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Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC

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Abstract
Purpose A widely used theory of the computational structure of life cycle assessment (LCA) has been available for more than a decade. The case of environmental life cycle cost (LCC) is still less clear: even the recent Code of Practice does not specify any formula to use.
Methods This paper does not aim to resolve all the issues at stake. But it aims to provide an explicit and transparent description of how to calculate the life cycle cost (in whatever way defined), and the value added across the life cycle.
Results and discussion The expressions obtained can be fed into the formulas for eco-efficiency, so that an explicit and reproducible eco-efficiency indicator can be calculated.
Conclusions The results are useful for developing life cycle sustainability analysis, combining LCA, LCC, and social LCA.

Keywords Computational structure · Eco-efficiency · LCA · LCC · LCSA

1 Introduction

Life cycle assessment (LCA) has grown into a widely used method for addressing the environmental aspects of products and services. Obviously, LCA is not "ready" and developments are taking place at many sides: from foundational (e.g., consequential vs. attributional) to practical (e.g., estimating missing data), from goal definition to interpretation, from theory to software (Finnveden et al. 2009). Such developments serve to improve the theory and practice of LCA. This is one aspect of deepening LCA (Guinée et al. 2011). Another development is toward broadening LCA, in two directions (Guinée et al. 2011). First, the object of LCA is broadened, from simple products to more complicated systems, and from microlevel decisions to economy-wide policy choices. Second, the scope of LCA is broadened, from an environmental analysis to a sustainability analysis. The range of impacts addressed has gradually increased, first and foremost in the environmental area: from an initial assessment of waste and energy to climate change, acidification, toxicity, resources, land use impacts, and resource depletion, and additional seminal proposals for thermal pollution, noise, etc. But there is a broadening to other areas as well, both from the demand side and the supply side. Economic aspects in a life cycle perspective are addressed by environmental[1] life cycle costing (LCC; see, e.g., Hunkeler et al. 2008; Swarr et al. 2011), and social life cycle assessment (SLCA; see, e.g., Hunkeler 2006; Dreyer et al. 2006; UNEP/SETAC Life Cycle Initiative 2009) is addressing social aspects. This is a natural development. LCA claims to be a "holistic" approach (Baumann and Tillman 2004). It addresses the entire life cycle of a product, and moreover takes into account many types of impact. The extension to include economic and social criteria in a life cycle perspective is natural, even though the ISO standards themselves are restricted to the "environmental aspects and impacts of a product system" and "Other tools may be combined with

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[1] The term "environmental" needs some explanation. The LCC focuses on a complete array of real costs, so it is not restricted to environmental costs (say, for waste processing), and it does not address hypothetical externalities. The adjective "environmental" refers to the fact that the economic analysis has been made in a way that is consistent with that of the environmental analysis, i.e., that it largely follows ISO 14040.

LCA for more extensive assessments." (ISO 14040, p. 7). Furthermore, the recognition of the concept of sustainability, covering environmental, economic and social aspects (UNEP/SETAC Life Cycle Initiative 2011), provides another motivation. Yet, the role of LCC as a sustainability tool is not immune from criticism (Jørgensen et al. 2010).

LCC is not new, in fact it is older than LCA (Settanni et al. 2011a). But its connection to LCA is more recent, and so is the idea of combining LCA and LCC in a more comprehensive assessment, or even in an integrated way by means of eco-efficiency (E/E; see, e.g., Huppes and Ishikawa 2007). The development of SLCA is newer, and so is the idea of the broadened life cycle sustainability assessment (LCSA; Klöpffer 2012), in which LCA, LCC, and SLCA are combined.

LCA and LCC have been developed quite independently, and so there are differences in terminology, framework, and calculation rules. As long as LCA and LCC are performed and used as separate ingredients of decision-making, inconsistencies between them are not deemed problematic (Settanni 2008). But when LCA and LCC are performed and used together, by the same persons, in the same software, with the same databases, and in an integrated way (either E/E or LCSA), inconsistencies between the two underlying tools will provide a barrier in terms of efficiency, reproducibility, and transparency. ISO has been optimistic on the harmonized use of LCA and LCC: "LCA typically does not address the economic or social aspects of a product, but the life cycle approach and methodologies described in this International Standard can be applied to these other aspects" (ISO 2006, p. vi). As discussed by Huppes et al. (2004), this is not true. Differences in accounting principles, system boundaries, treatment of time, etc., are obstacles, and aligning LCA and LCC is a task to be carried out. Recent work has been undertaken in order to harmonize the set-up and principles of LCA and LCC (Hunkeler et al. 2008; Swarr et al. 2011). Hunkeler et al. (2008) define "conventional LCC" as the "assessments of all costs associated with the life cycle of a product that are directly covered by any 1 or more of the actors in the life cycle" (p. 173). This resembles ISO's definition of LCA as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle". (ISO 2006, p. 2), and as such provides a good starting point for harmonization.

For practical purposes, SLCA, and with that, LCSA is still underdeveloped. Despite some serious data gaps and quality issues, LCA databases are flourishing in different countries, and cost factors are often well-known by the management. Data for social LCA, on the other hand, is often lacking, and this is especially true for quantitative data.

Altogether, SLCA is lagging behind somewhat, but LCA and LCC appear to be well-developed and on track in being harmonized. However, this optimism is not fully grounded. The reason is that a framework, terminology, guidelines, and data do not make an LCA or an LCC. One also needs operational formulae, representing the computational rules that define how the data is to be combined. In particular, the alleged parallel between LCA and LCA will be challenged in this article. This parallel builds on the idea that the principles of LCA "can be applied" (ISO 2006, p. vi) to LCC, and that LCC should use system boundaries "equivalent to" (Hunkeler et al. 2008, p. xxvii) those of LCA. This suggests that, like the environmental impacts of upstream processes, the costs of upstream processes should be included in a life cycle study. But obviously, the costs of iron are included in the costs of steel, which are included in the costs of a car. Equivalent system boundaries and computational rules thus have the danger of double counting, triple counting, or even worse. The computational structure of LCC must therefore be different from that of LCA.

This paper addresses the issue of the computational structure of LCC on the basis of the computational structure of LCA. It first reviews what the recent literature has to say on this (Section 2). Then, Section 3 proposes a computational structure for addressing economic aspects, LCC, or otherwise. Section 4 concludes the paper by pointing out issues to be worked out further.

