for metal resources (Brent and Hietkamp, 2006).

The indicator cumulative exergy demand (CExD) was introduced by Bösch *et al.* (2007) to provide a product, summing up the exergy of all resources required. CExD includes the exergy of energy carriers as well as of non-energetic materials. It was used in the present study excluding the contribution from the category renewable water, to express depletion of natural resources; see Figure 9 and Table E1 in the Enclosure.

Bösch *et al.* (2007) mean that CExD would be a valuable indicator to assess energy and resource demand from the perspective of energetic quality. It is a more comprehensive indicator than cumulative energy demand (CED), but is simpler in the setting up, as compared to the damage category resource of Ecoindicator 99.

Bösch *et al.* (2007) claim that in other depletion indicators used, not all resources are considered exhaustible or scarce by the indicators, and information on global resource scarcity of specific resources is difficult to obtain. An advantage of CExD in contrast to Eco-indicator 99 and EPS would be that fewer assumptions are needed, because CExD considers just an inherent property of the resource, that is, exergy. The main source of uncertainty is the (sometimes unknown) composition of mineral resources, such as rocks and ores, Bösch *et al.* (2007) point out.

As can be concluded from the data shown in Table 1, the modified CExD indicator used in the present study gives less preference to the aluminium alternative versus the copper alternative than the other indicators do. If only the energy part of the exergy is considered, the Cu/Al ratio is 2.7, with the modified CExD indicator. This value should be compared with 2.9 obtained when the category 'Ecoindicator 99SE energy' is used (see Table E3 and Table E4 in the Enclosure). If then only depletion of material resources is considered, the same ratio is 10 when the modified CExD indicator is used. With 'Ecoindicator 99SE material', it is 48. Consequently, reduction in material quality versus reduction in energy quality is given higher weight with Ecoindicator 99 than with CExD, as has been pointed out previously.

It is worth noticing that when changes in stored exergy are considered (cf data in Table E3 and Table E4 in the Enclosure), the aluminium alternative turns out to be less favourable than the copper alternative. The Cu/Al ratio becomes 0.50, due to the fact that the exergy content of pure metallic aluminium is so much higher that of pure metallic copper.

To conclude, stored exergy would be a relevant indicator if the metal were used as a fuel. However, the cumulative exergy demand indicator would be more relevant in product design.

To be able to judge which is the best indicator to assess sustainability when considering depletion of natural resources, the economic perspective had to be taken into consideration.

4.2 Minimal Cost of Probable Environmental Damage C_E as Indicator for Sustainability

Translating the indicator values for depletion of natural resources into cost is a rather straightforward task when the modified CExD indicator is used, because it is given in exergy units. Using the price of electricity, which stands for pure exergy, it can readily be converted into cost.

The environmental impact from depletion of natural resources according to Ecoindicator 99 was in the present study translated into cost starting from the Müller-Wenk (1998)-based surplus energy values given in the Ecoinvent database (2007). As described in section 3.3, the corresponding values for impacts related to damage to human health and ecosystem quality could also be translated into energy units. Values for the total Ecoindicator 99 obtained this way were then translated from energy units into cost by use of the price of electricity. This may not be entirely correct in the Ecoindicator 99 case, as low value heat may also play a role that maybe should be compensated for in a better way than practiced in the present study.

The problem of translating the EPS indicator's values into cost is more difficult, as discussed in section 3.4.

A general advantage of expressing environmental impact in cost terms is that it can be compared with other kinds of costs associated with conduct of a product life cycle. The advantage of using Ecoindicator 99 and CExD (materials+energy) for that is that the meter for environmental impact will follow the price of electricity, and consequently, also the steadily increasing price of crude oil and other natural resources.

However, there may sometimes be problems in understanding the interrelationship between cost/availability and price of an abiotic natural resource; see, for example, Table E3 and Table E4 in the Enclosure. Some metals, for example, aluminium, may not be as sensitive to changes in price of energy and electricity as expected. For example, the price of electricity in the United States between 1999 and 2009 increased from \$.03 to \$.10 per kWh, which is more than three times. The price of metallic copper increased from \$1.80 to \$6.90 per kg during the same time period, which is nearly four times. But, the price of aluminium increased from \$1.60 to \$2.00 per kg, which is only 25%. In 2008 the price of aluminium had a peak corresponding to \$3.10 per kg. If this peak value were used to estimate the average increase in the price per year, the result would be 8%. This corresponds to an average increase by a factor of around two within a time frame of ten years. Considering the general increase in the price of electricity, the price of metallic aluminium would also have been at least double in 2009.

Regarding the time perspective, there are differences between the three indicators for environmental impact. The EPS indicator has the longest time frame, so far away from now so that it is presumably not a concern of our society (Müller-Wenk, 1998). With the Ecoindicator 99 approach based on the Müller-Wenk model, the time perspective is around 100 years or even less (see above), and is therefore recommended for impact assessment in life cycle analysis by Brent and Hietkamp (2006).

The modified CExD indicator places less emphasis on future problems associated with scarcity of metal in ores than the Ecoindicator 99 does. On the other hand, it places more emphasis on energy and energy quality. However, the Ecoindicator 99 has the advantage over the cumulative exergy demand indicator of also taking into account environmental impact related to human health and ecosystem quality.

Besides the environmental quality aspect, the service or function the product represents must be considered from a sustainability point of view. The sustainability concept requires that not only the environmental impact cost is taken into account but also other costs of the kind considered in the total cost accounting approach.

4.3 Total Cost as Indicator for Sustainability

Total cost accounting combines the factors of functional quality, cost effectiveness, reliability, and long-term performance of a particular design alternative of a functional unit with the environmental performance aspects related to ecological soundness and recoverability.

In the present study this approach was used to illustrate how different costs contribute to the total cost. When comparing transformer alternatives made with copper winding and with aluminium winding, the pattern of costs differs significantly depending on which indicator for environmental impact that was used, as shown in Figures 5, 8 and 9. It is, however, clear that the aluminium alternative is more favourable than the copper alternative for all kinds of costs, except the end-of-life cost. Because of its high negative value for the end-of-life cost, the copper alternative becomes the better choice, when Ecoindicator 99 is used to represent the environmental cost, as is illustrated in Figure 5. The main reason for the copper alternative becoming the more favourable can be related to the much higher residual value for copper compared to that for aluminium. Thus, it can be concluded that without recycling, copper cannot compete with aluminium in terms of total cost.

However, there is a time span between initial and final stage of product element life cycle that may influence cost characteristics. Therefore, present value costs related to the initial phase of the life cycle were calculated by employing methods commonly used in life cycle cost analysis (see e.g. Fuller and Petersen, 1996). In the calculations a life cycle of ten years was assumed. The interest rate was set at 8%. The increase in the price of energy was set at 10% per year in constant dollar value. The increase in the price of metallic copper was set to 11% per year in constant dollar value, and that of metallic aluminium to 8% per year in constant dollar value. As is shown in Figure 10, the difference in total cost between the two design alternatives is increased. The copper alternative is more favourable to the aluminium alternative to an even larger extent than shown in Figure 5. Investing in copper would be more profitable compared to investing in aluminium in terms of residual value.

The general conclusion of the present study is, thus, that copper is the better alternative in terms of total cost. In terms of added value, this is also true for the metal supplier and also be so for the consumer. The consumer might sell the product at its end-of-life and be paid for it in an amount of money that maximally corresponds to its value in scrap metal (see Figure 7). From a product manufacturer's point of view, an aluminium winding seems at first glance the better choice because of its lower material price. However, the aluminium winding requires a bigger transformer volume relative to a transformer with a copper winding. This may change the relative preference between copper and aluminium for the product manufacturer.

The total cost accounting approach adopted in the present study identifies the essential elements in the pattern of cost and therefore would constitute a valuable tool in sustainable product design. By the method used, difference in product life cycle cost, product cost effectiveness and profitability of two product design alternatives may be assessed and the more suitable alternative in economic terms be identified. From the results, you also can extract information on differences in resource consumption and waste regeneration and on matters related to human satisfaction, that is, differences in functional quality and reliability of the two alternatives.

