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Coupling attributional *and* consequential life cycle assessment: A matter of social responsibility

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ABSTRACT

A long-running debate within the life cycle assessment literature concerns the appropriate uses for attributional and consequential forms of life cycle assessment. A recently published contribution to this debate suggests that social responsibility necessarily requires a consequential perspective, and that taking an attributional perspective is optional, but not necessary. The present paper critiques this suggestion by exploring two limitations with only taking a consequential perspective. First, consequential assessments are not additive, in the sense that when added they do not approximate to total aggregate environmental burdens. Second, consequential assessments are not suitable for creating an initial scope of responsibility, as the number of possible decisions available to an agent may be intractably large, and the notion of 'role' responsibility is not defined by specific decisions and consequences. This second limitation is derived from a previously identified parallel between attributional and consequential methods and the normative ethical theories of deontology and consequentialism. Based on the exploration of the two limitations, a coupled accounting solution is proposed which uses both consequential and attributional approaches for different but complementary purposes. The paper concludes by suggesting that although the debate on attributional versus consequential methods has occurred largely within the field of life cycle assessment, the proposed coupled accounting solution has broader applicability to other areas of social and environmental accounting.

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1. Introduction

This article is written largely in response to Weidema et al.'s (2018) paper titled 'Attributional or consequential life cycle assessment: a matter of social responsibility', which is a recent addition to a long-running debate within the life cycle assessment (LCA) literature on the appropriate use of 'attributional' and 'consequential' forms of LCA. This debate traces back to 1993, with Weidema's original observation that none of the published guidance for LCA at the time 'adequately reflects the importance that market aspects and the economic disciplines may have in life cycle inventory methodology' (Weidema, 1993, p. 161). Weidema suggested that life cycle inventories should reflect 'to the largest extent possible, the actual consequences of implementing the results of the investigation' (Weidema, 1993, p. 166). This emphasis on quantifying the consequences of a decision is the essence of the 'consequential' approach, and can be contrasted with quantifying the environmental burdens associated with the processes directly used or physically connected with the product studied, which is the essence of the 'attributional' approach (Brander and Ascui, 2016).

A more formal definition of the attributional-consequential distinction is supplied by UNEP/SETAC, which define attributional LCA as a 'modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule' (2011, p. 132). In contrast, consequential LCA is defined as a 'modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit' (2011, p. 133).

The labels 'attributional' and 'consequential' themselves did not emerge until an international LCA workshop in 2001 (Curran et al., 2005; Ekvall and Weidema, 2004), and by 2008 Finnveden was able

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to state that there 'is today a general agreement within the life cycle assessment (LCA) community that there are two types of LCA … These are often called attributional and consequential LCA' (2008, p. 365). Although there may be broad agreement on the *existence* of the distinction, there remains considerable disagreement on the appropriate uses of the two different approaches. At one extreme, Wenzel (1998) suggests that the only purpose of an LCA is to inform decision-making, and therefore the only appropriate method is a consequential one, as it is only this approach that aims to quantify the total consequences of decisions. In contrast, Tillman (2000) argues that although decision-making is central to life cycle assessment there is still a role for attributional LCA, such as the 'identification of improvement possibilities' (2000, p. 120). In addition to suggesting that there are appropriate uses for attributional LCA, the arguments in Tillman (2000) also mingle concerns about the practical feasibility of consequential LCA, e.g. the difficulty in identifying the systems that change as a result of a decision.

A more recent contribution to this debate is Plevin et al.'s article 'Using attributional life cycle assessment to estimate climate-change mitigation benefits misleads policy markers' (Plevin et al., 2014a), which prompted numerous replies (Brandão et al., 2014; Dale and Kim, 2014; Hertwich, 2014; Suh and Yang, 2014), and counter-replies (Plevin et al., 2014b, 2014c). The key argument in Plevin et al. (2014a) is largely the same as that in Wenzel (1998), i.e. attributional LCA does not aim to quantify the total change caused by the decision in question, and that this information is essential for rational decision-making. However, contrary to Wenzel, Plevin et al. do allow that attributional LCA may have some appropriate uses such as 'normative analyses (e.g., when allocating responsibility for environmental harm)' (2014a, p. 79).