2 The current computational framework for LCA and LCC

For the case of LCA, the topic of computation is an underemphasized one. As observed by Heijungs and Suh (2002), "there is a large number of guidebooks for applying the LCA technique, but [...] the computational structure of LCA is hardly addressed in these books." (Heijungs and Suh 2002, p. 2). This critique still holds, despite the publication of newer documents. The revised ISO standard for LCA writes: "Based on the flow chart and the flows between unit processes, the flows of all unit processes are related to the reference flow." (ISO 2006, p. 13). No guidance is provided on how to do so. In the recently published ILCD handbook, the situation is not any better: "Determine for each process within the system boundary how much of its reference flow is required for the system to deliver its functional unit(s) and/or reference flows(s) (i.e. the extent to which the process is involved in the system). Scale the inventory of each process accordingly. This way it relates to the functional unit(s) and/or reference flow(s) of the system." (European Commission 2010, p. 273–274).

The situation can be summarized as follows:

- LCA deals with a lot of data (process data, characterization factors, etc.)

- A lot of research has been put in deciding which data to use (marginal, average, etc.) and in gathering the right data (peer-reviewed databases, etc.)
- Hardly any research has been done on the art of combining the data in the correct way

Yet, the subject is certainly not a trivial one. Heijungs and Suh (2002) use some 250 pages to fill this gap for LCA, using several hundreds of matrix equations.

The case of LCC is even worse. Although the books by Hunkeler et al. (2008) and Swarr et al. (2011) are of definite interest in many respects, they suffer from the same shortcomings as the ISO standards for LCA, SETAC's Code of Practice, ILCD's Handbook, and similar texts for LCA in failing to give a precise and general form of the computational structure of LCC (cf. ISO 15686-5 (2008)). The easiest way to demonstrate this is by pointing out that the books contains hardly any formulae. To be precise, Swarr et al. (2011) contains one single formula (on p. 6), and Hunkeler et al. (2008) just five (one on p. 47, one on p. 51, two on p. 52, and one on p. 158). More in general, although the literature on LCC in the context of LCA is vast, to the authors' knowledge there are only a few previous works that critically addressed the methodological issues involved and propose explicitly a computational structure of LCC (Settanni and Emblemsvåg 2010; Settanni et al. 2011b, c). The merit of such works is that they make an effort to apply a computational basis for LCC similar to that for LCA (Heijungs and Suh 2002). However, they focus mainly on the LCC side, with the application of both LCC and LCA being limited to one case (Settanni et al. 2010).

From the perspective of operationalizing environmental LCC in this paper, there is only one formula of interest in Hunkeler et al. (2008) on p. 47, which we cite in full below:

$$\mathrm{LCC} = \sum_{\text{life cycle phase } 1}^{\text{life cycle phase } n} \sum_{\text{process } 1}^{\text{process } i} \left(\mu_i \times \sum_{\text{cost el. } 1}^{\text{cost el. } p} \sum_{\text{flow } 1}^{\text{flow } q} \text{amount}_p \times \text{costs}_p \right) \tag{1}$$

where μ is a "process scaling factor related to the product system".

This formula is interesting to the extent that it shows the only explicit recipe on calculating the life cycle cost shown in a book explicitly aiming to be a "precursor to a code of practice". However, it is not clear enough. First, it does not show how to calculate the process scaling factors μ. For this, we could refer to the explicit scaling factors by Heijungs and Suh (2002), who offer an equation like

$$\mathbf{s} = \mathbf{A}^{-1}\mathbf{f} \tag{2}$$

where the symbol $\mathbf{s}$ now represents the scaling factors which are μ in Hunkeler et al. (2008). Next, the formula displays some unconventional uses of the summation symbol. A more standard form would be like below

$$\mathrm{LCC} = \sum_{n=1}^{\text{all life cycle phases}} \sum_{i=1}^{\text{all processes}} \left(\mu_i \times \sum_{p=1}^{\text{all cost el.}} \sum_{q=1}^{\text{all flows}} \text{amount}_p \times \text{costs}_p \right) \tag{3}$$

This, however, makes explicit that two of the summation indices (n and q) are missing as a subscript to other symbols in the formula itself, which is another riddle.

More fundamentally, it is not clear how the symbol $\mathbf{A}$ relates to the notation of this formula. An element a_{ij} of $\mathbf{A}$, in Heijungs and Suh (2002), refers to amounts, namely the amount of product i used (when negative) or produced (when positive) by process j. We might thus try to identify

$$\text{amount}_p = -a_{qi} \tag{4}$$

where the minus sign accounts for the idea that inputs usually refer to costs and outputs to proceeds. This is, however, just a speculative attempt, with still issues to be resolved (e.g., where are n and p in a_{qi}?). Moreover, the formula is just stated, without any proof or argument.

An even more disturbing problem in interpreting the concept of LCC in general is related to the fact that there is a problem with defining the life cycle costs in relation to the concept of the "physical" life cycle which is studied in LCA. For the business management literate, the concept of LCC is a rather straightforward aspect of any procurement policy: it is all about extending the cost of purchasing some durable asset or equipment, by including the post-purchase costs that will likely be incurred by the producer and the user of that durable good, during its economic life (Settanni et al. 2011a, and references therein).

The concept of life cycle embodied in LCC is evidently different from the one in LCA. From the perspective of the user of a car, the life cycle costs are made up by the costs for purchasing the car, for buying gas, for maintenance, for disposal, for insurance, etc., all occurring at certain times during the car's economic life. But the upstream costs of mining the metals, for drilling and processing the oil, or for designing and manufacturing the car are not part of the life cycle costs. Such activities are part of the "physical" life cycle, or supply chain, but the costs associated with them are not seen explicitly as part of the life cycle costs for the user's procurement purposes. Consider the car producer and the gas station owner: these actors charge a price for their products that is partly the result of their cost factors and partly the result of an economic market. Swarr et al. (2011, p. 16) point to this issue by their clause "while avoiding double-counting", but the example in their Table 3-1a lists a "Total" that includes cost items from "R&D", "Pre-production", and "Production" that will typically be included in the price.