In the present study the pros and cons of using either copper winding or aluminium winding in transformers designed for equal functional capability were analysed. The results showed that from an economic point of view, that is, from a total cost point of view excluding environmental impact cost, copper winding would be the better alternative. But, does that mean the copper alternative would also be the best choice in terms of sustainability? To arrive at a sustainable situation would mean, as previously pointed out, that the acceptable levels of resource consumption and waste regeneration, of human satisfaction, and of consumer price and profitability coincide. A sustainable product shall provide enduring economic value to the business enterprise pursuing the product. This means that the product or the product service it represents can only exist on the market if there is a willingness or ability by customers to pay for it. Therefore, the price of the product should cover all the expenses associated with the product or the product service for the techno-economic system providing it. The price of the product should also be set at a level that gives a competitive profit back to all partners of this techno-economic system. To change the preference from copper to aluminium in the example considered would require the manufacturer of the transformer with the aluminium winding to lower the price of his product. If he is unable to do so, his transformer would not be competitive with the transformer using copper winding, and as a consequence, his product would sooner or later disappear from the market. In this case, the transformer with the aluminium winding would never become a truly sustainable product, because it would not sustain the competition of the transformer with the copper winding on the market.

Following this logic, the product alternative with the lowest life cycle cost would be the most favourable and sustainable product alternative, at least in a short-term perspective. To take into account the long-term perspective and the aspect of environmental sustainability, in the total cost accounting approach adopted, an environmental impact cost term is introduced. By use of Ecoindicator 99 translated into cost this introduces a longer time perspective into the analysis, corresponding to at least one hundreds of years. However, the weight this environmental impact cost term would be given relative other cost terms in the total cost assessment is, however, open for a potential consumer of the product service to decide.

According to Klöpffer (2003), life cycle thinking is the prerequisite of any sound sustainability assessment of environmental, economic, or social performance. However, Klöpffer (2003) also concludes that life cycle thinking is not enough, since to estimate the magnitude of the trade-offs, the tools required have to be as quantitative as possible. The methodological problem is, according to Klöpffer (2003), to combine methods like environmental life cycle analysis (LCA), life cycle cost analysis (LCCA) and social-environmental life cycle assessment, as, for example, SELCA (O'Brian *et al.* (1996). Norris (2001) discusses integrating life cycle cost analysis and LCA and claims that the traditional separation of life cycle environmental assessment from economic analysis limits the influence and relevance of LCA for decision making. This leaves uncharacterized the important relationships and trade-offs between the economic and life cycle environmental performance of alternative product design decision scenarios. The LCA perspective and its results can have important economic relevance for companies, which may be missed when cost analyses neglect LCA's scope and findings. Norris also suggests total cost accounting, as presented by Beaver *et al.* (2000), a suitable mechanism for integrating LCCA and LCA results within a consistent framework to support holistic decision making. Users import LCA results from their existing LCA software, and they import traditional economic analysis results from their existing financial accounting systems. We believe the total cost accounting approach used in the present study can function in that respect.

4.4 Limitations in Realization of Sustainable Product Life Cycles with Respect to Utilization of Natural Abiotic Resources

There exist a variety of non-ideal processes, which may limit sustainability of a product life cycle and give rise to depletion of abiotic natural resources and energy quality.

To illustrate the importance of metal loss in the product element life cycles of the present study, it was assumed that the metal loss to the surroundings could be reduced to zero. With data presented in Table E3 and Table E4 as a point of departure, sustainability data for this ideal case were therefore calculated. The results are shown in Table 2, together with more realistic sustainability data taken from Table 1. The data for the copper alternative representing 100% recycling capability of metal differ significantly from the data when metal loss is 11%. When Ecoindicator 99 is used, the decrease is 66% when considering only resources, 53% in terms of environmental cost, and 18% when considering total cost. The corresponding decrease for aluminium, where metal loss is equal to 3%, is 21%, 15%, and less than 1%, respectively.

As the cumulative exergy demand indicator (materials + energy) places relatively more emphasis on contribution from energy flows than on material flows, the corresponding effects are less pronounced when this indicator is used.

To further demonstrate the importance of recycling, characteristics of product element cycles with no recycling of metal are illustrated in Table 2. The comparison reflects the utmost importance of recycling, especially for the copper design alternative. In terms of environmental cost, the reduction between a case with 0% recycling and 100% recycling is 80% and in terms of total cost it is 65%. For the aluminium alternative the corresponding numbers are 85% and 12%.

5. Conclusions and Recommendations

This study has shown that the total cost accounting approach is a valuable tool for assessing degree of sustainability of a product life cycle when combined with the Ecoindicator 99 indicator converted into cost terms. In particular, this approach was found useful in identifying which limitations exist for use of natural abiotic resources such as metals. It could further be concluded that the total cost accounting approach adopted is most suited to comparing two design alternatives of a product functional unit to decide which of the two is the more favourable. Using the total cost method, one can simultaneously take into account not only the functional and environmental quality aspects of a product life cycle but also its economical merits.

By analysing the suitability of copper and aluminium as winding material for transformers with respect to sustainability, and using available data from the Ecoinvent database for this analysis, the following conclusions can be drawn:

Repeated recycling of metal is a necessary requisite to obtain product element life cycles with a high degree of sustainability. With the data from the Ecoinvent database, it seems possible to obtain metal mass efficiencies of 88% for the product element life cycles in the case of copper, and 97% in the case of aluminium. In terms of cumulative energy demand, the second generation life cycle with copper winding requires 50 MJ per unit mass of copper, while the corresponding value for the aluminium case is 21 MJ.

When adopting the total cost accounting approach in assessing suitability of the two kinds of winding material, the outcome depends on costs other than those of materials and energy. To such belong costs for labour and interest for the metal supplier and the product manufacturer. With reasonable assumptions about those, it could be concluded that in terms of production cost the aluminium alternative would be the more favourable. This is also true when considering costs associated with the service phase. However, in terms of end-of-life cost the situation would be the opposite, and contributing to this fact is, above all, the much higher price of recycled copper compared to that of recycled aluminium.

When considering environmental cost, useful indicators are those that can be expressed in cost terms. Of those taken into account in our study, Ecoindicator 99 is recommended as most suitable compared with the EPS indicator and the cumulative exergy demand indicator. Ecoindicator 99 is preferable to the EPS indicator when considering the time perspectives of each. It is also preferable to the cumulative exergy demand indicator, considering the Ecoindicator 99's ability to take into account not only environmental damages related to natural resources, but also to human health and to ecosystem quality. With the Ecoindicator 99 indicator as the basis for estimating environmental cost, the aluminium alternative is better than the copper alternative. However, the contribution of environmental cost to the total cost is relatively small when compared with the production cost and the end-of-life cost. Therefore, the copper alternative is the better choice in terms of least total cost.

Thus, mainly because of the much higher price of recycled copper compared with the price of recycled aluminium, the copper alternative would be preferable to the aluminium alternative in terms of least total cost. This situation is even more pronounced when costs are converted to present value costs and the expected increase in the prices of energy, copper, and aluminium during life cycle is taken into account.

The utmost importance of recycling can be understood when comparing cases with 0% recycling and 100% recycling. In terms of environmental cost the reduction for the copper alternative is 80%, and in terms of total cost it is 65%. For the aluminium alternative the corresponding numbers are 85% and 12%.

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Table 1. Comparison of the metallic copper conductor transformer and the aluminium conductor transformer using different indicators of sustainability and illustrated by the examples shown in Table E3 and Table E4 in the Enclosure.

Abbreviations: CExD = Cumulative exergy demand; EI99 = Ecoindicator 99; EPS 2000 = Environmental Priority System, ELU= Environmental load units.

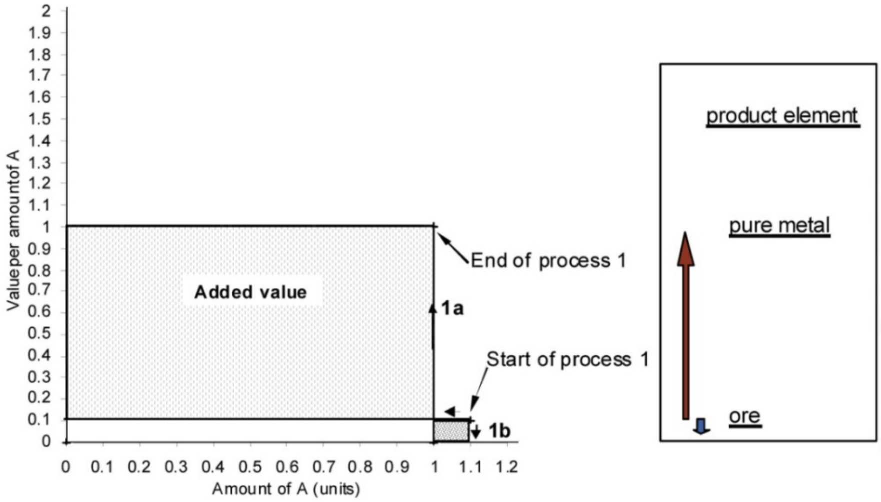
Indicator	**EI99**			**EPS 2000 indicator**			**CExD (materials + energy)**		
	Cu	Al	Cu/Al	Cu	Al	Cu/Al	Cu	Al	Cu/Al
Resources[1]	7.2	0.90	8.0	33	0.60	55	60	21	2.9
Environmental cost C_E ($)[2]	0.97	0.085	11	34	0.8	43	1.7	0.59	2.9
Total cost C_T ($)	9.6	10	0.96	43	11	3.9	10.3	10.6	0.97

[1] EI99: Resources in surplus energy in MJ eq; EPS 2000: Depletion of reserves in ELU, CExD (materials + energy) = Cumulative exergy demand regarding energy and materials in MJ eq; [2] When translating the EPS values into cost, 1 ELU has roughly been set equal to 1$; see equation (2).