The arguments in response to Plevin et al. (2014a) are also similar to those in Tillman (2000), i.e. methods for quantifying system-wide change, e.g. economic models, may not accurately predict how systems will change (Dale and Kim, 2014; Suh and Yang, 2014); and there are still appropriate uses for attributional LCA, such as product labelling or as a metric for regulatory compliance (Brandão et al., 2014). In response, Plevin et al. accept that methods for quantifying system-wide change may be uncertain, but argue that this uncertainty represents our state of knowledge of the consequences of the decision in question, which is decision-relevant information in its own right, i.e. options that we know have positive outcomes under all plausible scenarios should be preferred to those that do not (Plevin et al., 2014a). Plevin et al. do not respond specifically on the issue of what the remaining appropriate uses of attributional LCA are, but do emphasize that 'we do not argue against all uses of ALCA [attributional life cycle assessment]' (2014c, p. 1560).

One of the most recent contributions in this long lineage is Weidema et al. (2018), which takes the debate in a new direction by focusing on the concept of *responsibility*. It argues that the 'literal meaning of responsibility implies a focus on consequences that can be meaningfully acted upon and changed' (Weidema et al., 2018, p. 312) and that a 'consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle and *may* additionally take responsibility for consequences of other activities in the value chain or supply chain' (2018, p. 313). In other words, consequential LCA is essential, while attributional LCA is optional.

The present paper seeks to advance the debate by critiquing this contention (i.e. that consequential LCA is essential, while attributional LCA is optional) by exploring two limitations with the consequential perspective. Moreover, these limitations can be overcome by using an attributional approach, and therefore both approaches are necessary for managing social or environmental responsibility. The paper proceeds as follows: sections 2 and 3 analyse the identified first and second limitations respectively; section 4 proposes a coupled accounting solution, which uses both consequential and attributional approaches for different but complementary purposes; section 5 discusses strengths and potential objections to the proposed approach, similarities and differences with other approaches previously suggested in the literature, and the main implications of the paper for theory and practice; the final section concludes by highlighting the applicability of the findings to other forms of social and environmental accounting.

2. First limitation: additivity

A first limitation with only using a consequential method is the lack of additivity, i.e. the results from consequential assessments cannot be summed to approximate total aggregate environmental impacts. Tillman (2000) offers two distinct interpretations of additivity, and it is therefore worth analysing and clarifying which is genuinely a limitation for consequential LCA. One interpretation is that an 'important characteristic of an accounting LCA [i.e. attributional LCA] is that of additivity, so that, for example, a LCA of a waste water system can easily be added to one for, say, a detergent.' (2000, p. 116). While a second interpretation of additivity is that 'LCA results of all the products in the world should add up to the total environmental impact in the world' (2000, pp. 116–117). These two interpretations of additivity are not necessarily equivalent, and it appears that consequential LCA may be additive in the first sense, but not the second.

Considering the first sense of additivity, a consequential LCA for a waste water system (i.e. for an increase in demand for waste water treatment) could be combined with a consequential LCA for detergent (i.e. an increase in demand for detergent) to provide an overall assessment of the change caused by the decision to 'produce more waste water and use more detergent'. There may be a problem if the two assessments are undertaken separately but there are interaction effects between the two decisions (e.g. if the detergent assessment itself assumes a specific level of demand for waste water treatment which does not hold true if the separate decision to increase demand for waste water treatment has been taken). However, such cases may be unlikely, or if they do occur it would be possible to undertake the consequential LCAs sequentially and then add them together (which would capture any interaction effects). It may be true that it is easier to add separate attributional LCAs together, as interaction effects do not arise when modelling or allocating environmental burdens within a static system, but this form of additivity does not appear to be an insurmountable or *in principle* limitation for consequential LCA.

In contrast, the second interpretation of additivity (i.e. 'LCA results of all the products in the world should add up to the total environmental impact in the world') does appear to be something that consequential LCA cannot provide. This is because consequential LCA is concerned with marginal systems, i.e. the systems that change as a result of the decision studied (Ekvall and Weidema, 2004), rather than average systems or the product systems that are physically consumed. This is illustrated in Fig. 1 which provides a schematic 'order of dispatch', i.e. the order in which product systems will enter into production to meet demand within a market, and the greenhouse gas intensity of the different product systems. For a consequential LCA, if the baseline level of aggregate demand in the market is $Demand_1$ then any subsequent decision that increases demand (e.g. to $Demand_2$) will cause an increase in the production of Product D, i.e. Product D is the marginal system. Even if a consumer decides to purchase and consume Product A (perhaps because Product A has the lowest emissions intensity) the actual change in emissions caused by the decision is the amount of emissions associated with the marginal system (Product D). This is