In general, all the costs that are borne upstream in a certain stage of the supply chain are summarized in the price of any good or service purchased at that stage. The costs incurred for purchasing inputs produced in other stages are then added with the stage-specific conversion costs (e.g., labor and overheads) and mark-ups (profit margins), and then "transferred into" the next operational stage, thus contributing in their turn to the production cost of the latter's output. As transactions among actors operating at different stages take place, the resulting "flow of costs" conceptually mimics the flow of materials and other inputs[2] along the chain. Hence, there is evidently some room for integrating how the two analyses, usually kept separated, can relate to each other, using similar computational framework—or even a single harmonized computational framework.

From the above, it follows that if we simply add the costs of "all life cycle phases", we will do quite some double-counting. If a car costs you 10,000€ and the operational costs over its life are 40,000€, the life cycle costs can be summarized as 50,000€. It would be incorrect to add to this the 6,000€ for materials and components purchased by the car producer that are already charged as part of the 10,000€ for the car itself.

This raises some fundamental questions: What do we in fact want to learn from life cycle costing? Does that indeed require the full life cycle? Is specifying the costs of a limited set of "foreground processes" not enough? Is it perhaps something else that we're looking for, e.g., value added along the supply chain? In the next section, we will approach these questions by studying a simple case.

3 A computational framework for LCC

As mentioned, general frameworks already exist which use matrix algebra as a common ground to integrate LCC and LCA. Nakamura and Kondo (2009) proposed the use of input–output analysis for both LCC and LCA purposes. Their framework however works at a more macro-economic level. A supply chain view is adopted instead by Settanni et al. (2011b, c); this contribution can be regarded in the realm of environmental management accounting rather than.

This paper improves the existing proposals, which remain mainly theoretical, emphasizing the practical interaction of LCA and LCC as it would work, e.g., in a software package. This is particularly important, since several LCA software providers claim their programs (such as GaBi and SimaPro) can carry out an integrated LCA and LCC analysis. Given the lack of documentation, we guess that they uncritically and not transparently apply formulations of LCC taken from the managerial literature which are not consistent with LCA. Even an explicit report on LCC and SimaPro (Ciroth and Franze 2009) fails to give details.

In this section, we will start with a simple example, and develop matrix-based expressions for the life cycle costs, fully compatible with the concepts and notation of the LCA. Throughout the example, we will, for simplicity, assume that prices are homogenous, in one currency, and free from taxes and subsidies. We will also assume that there are no environmental costs, e.g., emission tariffs or mining concessions. Later on, we will come back to these issues.

3.1 A simple but instructive example

We start with the life cycle cost of a stand-alone product. With a stand-alone product, we denote a product that is used as such, without any other products, so for which the use process requires only the product. All products that require fuel or electricity (such as cars and refrigerators) are clearly not stand-alone products, as they require the product and some energy source. Neither are products that require regular maintenance (such as roads) stand-alone products. But a chair and a pair of sunglasses come close to being a stand-alone product.

A simplified flow diagram for a chair is shown in Fig. 1. It is, among others, simplified to the extent that maintenance has been excluded; see Section 3.3 for an extended system with maintenance. In our example, we will use the perspective of the user of the product.

Let us assume some—purely hypothetical—data as in Table 1. All these data are specified per standard amount

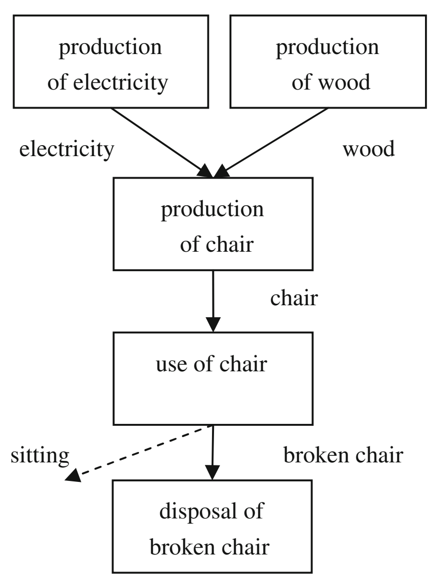

Fig. 1 A simplified flow diagram for the life cycle of a chair. *Boxes* are unit processes, *arrows* are flows of products and services. The *dashed arrow* represents the function of the use process, which is taken here as the reference flow

[2] In principle, this can even include labor; see Rugani et al. 2012.

Table 1 Hypothetical data for the processes and flows in Fig. 1. Physical outflows are denoted with a positive amount, inflows with a negative amount. Goods (like wood) have a positive price, and wastes (the broken chair) have a negative price (Note well that "waste" refers here to an undesired residual product, not to the service of waste handling. We could rephrase every output of waste as an input of waste handling, and every input of waste as an output of waste handling. This is the convention employed by some other authors and databases (such as ecoinvent). Using that convention, we would have to redraw Fig. 1 such that the process "use of chair" has an input of "treatment of broken chair" instead of an output of "broken chair"). Positive transactions represent revenues (benefits) for the process owner; negative transactions represent expenses (costs)

Process	Product	Physical amount	Market price per unit	Amount of transaction (€)
Production of electricity	Electricity	1 MJ	5 €/MJ	5
Production of wood	Wood	1 kg	1 €/kg	1
Production of chair	Electricity	−2 MJ	5 €/MJ	−10
Idem	Wood	−5 kg	1 €/kg	−5
Idem	Chair	1 piece	25 €/piece	25
Use of chair	Chair	−5 pieces	25 €/piece	−125
Idem	Broken chair	5 pieces	−2 €/piece	−10
Idem	Sitting	10 year	0 €/year	0
Disposal of broken chair	Broken chair	−1 piece	–2 €/piece	2

of output (or input), not in the way they are used in this specific life cycle. Table 1 also lists the prices of each product. Observe that these are indeed prices, not production costs.

The functional unit is the basis for scaling the data to a common metric. Let us take 1 year of sitting as the reference flow. It is easy to see that we need to scale the use process by a factor 0.1. That scales the input of the use process to 0.5 chair. This in its turn determines that we need to scale the production of chairs with a factor 0.5, so that we need 1 MJ of electricity and 2.5 kg of wood. In this way, we can work along the entire life cycle, to obtain the scaling factors of Table 2.

In a traditional LCA, the scaling factors are the first step to compiling the inventory table, as the environmental extensions that accompany each process specifications are to be scaled with these factors as well (Heijungs and Suh 2002). The scaling factors also provide a clue to the economic analysis, however in a more limited way (Table 3).