Table 2. Comparison between the copper conductor transformer and the aluminium conductor transformer using different indicators for sustainability. Data presented in Table 2 within brackets refer to the cases of zero material loss and are shown together with corresponding data from Table 1, which are given within parentheses in the table. Abbreviations: CExD = Cumulative exergy demand; EI99 = Ecoindicator 99.

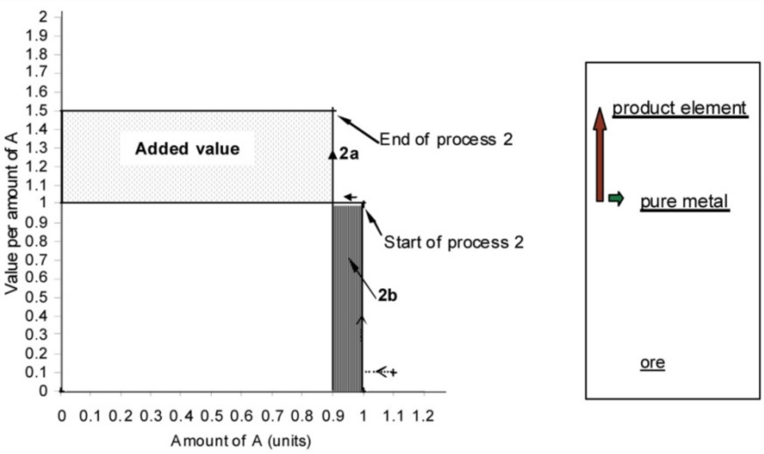
Indicator	**EI99**			**CExD (materials + energy)**		
	Cu	Al	Cu/Al	Cu	Al	Cu/Al
Resources[1]	2.4 (7.2) [48][3]	0.71 (0.90) [7.1][3]	3.4 (8.0) [6.8][3]	55 (60)	18 (21)	3.1 (2.9)
Environmental cost C_E ($)[2]	0.45 (0.97) [2.3][3]	0.072 (0.085) [0.48][3]	6.3 (11) [4.8][3]	1.5 (1.7)	0.50 (0.59)	3.0 (2.9)
Total cost C_T ($)	5.8 (7.1)[2] [16.4][3]	10 (10)[2] [11.4][3]	0.58 (0.71) [1.4][3]			

[1] EI99: Resources in surplus energy in MJ eq; CExD (materials+energy)=Cumulative exergy demand (energy+materials) in MJ eq; [2] Data from Figure 10; [3] Data representing product life cycles with no recycling of metal



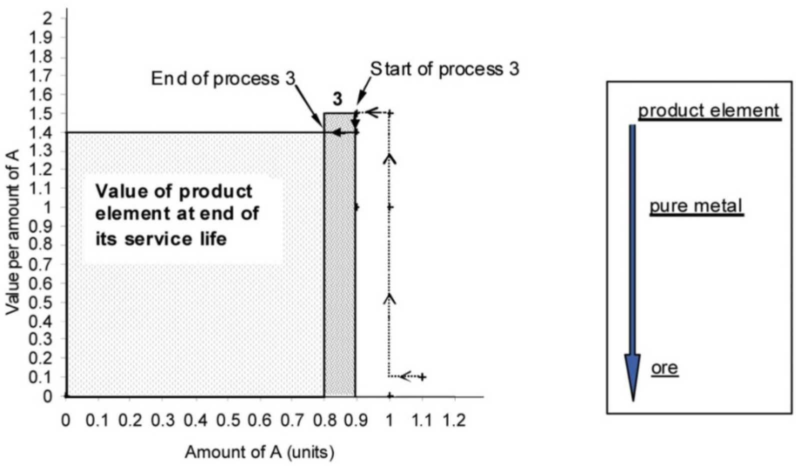
1a Net increase in value per unit mass of A during manufacturing of virgin metal A form ore containing A
1b Loss of value per unit mass of A due to substance loss of A to the surroundings during manufacturing of virgin metal A

1 Manufacturing of virgin metal A from ore containing A



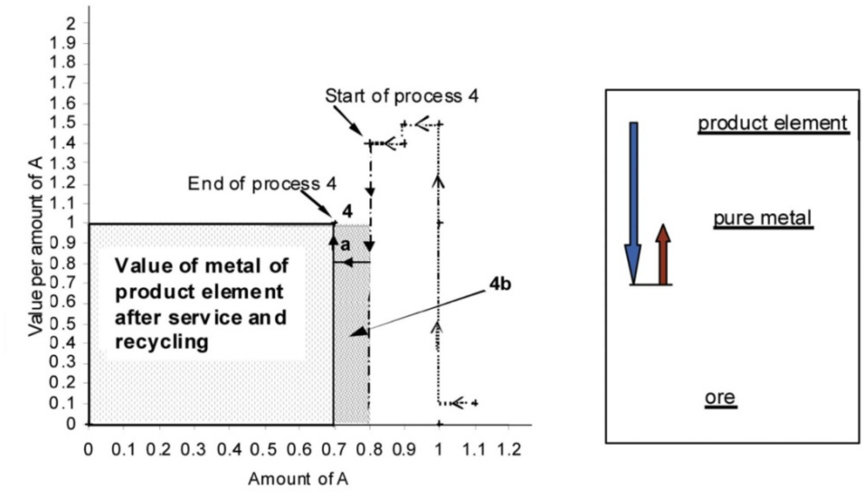
2a Net increase in value per unit mass during manufacturing of product element composed of metal A
2b Value stored in scrap of metal A from manufacturing of product element

2 Manufacturing of product element composed of metal A



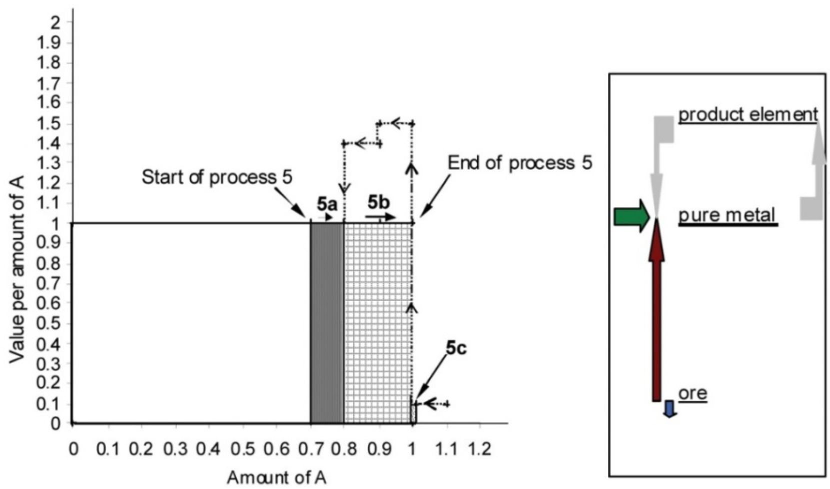
3 Loss in value per unit mass during service partly due to corrosion and associated run off and substance loss of A

3 Service of product element



4a Net increase in value per unit mass of metal A due to recycling
4b Loss of value due to substance loss of A to the surroundings in the recycling process

4 Recycling of metal A after use in product element



5a Supply of stored scrap from process **2**
5b Supply of virgin metal to compensate for substance loss of A in processes **3** and **4**
5c Loss of value per unit mass due to substance loss of A to the surroundings when manufacturing the new extra virgin metal A needed for second generation of product element

5 Addition of metal A for producing second generation of product element

Figure 1. Principle sketch of the product element life cycle studied, comprising the following steps:
Step 1: Manufacturing of pure metal A from ore containing A;
Step 2: Manufacturing of product element; Step 3: Product element during its service life;
Step 4: Recycling of metal A after use in product element;
Step 5: Addition of compensatory metal A for production of second generation of product element

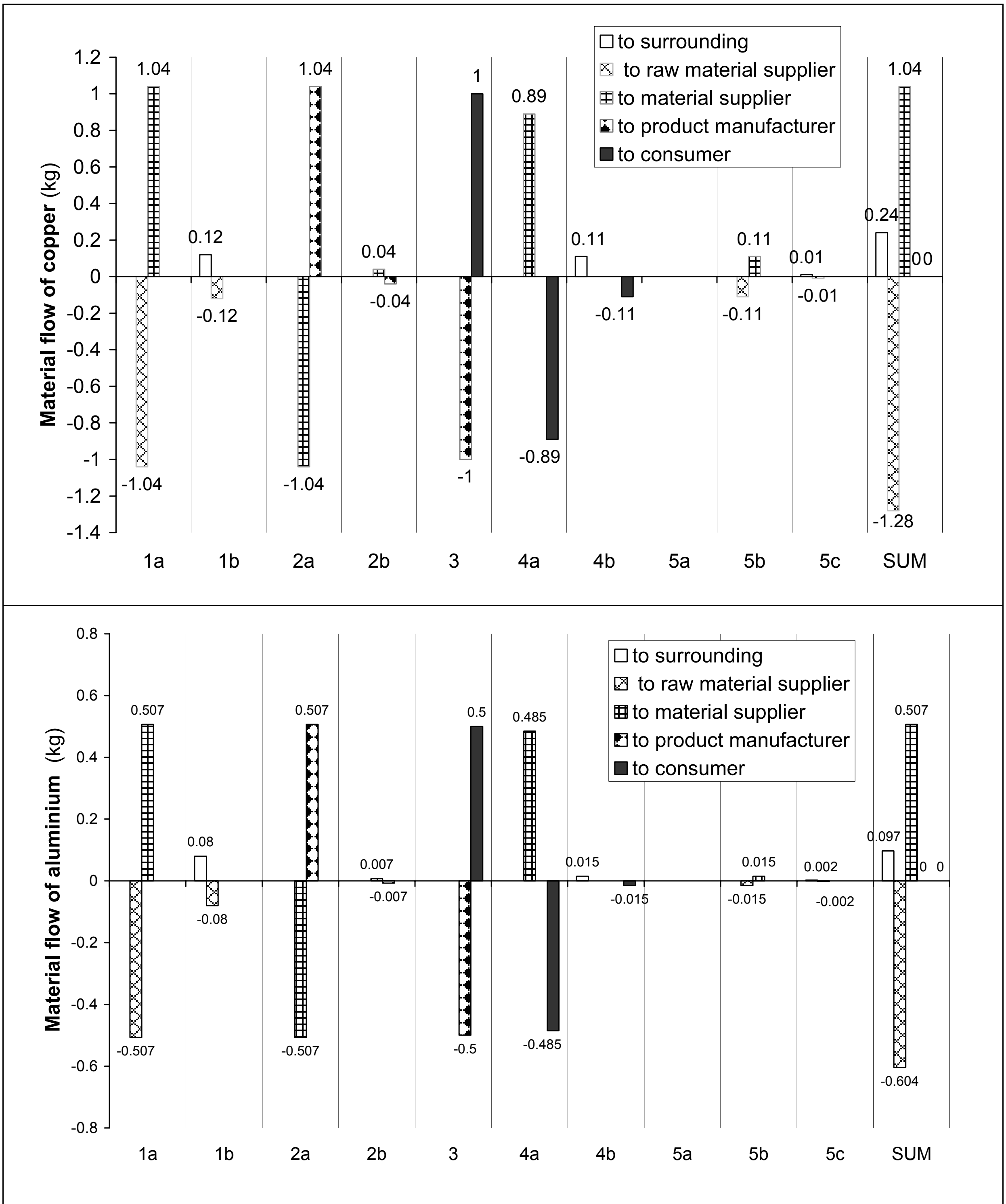


Figure 2. Material flow to the different actors in the product life cycle shown in Figure 1. For notations, see Tables E3 and E4 in the Enclosure. Upper diagram: Case of metallic copper conductor; Lower diagram: Case of metallic aluminium conductor.

Step 1: Manufacturing of pure material A from ore containing A: (a) manufacturing, (b) material loss of A during manufacturing. Step 2: Manufacturing of product element: (a) manufacturing, (b) scrap production during manufacturing. Step 3: Product element during its service life. Step 4: Recycling of material A after use in product element: (a) recycling, (b) material loss of A during service and recycling. Step 5: Addition of metal for production of second generation of product element: (a) supply of stored scrap from production, (b) supply of virgin metal to compensate for material loss of A during service and recycling, (c) loss of material of A associated with process 5b.

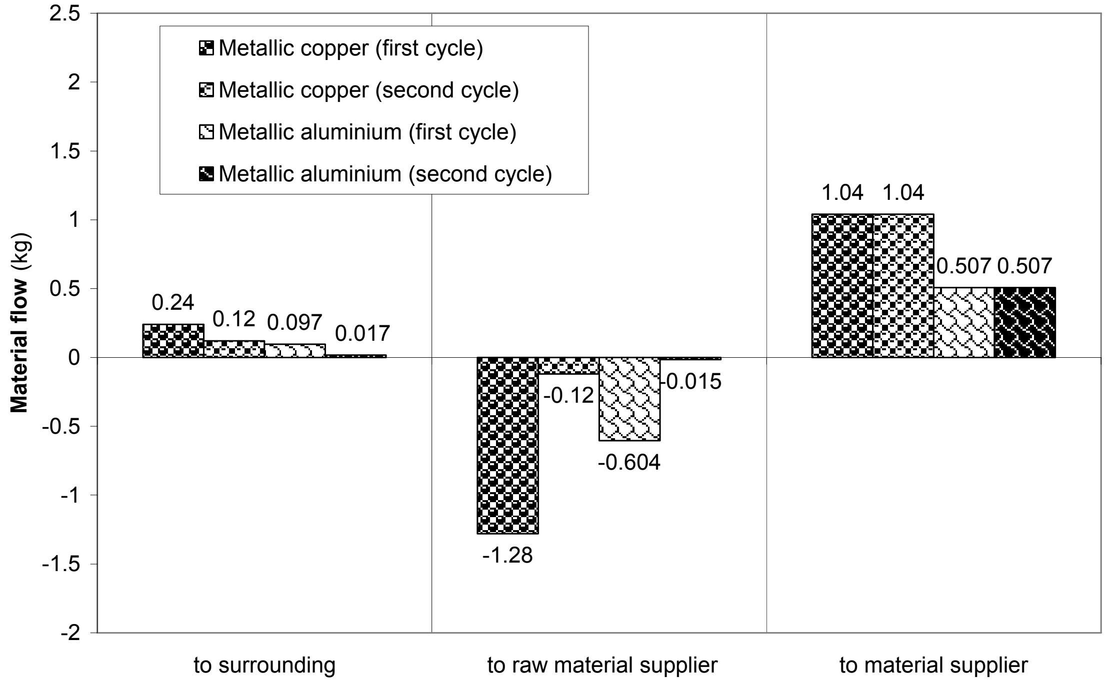


Figure 3. Net material flows related to the product element life cycles with a metallic copper conductor and with a metallic aluminium conductor. Diagrams are based on data from Figure 2 and Tables E3 and E4 in the Enclosure.