First note that the price charged for a chair is 25€, and that the user of the chair needs 0.5 chair for 1 year of sitting, so he will have to spend 12.5€ on chairs. He does not need to pay for the wood or the electricity to make these chairs, as such the costs for these factors of production are borne by the chair producer, and co-determine the price of a new chair. So, the life cycle cost of the chair usually does not exhibit the costs of wood and electricity. It does, however, comprise more than the cost of the chair only, because the disposal of the chair will cost the user 1€ for 1 year of sitting. Thus, the life cycle cost of the chair is 13.5€ for 1 year of sitting: 12.5€ for purchasing the chair itself, and 1€ for purchasing the waste treatment of the chair after use.[3]

For every process, we can make a balance of costs and proceeds. The chair producer, for instance, pays for wood (5€) and electricity (2.5€) and receives for chairs (12.5€). Thus, in producing a half chair, he has a value added[4] of 5€, part of which is required for labor costs, rent, and taxes, and part of which is profit. Table 4 shows the balance per process as the value added.

One observes that the value added for all processes, except for the use process, is positive. Indeed, no feasible economy will tolerate activities with a negative value added, unless a government decides to subsidize it (as is done for symphony orchestras and defense operations). How should we understand the negative value added for the use process, in this case of using a chair? The point is that using a chair requires money (for buying the chair and for disposing the broken chair), and does not create proceeds, but rather creates a utility ("the joy of sitting"), for which we are

Table 2 Scaling factors for the example data of Table 1 and a reference flow of 1 year sitting

Process	Scaling factor
Production of electricity	1
Production of wood	2.5
Production of chair	0.5
Use of chair	0.1
Disposal of broken chair	0.5

[3] Note well, this is not the life cycle cost of the chair itself, but of 1 year of sitting. As the chair lasts for 2 year, the life cycle costs of the chair itself is 27€; for 1 year of sitting, the value is thus an average over the life span.

[4] Value added is defined as the difference between the value of the outputs minus the value of the inputs purchased from others (Lipsey et al. 1989).

Table 3 The data of Table 1, scaled to a reference flow of 1-year sitting with the scaling factors of Table 2

Process	Product	Physical amount	Market price per unit	Amount of transaction (€)
Production of electricity	Electricity	1 MJ	5 €/MJ	5
Production of wood	Wood	2.5 kg	1 €/kg	2.5
Production of chair	Electricity	−1 MJ	5 €/MJ	−5
Idem	Wood	−2.5 kg	1 €/kg	−2.5
Idem	Chair	0.5 piece	25 €/piece	12.5
Use of chair	Chair	−0.5 pieces	25 €/piece	−12.5
Idem	Broken chair	0.5 pieces	–2 €/piece	−1
Idem	Sitting	1 year	0 €/year	0
Disposal of broken chair	Broken chair	−0.5 piece	−2 €/piece	1

apparently prepared to pay at least the costs, and perhaps even more. But it is not a commodity that is exchanged on a market, so there is no price for it. Might we be prepared to pay more for it, we are lucky to be cheap off, and we are experiencing a consumer surplus. At any rate, the (negative) value added at the use process is in this example equal to the intuitive and informally defined life cycle costs. This is a first clue to extracting life cycle costs from a monetary analysis of the "physical" life cycle.

3.2 Matrix formulation of life cycle costing

Let us now try to make a step toward a more general approach. The matrix-based LCA will serve as a starting point for this. We construct a linear space, in which the rows indicate the products (megajoules of electricity, kilogram of wood, piece of chair, piece of broken chair, and year of sitting), and the columns indicate the processes (production of electricity, production of wood, production of chair, use of chair, and disposal of broken chair). The technology matrix $\mathbf{A}_\mathrm{p}$ of this "physical" system is given by

$$\mathbf{A}_p = \begin{pmatrix} 1 & 0 & -2 & 0 & 0 \\ 0 & 1 & -5 & 0 & 0 \\ 0 & 0 & 1 & -5 & 0 \\ 0 & 0 & 0 & 5 & -1 \\ 0 & 0 & 0 & 10 & 0 \end{pmatrix} \quad (5)$$

Table 4 Value added for the example data of Table 1 and a reference flow of 1 year sitting

Process	Value added (€)
Production of electricity	5
Production of wood	2.5
Production of chair	5
Use of chair	–13.5
Disposal of broken chair	1

and the final demand vector $\mathbf{f}_\mathrm{p}$ is chosen to be

$$\mathbf{f}_p = \begin{pmatrix} 0 \\ 0 \\ 0 \\ 0 \\ 1 \end{pmatrix} \quad (6)$$

The scaling factors $\mathbf{s}$ are obtained through the usual

$$\mathbf{s} = \left(\mathbf{A}_p\right)^{-1}\mathbf{f}_p = \begin{pmatrix} 1 \\ 2.5 \\ 0.5 \\ 0.1 \\ 0.5 \end{pmatrix} \quad (7)$$

Scaled to the functional unit, embodied in $\mathbf{f}$, the process matrix reads

$$\mathbf{A}_{p,\text{scaled}} = \mathbf{A}_p \mathrm{diag}(\mathbf{s}) = \begin{pmatrix} 1 & 0 & -1 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 0.5 & -0.5 & 0 \\ 0 & 0 & 0 & 0.5 & -0.5 \\ 0 & 0 & 0 & 1 & 0 \end{pmatrix} \quad (8)$$

where diag($\mathbf{s}$) represents the square matrix with the elements of vector $\mathbf{s}$ on the diagonal. So far, no news.