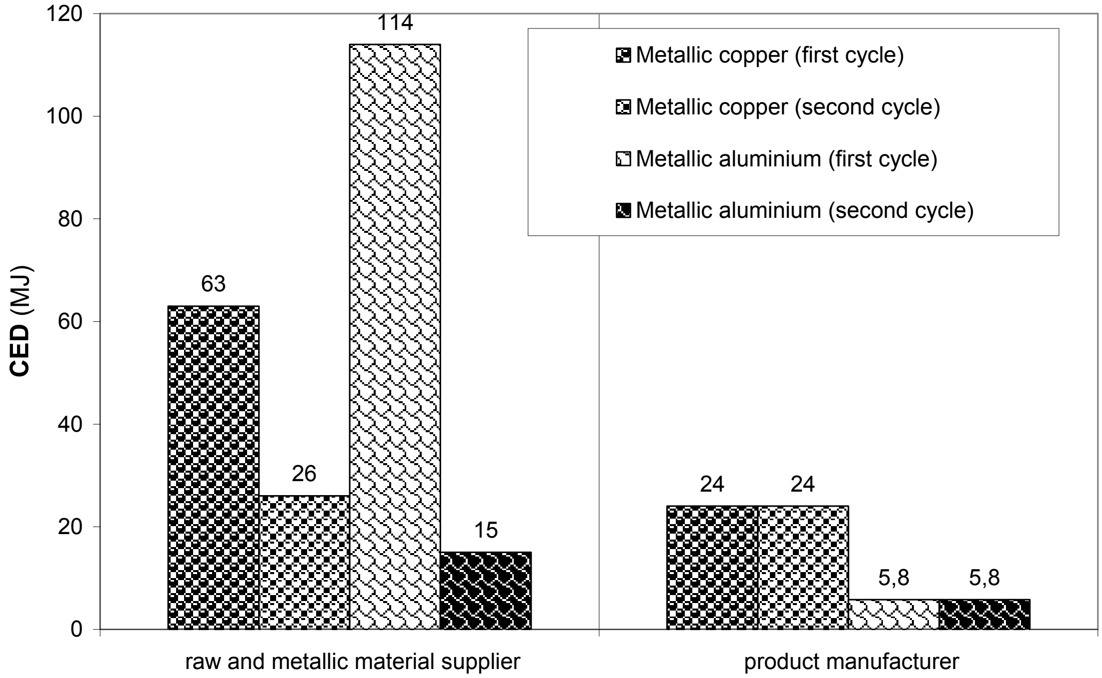


Figure 4. Net cumulative energy demand to the product element life cycles with a metallic copper conductor and with a metallic aluminium conductor. Diagrams are based on data from Tables E3 and E4 in the Enclosure. CED = cumulative energy demand.

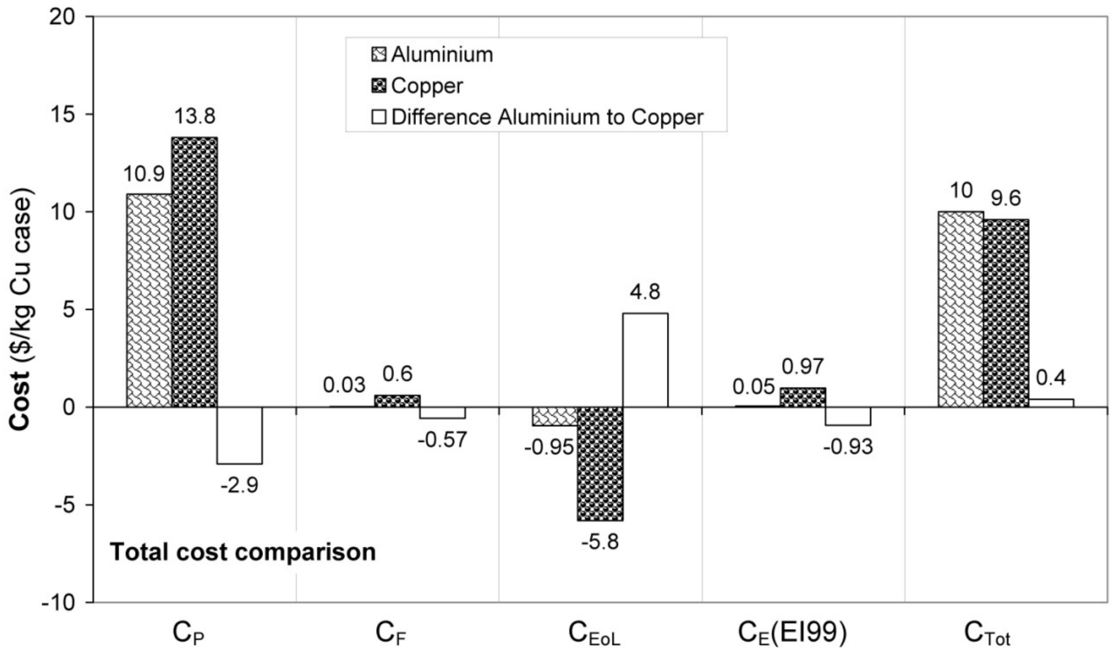


Figure 5. Comparison of the total cost of the metallic copper conductor transformer and the aluminium conductor transformer, using the modified Ecoindicator 99 cost indicator and cost data for year 2009; see Table E3 and Table E4 in the Enclosure. For other notations related to the terms of the total cost expression, see equation (1).

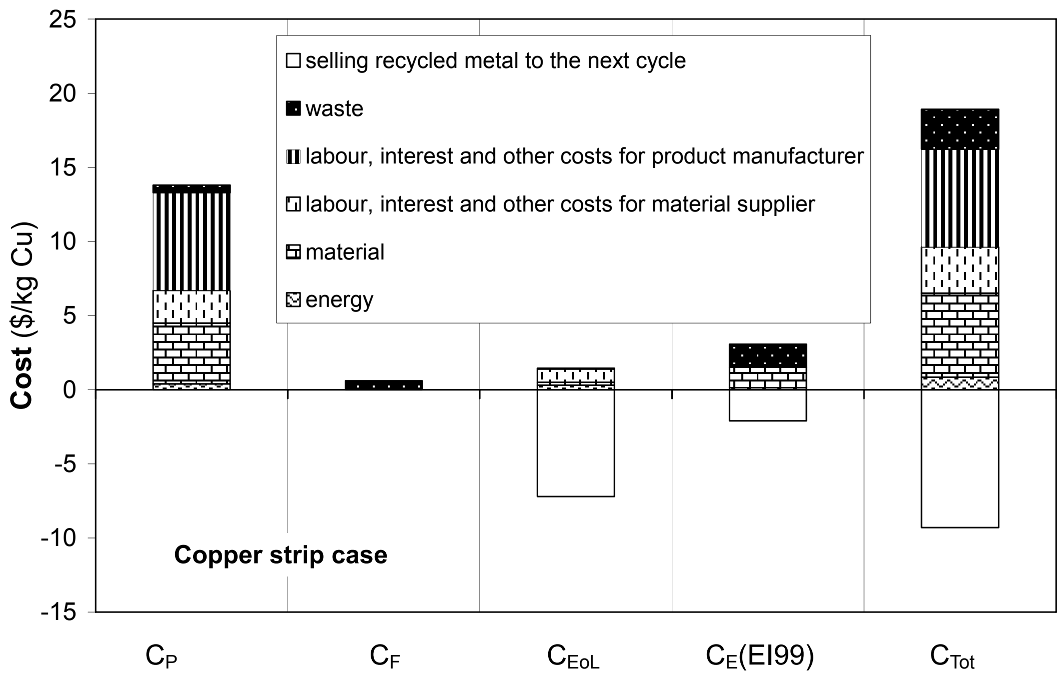


Figure 6a. Contributions to the total cost of the metallic copper conductor using the modified Ecoindicator 99 cost indicator and cost data for year 2009; see Table E3 in the Enclosure. For other notations related to the terms of the total cost expression, see equation (1)

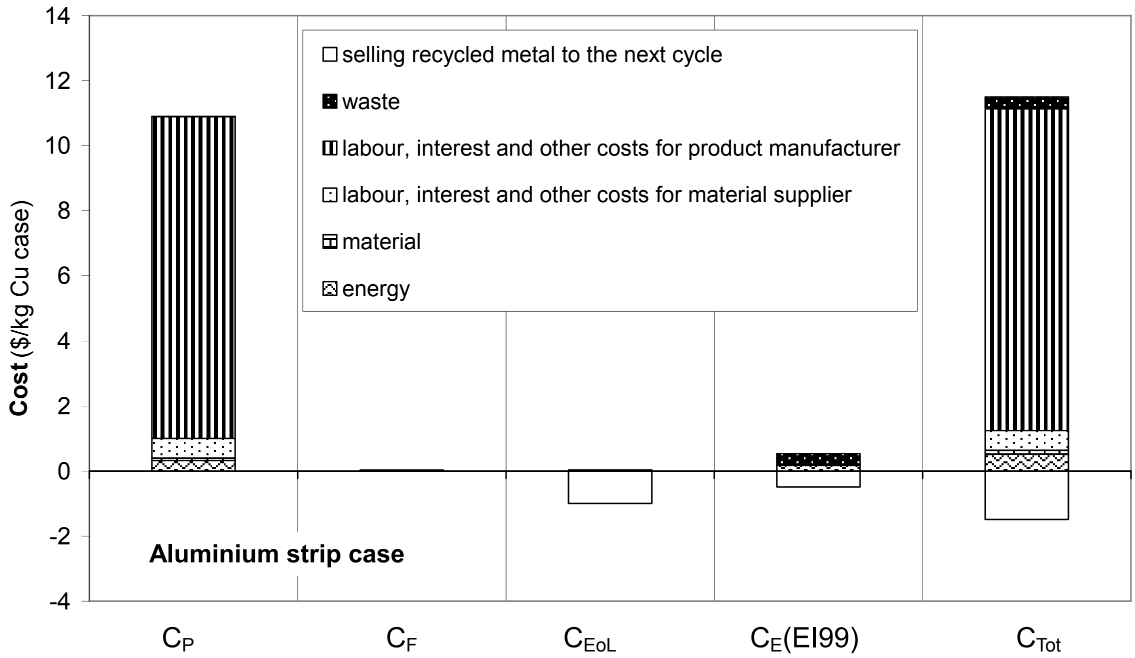
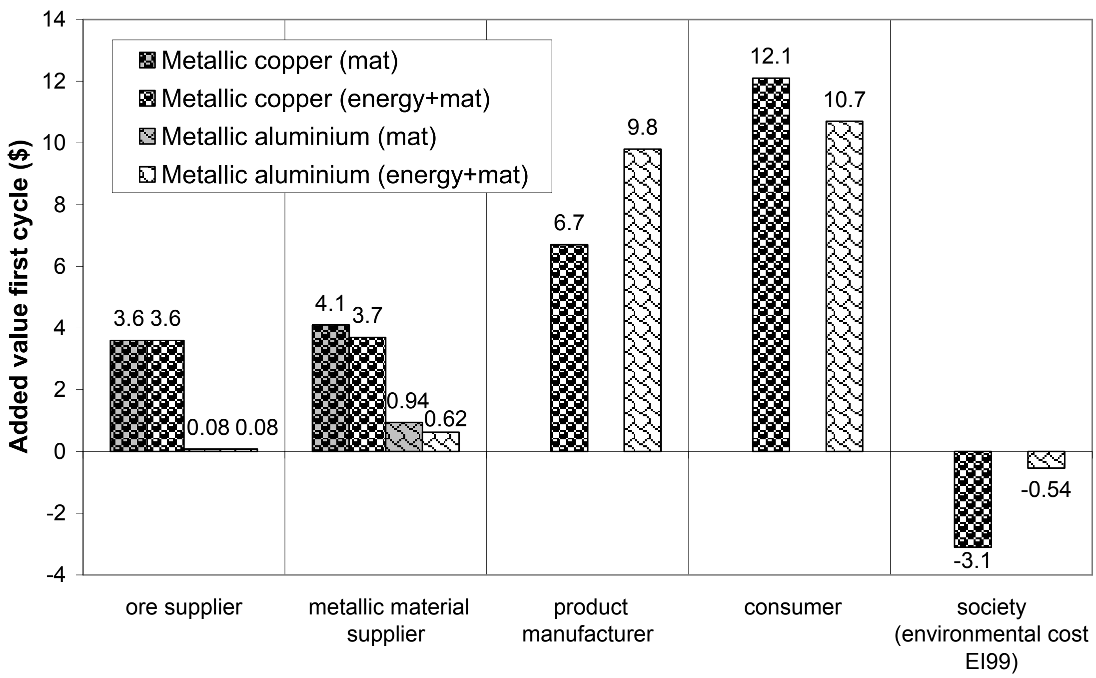


Figure 6b. Contributions to the total cost of the metallic aluminium conductor using the modified Ecoindicator 99 cost indicator and cost data for year 2009; see Table E3 in the Enclosure. For other notations related to the terms of the total cost expression, see equation (1).



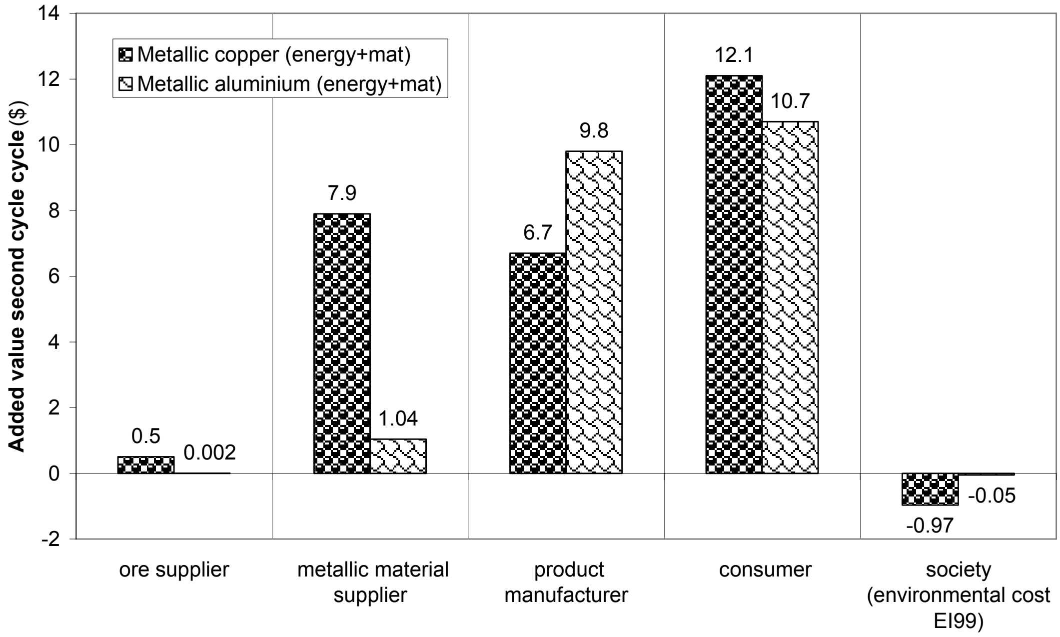


Figure 7. Added value analysis of the two product element life cycles with metallic copper conductor and with metallic aluminium conductor. Diagrams are based on data from 2009 in Table E3 and Table E4 in the Enclosure. Added value for a process has been taken as the difference between the financial value and the sum of material and energy costs as estimated for the different processes shown in the tables. For the added value analysis, it has been hypothesized that the metallic copper conductor transformer element has a price of $13.80 per unit mass of copper. The price for the metallic aluminium winding of equal functional performance was set at $10.90. The value of the service the transformers represent was equal for both cases and hypothesized to be $20.70.

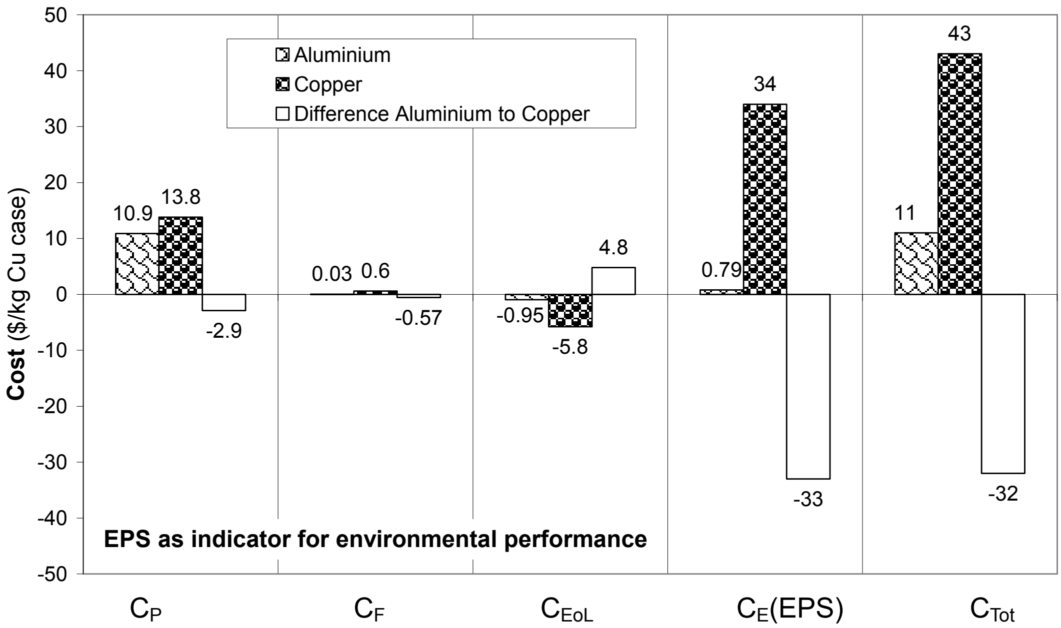


Figure 8. Comparison in total cost between the copper conductor transformer and the aluminium conductor transformer using the EPS indicator and cost data for year 2009; see Table E3 and Table E4 in the Enclosure. For other notations related to the terms of the total cost expression, see equation (1).