The first step in adding the formalism of LCC is by defining a price vector $\boldsymbol{\alpha}$:

$$\boldsymbol{\alpha} = \begin{pmatrix} 5 \\ 1 \\ 25 \\ -2 \\ 0 \end{pmatrix} \quad (9)$$

where the implicit choice of a uniform currency (Euro) is assumed. Using the ingredients we have now available, we can form several new quantities of interest. First, we can rephrase the "physical" technology matrix $\mathbf{A}_p$ into its monetary form $\mathbf{A}_m$:

$$\mathbf{A}_m = \mathrm{diag}(\boldsymbol{\alpha})\mathbf{A}_p = \begin{pmatrix} 5 & 0 & -10 & 0 & 0 \\ 0 & 1 & -5 & 0 & 0 \\ 0 & 0 & 25 & -125 & 0 \\ 0 & 0 & 0 & -10 & 20 \\ 0 & 0 & 0 & 0 & 0 \end{pmatrix} \tag{10}$$

Likewise, the monetary form of the final demand vector $\mathbf{f}$ is

$$\mathbf{f}_m = \mathrm{diag}(\boldsymbol{\alpha})\mathbf{f}_\mathrm{p} = \begin{pmatrix} 0 \\ 0 \\ 0 \\ 0 \\ 0 \end{pmatrix} \tag{11}$$

Observe that $\mathbf{f}_m$ and the last row of $\mathbf{A}_m$ contain only zeros, due to the zero price (in fact, the absence of a market) of the service of sitting. We will come back to this later.

The monetary matrix can be expressed in scaled form as

$$\mathbf{A}_{m,\mathrm{scaled}} = \mathbf{A}_m \mathrm{diag}(\mathbf{s}) = \begin{pmatrix} 5 & 0 & -5 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 12.5 & -12.5 & 0 \\ 0 & 0 & 0 & -1 & 1 \\ 0 & 0 & 0 & 0 & 0 \end{pmatrix} \tag{12}$$

This helps us to calculate the value added per process $\mathbf{v}$ is calculated as

$$\mathbf{v} = \left(\mathbf{A}_{m,\mathrm{scaled}}\right)^T \mathbf{1} = \begin{pmatrix} 5 \\ 2.5 \\ 5 \\ -13.5 \\ 1 \end{pmatrix} \tag{13}$$

where $\mathbf{1}$ is a vector of ones if appropriate length, also known as the summation operator.

Finally, the life cycle cost for the product at stake. In the simple case here, it is equal to the "value lost", the negative of the value added, for the use process. This number, 13.5, represents the costs made for getting the product and disposing it after use. Formally, we can extract the life cycle cost, l, using

$$l = -v_j \tag{14}$$

where j represents the reference process, i.e. the process that fulfills the reference flow. The reference flow is the flow i that has a non-zero entry in the final demand vector, and it is related to the reference process j by the requirement that a_{ij} is positive. Thus, we have

$$l = -v_j, \text{ where } j \text{ is such that } a_{ij} > 0, \\ \text{where } i \text{ is such that } f_i \neq 0 \tag{15}$$

In the example, the reference flow is i=5 (year of sitting) and the reference process is j=4 (use of chair).

Observe that we have defined the life cycle cost as the negative value added for the use process, but have calculated much more than that: the value added in all processes of the life cycle.[5]

3.3 A more complicated example

Non-stand-alone products are products that require another product to work. In life cycle costing, these represent the interesting cases, where a trade-off between purchase cost and operational cost may be at work. For the framework described, there are no consequences. The formulas take all activities into account that are defined to be needed for obtaining the consumer utility. The only issue is that one needs to carefully define the use process. Below, we give an example of this.

We modify the chair example, by introducing maintenance: the chair will be cleaned with a spray once per year; see Fig. 2. Spray production requires only electricity in this example.

We add a row to matrix $\mathbf{A}$ to account for "kg of spray", and assign a value to the coefficient at this row and the column for the process "use of chair". Let us say we need 10 g of spray for 1 year of sitting, so we insert a coefficient −100 g of spray/10 year of sitting. We also add a column to store information on how spray is produced in a newly defined process: the spray producer needs 50 MJ of electricity to produce 1 kg of spray. The new technology matrix is shown below.

$$\mathbf{A}_p = \begin{pmatrix} 1 & 0 & -2 & 0 & 0 & -50 \\ 0 & 1 & -5 & 0 & 0 & 0 \\ 0 & 0 & 1 & -5 & 0 & 0 \\ 0 & 0 & 0 & 5 & -1 & 0 \\ 0 & 0 & 0 & 10 & 0 & 0 \\ 0 & 0 & 0 & -100 & 0 & 1,000 \end{pmatrix} \tag{16}$$

The final demand vector is extended by one extra row, containing the value zero. Let us finally assume that spray costs 1€/g. The new system can be subject to the calculation rules as before, yielding a new table of value added (Table 5) and a new life cycle cost.

As we see, the life cycle costs have increased by 10 to 23.5€, and the value added along the chain has changed: production of electricity has gone up by 2.5€ and production of spray has appeared.

[5] The more remote parts of the life cycle, such as car production and oil refining, play no role in determining the life cycle cost. They are of course important in a broader economic analysis, where the value added over the full life cycle is of concern. In the context of eco-efficiency analysis, one may determine the ratio of environmental impact and value added for all processes in the life cycle.

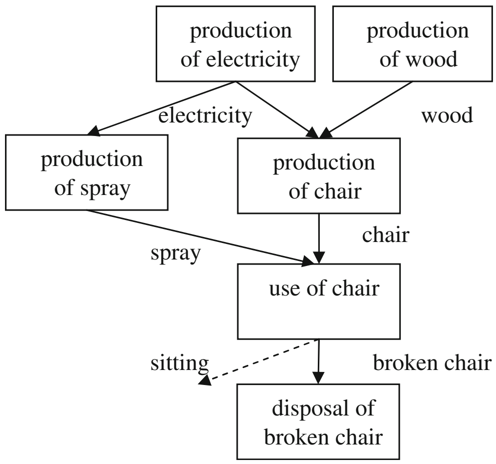

Fig. 2 Modification to Fig. 1 with the use process requiring maintenance by spraying

3.4 Some further complications

In the exposition of life cycle costs and value added, a number of simplifications were introduced. We will briefly discuss a few of the complications that can arise when these are relaxed:

- price inhomogeneity
- cost of environmental services
- costs of utility services
- non-final-use products
- multiple reference flows
- discounting

Price inhomogeneity refers to the fact that the same product can have a different price for different customers. It is a subject extensively discussed in input–output economics (see, e.g., Miller and Blair 2009). So-called price layers are distinguished to account for differences between producer prices, purchaser prices and basic prices, where trade and transport margins and taxes less subsidies make the difference. In the context of the activity-based system of this article, things are far easier, at least in theory. A transport process is a column with the untransported product as an input, and the transported product as an output, possibly along with inputs of fuel and trucks, and outputs of used trucks and pollutants (Heijungs and Suh 2002). The difference in price between the untransported and the transported product is just part of the information of the price vector. Using this format, we can easily calculate the value added in the transport activity, and there is no need to add transport margins. For trade activities, the situation is likewise. Taxes and subsidies are just part of the apparent value added, and they introduce another row and column, a row distinguishing the product before and after taxation, and a column for the "taxation process".

Environmental considerations can complicate the scheme more profoundly, but not fundamentally. The extent to which it does so differs, depending on what one means with "environment". Costs associated with waste treatment are just part of the scheme. The example of the broken chair with a negative price illustrates this. Things are more complicated when waste refers to low-valued recyclable products (like used newspapers), residuals that are disposed of seemingly freely (like old batteries), and releases to the environment (pollutants, like SO_2). First and foremost, in our accounting scheme, we should strictly adhere to the principle that a product has a positive price and a waste a negative price.[6] Alleged residuals or waste products with a positive value, like used newspapers, are thus to be considered as a product, not as a waste. For a consumer, they can imply a benefit, and this should be subtracted from the life cycle cost. Our framework with positive and negative prices works effortless in this respect. In practice, we often deal with used products that have no price, and that are only valuable after collection and/or separation. Thus, the consumer can freely dispose of old paper, and the municipality or organization that collects it will get revenue. The user does not receive any benefits, and it seems fair to exclude this from the LCC framework. In other cases, the municipality pays for waste treatment services, and uses a tax system to claim these costs to the citizens in a way that is unrelated to the use of the product. Whether to include such social costs to the user, is partly matter of perspective, but not exclusively, because we should take care not to double count a pollutant by including it in the environmental analysis and in the economic analysis by means of a shadow cost. Environmental flows may have costs as well. Pollutants may be subject to taxation, emission trading, or other forms of costs, and resource extractions (including land use) may be subject to costs for rent, concessions or otherwise. For environmental flows, we can define a second price vector $\boldsymbol{\beta}$. Care should be taken with the signs of its

Table 5 Value added for the modified example with a reference flow of 1 year sitting, including yearly maintenance by spraying

Process	Value added (€)
Production of electricity	7.5
Production of wood	2.5
Production of chair	5
Use of chair	–23.5
Disposal of broken chair	1
Production of spray	7.5

[6] As pointed out by one reviewer, this is not always realistic, as the example of free cell phones demonstrates; see also Nakamura and Kondo (2009). For the sake of simplicity, we leave these complications for future elaboration.

elements: if we associate an outflow with a positive number, the price for emissions should be negative, while it should be negative for resources, as their input (negative) should be penalized. The formulas for the technology matrix can be repeated for the intervention matrix. For instance, for the scaled monetarized intervention matrix, we have

$$\mathbf{B}_{m,\text{scaled}} = \text{diag}(\boldsymbol{\beta})\mathbf{B}\text{diag}(\mathbf{s}) \tag{17}$$

The value added for each process should now be modified to take the environmental costs into account.

$$\mathbf{v} = \left(\mathbf{A}_{m,\text{scaled}}\right)^{\mathrm{T}}\mathbf{1} + \left(\mathbf{B}_{m,\text{scaled}}\right)^{\mathrm{T}}\mathbf{1} \tag{18}$$

The possible price of environmental services leads to a similar revision of the formula for the life cycle cost: direct emissions and direct resource extractions of the reference process are to be added to the life cycle costs. By deriving the life cycle costs from the value added vector, this is automatically done whenever we adapt the formula for the value added in the aforementioned way.

Costs of waste treatment is one example of a cost for a utility service that is not always included in the price of a product. There are also examples outside the waste sphere. Costs of infrastructure are often borne by tax payers, not by users of the infrastructure. Roads, bridges, harbors, broadcasting stations, there is a long list of general purpose utilities that be included in a life cycle study of a car, a ship, or a television, but for which the costs are not naturally attributable to the product. Like with the waste treatment, it is a matter of taste (or convention) whether or not to do so.

Nonfinal-use products are a frequent subject of LCAs. Typical examples are the cradle-to-gate studies on materials and components, such as steel, plastic, electricity, and engines. In an LCA perspective, such studies are special, because they lack the use phase and often the disposal phase as well, just because the disposal depends on the type of use. In the LCC, there is one more complication: the product is, unlike the consumer utility in the first example, a marketable commodity, and has therefore a price. As a consequence, the reference process will not only be associated with costs, but also with proceeds, and typically with proceeds that are higher than the costs. Using the formulas derived above, this would yield a negative life cycle cost, because the life cycle cost has been defined as the negative of the value added (the value lost) in the reference process, and there is a positive value added in most cradle-to-gate studies. Probably, the concept of life cycle cost for an incomplete life cycle doesn't make any sense at all.

We have a situation with multiple reference flows when the final demand vector **f** is not of the usual LCA form with zeros everywhere except for one entry, but when more than one of the entries of **f** is non-zero. In LCA studies with system expansion, one includes additional functions, and in LCA (or IOA) studies focusing on the impacts of household consumption, the final demand vector may specify a complete commodity basket. The life cycle cost of such a composite final demand is simply the sum of the life cycle cost of its ingredients:

$$l = \sum_j -v_j, \text{where } j \text{ runs over all cases such that } a_{ij} > 0, \text{ where } i \text{ is such that } f_i \neq 0 \tag{19}$$

Traditionally, LCA adds emissions of CO_2 during production to emissions of CO_2 during use and during waste treatment into one overall CO_2 emission. In cost accounting, aggregation of costs in the past, now, and in the future are aggregated using a time preference. In fact, most mathematical treatments of LCC focus on how to introduce discounting into the scheme, calculating the net present value. The key to introducing discounting in the matrix-based LCC formalism is in having separate entries for costs in different years, and using the discounting formulas in the end to aggregate these into a net present value. We will not elaborate the formulas in this more steady-state oriented paper.