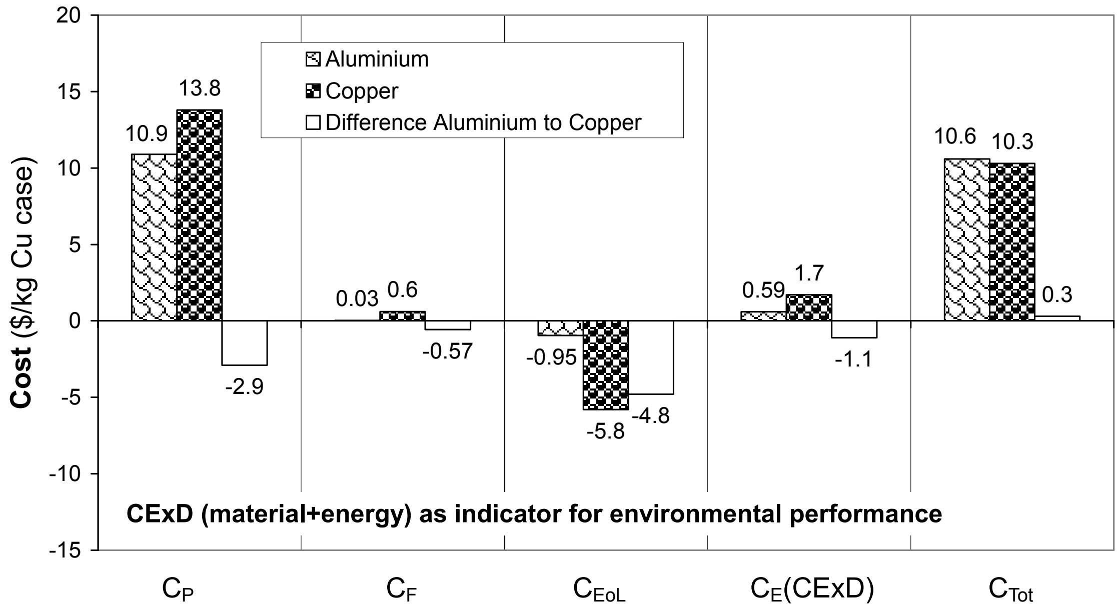


Figure 9. Comparison in total cost between the copper conductor transformer and the aluminium conductor transformer using the modified CExD (material + energy) cost indicator (see section 3.4), and cost data for year 2009; see Table E3 and Table E4 in the Enclosure. For other notation related to the terms of the total cost expression, see equation (1).

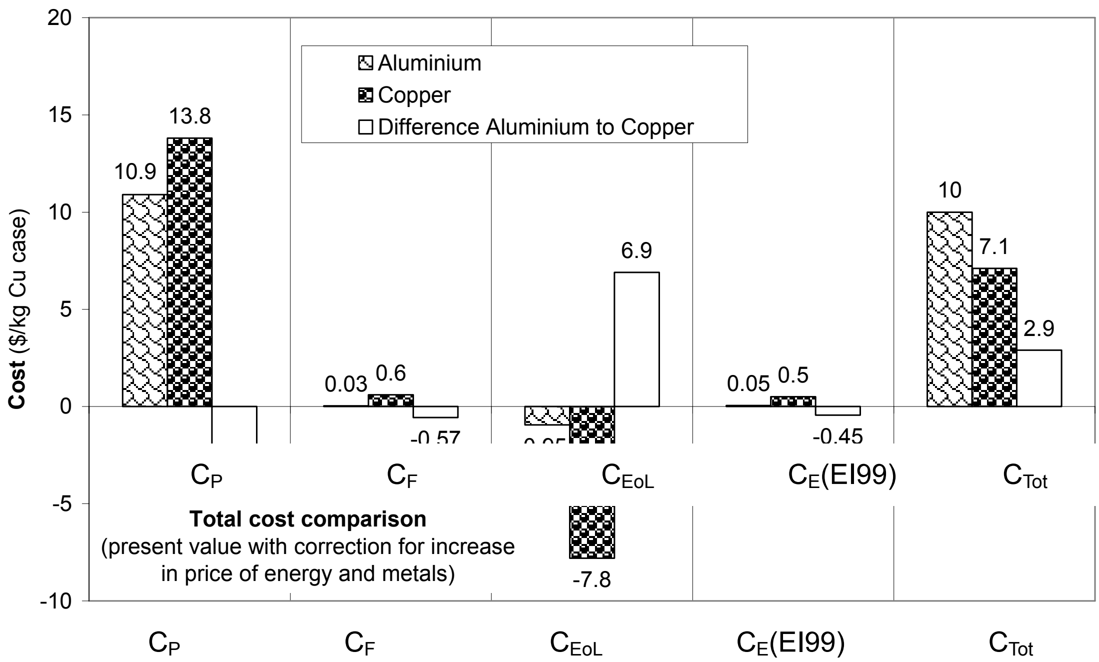


Figure 10. Comparison in present-value-corrected total cost between the metallic copper conductor transformer and the aluminium conductor transformer, assuming 10-year lifetime, an interest rate of 8% per year, increase in price of energy of 10% per year in constant dollar value, of metallic copper equal to 11% per year in constant dollar value, and of metallic aluminium equal to 8% per year in constant dollar value .As in Figure 5 the modified Ecoindicator 99 cost indicator is used and cost data for year 2009; see Table E3 and Table E4 in the Enclosure. For other notations related to the terms of the total cost expression, see equation (1).

Enclosure: Environmental data used in the study

Table E1 Process data for copper used in the analysis (Data from the Ecoinvent database in SimaPro 7, from 2009)

Abbreviations: CED = Cumulative energy demand; CElD = Cumulative electricity demand; CExD = Cumulative exergy demand; EI99RSE = Ecoindicator 99 resources in surplus energy; EPS 2000DR = Environmental Priority System/depletion of reserves, ELU= Environmental load units, el=electricity

Process	CED/CElD	CExD	EI99RSE	EPS 2000DR	Price[1]	Remark
Production of 1 kg of copper primary, at refinery/RER	36 MJ (30.3 non-renewable, 5.6 renewable) 9.3 MJ el (assoc. energy 20.2 MJ)	80.1 MJ eq (44.0 metals, 0.2 minerals, 30.2 non- renewable energy, 5.7 renewable energy)[3]	45.1 MJ eq (42.6 minerals, 2.5 fossil fuel)	283 ELU	\$6.90 (2009) \$1.80 (1999)	Price of ore[2] Waste of copper in terms of copper in ore = 12%.
Production of 1 kg of copper, secondary at refinery/RER	29 MJ (27.2 non-renewable, 1.8 renewable) 5.8 MJ el (assoc. energy 18 MJ)	34.4 MJ eq (5.2 metals, 0.05 minerals, 27.5 non- renewable energy, 1.6 renewable energy)	6.2 MJ eq (4.8 minerals, 1.4 fossil fuel)	32.4 ELU	\$6.90 (2009) \$1.80 (1999)	Loss of copper compensated by adding 11% primary copper
Scrap (burnt wire, solid, US consumer)					\$5.80 (2009) \$1.50 (1999)[4]	
Sheet rolling for production of 1 kg metallic copper conductor strips	24.7 MJ (23.2 non- renewable, 1.5 renewable) 7.0 MJ el (assoc. energy 22 MJ)	27.5 MJ eq (2.6 metals, 0.0 minerals, 23.2 non- renewable energy, 1.7 renewable energy)	2.2 MJ eq (1.1 minerals, 1.1 fossil fuel)	7.0 ELU		Loss of copper compensated by adding 4% primary copper

[1] London Metal Exchange; see http://www.metalprices.com.

[2] Prices of copper ore from sulphide or oxide ores were estimated from data for Chile by Bayer and Winkel (2004) to around \$1.00 per kg of copper in 1999 and \$3.90 per kg in 2009. For copper concentrate they report a price of \$1.30 per kg, valid for 2002. If this price follows the price of primary copper, the price of copper concentrate in 1999 would have been \$1.30 per kg. For 2009 it would be \$5.10 per kg.

[3] Concentration in ore 0.6–1.0%. Concentration in common rock is 0.01%. Dissolution of ore concentration 1% to earth-crust concentration corresponds to an exergy loss of 0.18 MJ/kg of copper.

[4] Estimated.