3.5 Some observations

In Section 3.2, we saw that the monetary form of the technology matrix can have a row of zeros, and that the final demand can consist of zeros only. This is an important observation: it demonstrates the primacy of a physical approach in which the unpriced function of sitting steers the system. In a monetary system, the inventory equation reads

$$\mathbf{A}_m\mathbf{s}_m = \mathbf{f}_m \tag{20}$$

and it should be solvable as

$$\mathbf{s}_m = (\mathbf{A}_m)^{-1}\mathbf{f}_m \tag{21}$$

from it would follow with the help of Eqs. (10) and (11) that

$$\mathbf{s}_m = \left(\text{diag}(\boldsymbol{\alpha})\mathbf{A}_p\right)^{-1}(\text{diag}(\boldsymbol{\alpha})\mathbf{f}) = \mathbf{A}_p{}^{-1}\mathbf{f}_p = \mathbf{s}_p \tag{22}$$

which means that the scaling factors for the physical and the monetary descriptions are the same. However, in practice, the presence of unpriced products will create a matrix $\mathbf{A}_m$ with one or more rows that comprise only zeros. Such a matrix is singular and cannot be inverted, so $\mathbf{s}_m$ can be incomputable. This suggests that we should consistently use the physical description for the basic calculations, and only derive the monetary description from the physical information, not the other way around. However, this only makes sense if the opposite case (non-zero monetary entries with a zero physical entries) does not occur. This is not necessarily so. If we take "physical" to stand for "mass", as most authors in physical input–output analysis do (Weisz and Duchin 2006), there are definitely flows with economic value but without mass. Electricity is just one example, but all services (transport, communication, haircuts) suffer from this limitation. In a multiphysical set-up, where all products and services can go in their natural units, we do not run into this issue. Materials can be expressed in kilogram, electricity in megajoules, haircuts in "number of". Some services can be expressed in terms of a time, e.g., hours of lawyer council. Some services may even best be expressed in terms of a monetary unit. So, we should understand the primacy of the "physical" framework as the primacy of the framework of "natural units", many of which are in mass units, but also covering other physical and nonphysical units.

The above comes into play especially when we consider the difference of the concept of life cycle in LCA and LCC. LCA focuses on the environmental dimensions, and thereby includes life cycle stages that are relevant from an environmental point of view, even when they are not important from an economic point of view. Lubricants, for instance, form in general a negligible cost for a mechanical device, but they can be quite harmful due to their toxicity. Mutatis mutandis, LCC includes important cost aspects, even when these are irrelevant from an environmental perspective. Advertisements and R&D are examples: expensive, but almost never dominating the LCA. The consequence is that the life cycle in LCA and LCC may be different in practice. The LCC may include R&D, the LCA in practice excludes it. On the other hand, the LCA may include lubricants, whereas the LCC will generally exclude them. More fundamentally, however, both types of analyses should include them. The contribution to the final result will become evident automatically once calculation results have been obtained.

4 Conclusions

Table 6 summarizes the main expressions for the LCA and LCC.

Let us summarize by reiterating the formula of Hunkeler et al. (2008):

$$\mathrm{LCC} = \sum_{\text{life cycle phase 1}}^{\text{life cycle phase } n} \sum_{\text{process 1}}^{\text{process } i} \left(\mu_i \times \sum_{\text{cost el. 1}}^{\text{cost el. } p} \sum_{\text{flow 1}}^{\text{flow } q} \text{amount}_p \times \text{costs}_p \right) \tag{23}$$

and compare to our

$$l = \sum_j -v_j, \text{where } j \text{ runs over all cases such that } a_{ij} > 0, \text{ where } i \text{ is such that } f_i \neq 0 \tag{24}$$

which we can develop into

$$l = \sum_j -\left[\left(\mathrm{diag}(\boldsymbol{\alpha}) \mathbf{A}_p \mathrm{diag}\left(\mathbf{A}_p{}^{-1} \mathbf{f} \right) \right)^{\mathrm{T}} \mathbf{1} + \left(\mathrm{diag}(\boldsymbol{\beta}) \mathbf{B}_p \mathrm{diag}\left(\mathbf{A}_p{}^{-1} \mathbf{f} \right) \right)^{\mathrm{T}} \mathbf{1} \right], \text{ where } j \text{ runs over all cases suchthat } a_{ij} > 0, \text{where } i \text{ is such that } f_i \neq 0 \tag{25}$$

Our newly developed formula looks less attractive, but it has the following advantages:

- it is written in a matrix notation consistent with contemporary LCA
- it employs largely the same symbols as LCA
- it explicitly avoids double-counting by excluding the upstream costs that are already part of the prices of the components of a product
- it is also valid for situations of multiple reference flows
- it contains the possibility of including environmental taxes

In the matrix-based LCA software CMLCA, price vectors for products (α) and elementary flows (β) can be introduced, so that the expression for the value added per process and the life cycle cost can easily be carried out along with a traditional LCA; see Fig. 3.

The big advantage, of course, is that all methodological choices (system boundaries, allocation, etc.) are made consistently for the LCA and the LCC. Moreover, combining LCA and LCC on the same technosystem in one calculation model saves the practitioner to specify twice the flow diagram and data. Finally, considerations of eco-efficiency, leading to expressions in which elements of the LCA and the LCC are combined in some ratio, are easily implemented. In addition to comparing alternatives in terms of eco-efficiency, we may also find those processes in a life cycle that have the highest or lowest eco-efficiency.

A final word on the prospect of even going further into LCSA. While Klöpffer (2008) simply equates LCSA to the combination of LCA, LCC, and SLCA, we have earlier commented on the necessity and efficiency of harmonizing these three tools (Heijungs et al. 2009; Heijungs 2010). In doing so, we proposed to model a technological system that

Table 6 Overview of the main symbols used and expressions derived for LCA and LCC