Table E2 Process data for metallic aluminium used in the analysis (Data from the Ecoinvent database in SimaPro 7, from 2009)

Abbreviations: CED = Cumulative energy demand; CElD = Cumulative electricity demand; CExD = Cumulative exergy demand; EI99RSE = Eco-indicator 99 resources in surplus energy; EPS 2000DR = Environmental Priority System/depletion of reserves, ELU= Environmental load units, el= electricity

Process	CED/CElD	CExD	EI99RSE	EPS 2000DR	Price[1]	Remark
Production of 1 kg of aluminium primary, at plant/RER	195 MJ (160.8 non-renewable, 34.2 renewable) 61.4 MJ el (assoc. energy 144.6 MJ)	203 MJ eq (7.6 metals, 0.2 minerals, 161.8 non-renewable energy, 33.4 renewable energy)[3]	13.9 MJ eq (3.1 minerals, 10.8 fossil fuel)	3.9 ELU	$2.0 (2009) $1.6 (1999)	Price of ore[3] Waste of aluminium in terms of aluminium in ore = 16%
Production of 1 kg of aluminium secondary from new and old scrap, at plant /RER	24 MJ (22.3 non-renewable, 1.7 renewable) 2.0 MJ el (assoc. energy 6.3MJ)	8.8–25.1 MJ eq (0.8 metals, 0.1 minerals, 21.7 non-renewable energy, 2.5 renewable energy)	2.3 MJ eq (0.1 minerals, 2.2 fossil fuel)	0.9 ELU	$ 2.0 (2009) $ 1.6 (1999)	Scrap needed for production of 1kg Al 1.03 kg, where scrap is classified as old scrap
Scrap (cans, USB)					$ 1.9 (2009) $ 1.5 (1999)[4]	
Sheet rolling for production of 1 kg of metallic aluminium conductor strips	11.6 MJ (10.8 (non- renewable, 0.8 renewable) 2.76 MJ el (assoc. energy 8.2 MJ)	11.5 MJ eq (0.11 metals, 0.0 minerals, 10.7 non-renewable energy, 0.7 renewable energy)	0.7 MJ eq (0.04 minerals, 0.7 fossil fuel)	0.2 ELU		Loss of aluminium compensated by adding 1.3% primary aluminium

[1] London Metal Exchange; see <www.metalprices.com>

[2] Prices of Bauxite was \$0.022 per kg (1999) and \$0.031 per kg (2009) as given by USGS (2009) corresponding to roughly to \$0.11 and \$0.16 \$ per kg related to amount of aluminium, respectively

[3] Concentration of aluminium in earth crust is 8% and in Bauxite roughly around 20%. Dissolution of aluminium from Bauxite to concentration in earth crush corresponds to an exergy loss of 0.03 MJ per kg of aluminium

[4] Estimated

Table E3 Data for life cycle of product element according to Figure 1, with metallic copper as metallic conductor

Abbreviations: CED = Cumulative energy demand; CElD = Cumulative electricity demand; CExD = Cumulative exergy demand; EI99RSE = Eco-indicator 99 resources in surplus energy; EPS 2000DR = Environmental Priority System/depletion of reserves

Step 1: Manufacturing of pure metal A from ore containing A: (a) manufacturing, (b) material loss of A during manufacturing

Step 2: Manufacturing of product element: (a) manufacturing, (b) scrap production during manufacturing

Step 3: Product element during its service life

Step 4: Recycling of metal A after use in product element: (a) recycling, (b) material loss of A during service and recycling

Step 5: Addition of metal for production of second generation of product element: (a) supply of stored scrap from production, (b) supply of virgin metal to compensate for material loss of A during service and recycling, (c) loss of material of A associated with process 5b.

Step/Process	1a	1b	2a	2b	3	4a	4b	5a	5b	5c
Amount copper (kg)	1.04	0.12	1.00	0.04	1.00	0.89	0.11	0.04	0.11	0.01
CED (MJ)	37		24			25			3.9	
CElD[1] (MJ)	9.7		6.9			4.8			1.0	
Assoc energy to el (MJ)	21		22			16			2.2	
CExD energy (MJ eq)	37		26			25			4.0	
CExD material (MJ eq)	46		0			0.4			4.9	
Exergy stored (MJ)	2.3	-0.02	2.2	-0.09		2.0	-0.24	0.09	0.24	-0.002
EI99RSE energy (MJ)	2.6		1.0			1.1			0.25	
EI99 RSE material (MJ)	45		0			0.11			4.7	
EPS 2000DR (ELU)	294		0.37			1.3			31	
Financial value ($ 1999)	1.9		See 3.1 or Figure 7	0.06	1.5	1.6		0.07	0.20	0
Cost (mat./en.)[2] ($ 1999)	1.4 (1.2)[3]	0.12	1.9			1.4	0.17	0.06	0.15 (0.13)[3]	0.01
Financial value ($ 2009)	7.2		See 3.1 or Figure 7	0.23	5.8	6.1		0.27	0.76	0
Cost (mat./en.)[2] ($ 2009)	5.6 (4.5)[3]	0.470	7.1			5.4	0.64	0.23	0.60 (0.50)[3]	0.05

[1] Price of electric energy in USA was $0.03 per kWh in 1999 and $0.10 per kWh in 2009.

[2] Cost of energy was roughly estimated from the sum of the price of electricity and (CED – assoc. energy)/3 multiplied by the unit price of electricity. Cost of materials was that for copper concentrate, processes 1a and 5b; that for ore, processes 1b and 5c; that for scrap, process 4a; and that for primary copper, process 2a.

[3] Material cost based on price of ore and energy.

Table E4 Data for life cycle of product element according to Figure 1, with metallic aluminium as metallic conductor

Abbreviations: CED = Cumulative energy demand; CElD = Cumulative electricity demand; CExD = Cumulative exergy demand; EI99RSE = Ecoindicator 99 resources in surplus energy; EPS 2000DR = Environmental Priority System/depletion of reserves

Step 1: Manufacturing of pure metal A from ore containing A: (a) manufacturing, (b) material loss of A during manufacturing

Step 2: Manufacturing of product element: (a) manufacturing, (b) scrap production during manufacturing

Step 3: Product element during its service life

Step 4: Recycling of metal A after use in product element: (a) recycling, (b) material loss of A during service and recycling

Step 5: Addition of metal for production of second generation of product element: (a) supply of stored scrap from production, (b) supply of virgin metal to compensate for material loss of A during service and recycling, (c) loss of material of A associated with process 5b.

Step/Process	1a	1b	2a	2b	3	4a	4b	5a	5b	5c
Amount Al (kg)	0.507	0.08	0.50	0.007	0.50	0.485	0.015	0.007	0.015	0.002
CED (MJ)	99		5.8			12			3.0	
CElD[1] (MJ)	31		1.1			1.0			0.9	
Assoc energy to el(MJ)	73		1.7			3.1			2.1	
CExD energy (MJ eq)	99		5.7			12			3.0	
CExD material (MJ eq)	4.0		0			0.4			0.12	
Exergy stored(MJ)	17	-0.02	16.5	-0.23		16.0	-0.50	0.23	0.48	-0.000
EI99RSE energy(MJ)	5.5		0.04			0.6			0.165	
EI99RSE material (MJ)	1.6		0			0.05			0.05	
EPS 2000DR(ELU)	2.0		0.09			0.4			0.06	
Financial value ($ 1999)	0.8		See 3.1 or Figure 7	0.01	0.75	0.77		0.01	0.02	0
Cost (mat./en.)[2] ($ 1999)	0.38 (0.05)[4]	0.002				0.72	0.03	0.01	0.01	0.000
Financial value ($ 2009)	1.0		See 3.1 or Figure 7	0.01	0.95	0.97		0.01	0.03	0
Cost (mat./en.)[2] ($ 2009)	0.40[3] (0.08)[4]	0.002	1.05	0.01		0.95[3]	0.03	0.01	0.01[3]	0.000

[1] Price of electric energy in USA was \$0.03 per kWh in 1999 and \$0.10 per kWh in 2009.

[2] Cost of energy was roughly estimated from the sum of the price of electricity and (CED – Assoc. energy)/3 multiplied by the unit price of electricity. Cost of aluminium was that for ore, process 1a, 1b, 5b, and 5c; that for scrap, process 4a; and that for primary aluminium, process 2a.

[3] Production of metallic aluminium is made at sites where the electricity price is much less than the industry in general has to pay: Burns (2009) reports a price of \$0.029 per kWh to be more valid in the production of primary aluminium.

[4] Material cost based on price of ore.