Item	Physical form	Monetary form
Technology matrix	$\mathbf{A}_p$	$\mathbf{A}_\mathrm{m} = \mathrm{diag}(\boldsymbol{\alpha})\mathbf{A}_p$
Intervention matrix	$\mathbf{B}_p$	$\mathbf{B}_m = \mathrm{diag}(\boldsymbol{\beta})\mathbf{B}_p$
Final demand vector	$\mathbf{f}_p$	$\mathbf{f}_m = \mathrm{diag}(\boldsymbol{\alpha})\mathbf{f}_p$
Scaling vector	$\mathbf{s} = \mathbf{s}_p = \mathbf{A}_p{}^{-1}\mathbf{f}_p$	$\mathbf{s}_m = \mathbf{s} = \mathbf{s}_p$
Scaled technology matrix	$\mathbf{A}_{p,\mathrm{scaled}} = \mathbf{A}_p\mathrm{diag}(\mathbf{s})$	$\mathbf{A}_{m,\mathrm{scaled}} = \mathbf{A}_m\mathrm{diag}(\mathbf{s})$
Scaled intervention matrix	$\mathbf{B}_{p,\mathrm{scaled}} = \mathbf{B}_p\mathrm{diag}(\mathbf{s})$	$\mathbf{B}_{m,\mathrm{scaled}} = \mathbf{B}_m\mathrm{diag}(\mathbf{s})$
Value added	–	$\mathbf{v} = \mathbf{A}_{m,\mathrm{scaled}}{}^{\mathbf{T}}1 + \mathbf{B}_{m,\mathrm{scaled}}{}^{\mathbf{T}}1$
Life cycle cost	–	$l = \sum -v_j$, where j runs over all case such that $a_{ij}>0$, where i is such that $f_i \neq 0$

is common to LCA, LCC, and SLCA (represented by the matrix **A**), and three separate satellite systems, one for LCA ($\mathbf{B}_\mathrm{env}$), one for LCC ($\mathbf{B}_\mathrm{econ}$), and one for SLCA ($\mathbf{B}_\mathrm{soc}$). The formula for calculating results then was supposed to look like

$$\begin{pmatrix} \mathbf{g}_\mathrm{env} \\ \mathbf{g}_\mathrm{econ} \\ \mathbf{g}_\mathrm{soc} \end{pmatrix} = \begin{pmatrix} \mathbf{B}_\mathrm{env} \\ \mathbf{B}_\mathrm{econ} \\ \mathbf{B}_\mathrm{soc} \end{pmatrix} \mathbf{A}_p{}^{-1}\mathbf{f} \tag{26}$$

Now, after the elaboration of LCC in this paper, the set-up has been changed. For the environmental (which was actually the starting point), this works fine. The economic part of the formula above would look like

$$\mathbf{g}_\mathrm{econ} = \mathbf{B}_\mathrm{econ}\mathbf{A}_p{}^{-1}\mathbf{f} \tag{27}$$

which is quite different from

$$l = \sum_j -\left[\left(\mathrm{diag}(\boldsymbol{\alpha})\mathbf{A}_p\mathrm{diag}\left(\mathbf{A}_p{}^{-1}\mathbf{f}\right)\right)^\mathrm{T}\mathbf{1} + \left(\mathrm{diag}(\boldsymbol{\beta})\mathbf{B}_p\mathrm{diag}\left(\mathbf{A}_p{}^{-1}\mathbf{f}\right)\right)^\mathrm{T}\mathbf{1}\right],$$
$$\text{where } j \text{ runs over all cases such that } a_{ij} > 0, \text{where } i \text{ is such that } f_i \neq 0 \tag{28}$$

So, is the earlier form incorrect? The answer, surprisingly, is yes and no. Life cycle costs differ from life cycle emissions in not being attached to processes, but to the products (and occasionally to the elementary flows) that are attached to the processes. The emissions come from car driving or steel production, but the costs are made for cars, fuel, and steel. So, the nature of the problem of LCC is different from that of LCA, and it is natural that the operational formulae are different. But economics is more than costs. There are other economic aspects which are of interest

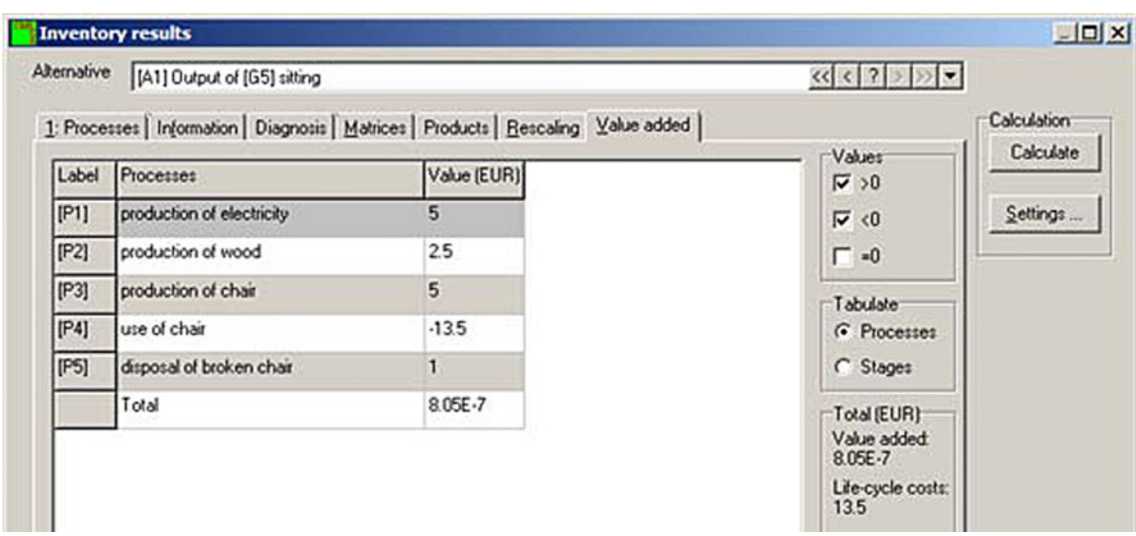

Fig. 3 Screenshot from CMLCA, showing the value added per process, the total value added, and the life cycle costs for the chair example. The total value added is very close to, but not precisely, zero due to small round-off errors

in a sustainability analysis than costs, benefits, or value added. Examples are return on investment, job creation, growth, solvability, the trade balance, and inflation. Narrowing the economic analysis to LCC is like narrowing the environmental analysis to waste. It is conceivable that some of these other economic variables can be addressed as satellite information to process data, so using $\mathbf{B}_{\mathrm{econ}}$. In that sense, the reformulation of LCC in a different form does not mean the end of the original form, but should be understood as an additional form.

The life cycle-wide analysis of social aspects, which are supposed to be addressed through SLCA, has not received specific attention in this paper. Its computational form has been left undiscussed as far as we are aware; it is at least not mentioned by Dreyer et al. (2006) or UNEP/SETAC (2009). We conjecture, however, that it has, like environmental aspects, primarily a process-related character. This would make the use of a social satellite matrix $\mathbf{B}_{\mathrm{soc}}$ appropriate.

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