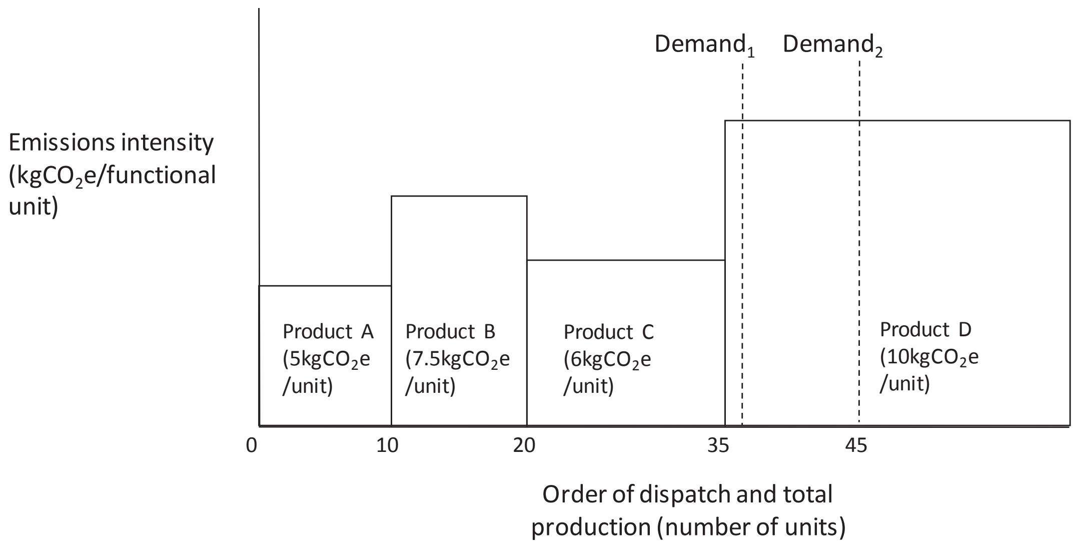

Fig. 1. Order of dispatch for product systems within a market.

because the production of Product A is already at its maximum capacity. Choosing to consume Product A only displaces existing consumers of Product A who then consume alternative products in the market, and ultimately total demand can only be met by increasing the production of the marginal system (Product D).

Relating this to the issue of additivity, consequential LCA only studies the marginal system, and therefore only provides information on the change in emissions caused by an additional unit of Product D (10 $kgCO_2e$/unit). If this is multiplied by the total number of products produced (10 $kgCO_2e$/unit * 45 units) the result is 450 $kgCO_2e$, whereas total emissions to the atmosphere are 315 $kgCO_2e$ (calculated as per Equation 1, or the area of each box in the order of dispatch up to the level of total demand), i.e. consequential LCA cannot be used to estimate total aggregate environmental impact.

Equation 1. Calculation of total aggregate emissions

$$\text{Total Emissions} = \sum_m TP_{m,y} * EF_{m,y}$$

TP_m = Total production from product system m
EF_m = Emissions factor for product system m
$_m$ = Each product system supplying the market
$_y$ = The relevant time period for the analysis, e.g. a year

Furthermore, consequential LCA treats cases of multi-functionality (e.g. the production of co-products) by a method called *substitution*, which involves identifying the product systems that are displaced by the production of co-functions, and crediting the displacement of those product systems to the decision studied, as the avoidance of those systems is a consequence of the decision (Weidema et al., 2009). This entails that consequential assessments do not represent, nor are they intended to represent, actual aggregate emissions to the atmosphere (Brander and Wylie, 2011), and therefore the sum of consequential LCA results for all the products in the world will not 'add up to the total environmental impact in the world' (Tillman, 2000, pp. 116–117).

This limitation is an important one, as arguably sustainability is a system-level attribute rather than a characteristic of individual practices, i.e. it is not possible to achieve sustainability if aggregate levels of consumption or resource use exceed system-level sustainable thresholds (Gray, 2010). Recognition of this underpins initiatives such as 'science-based target setting' (Science Based Targets Initiative, 2015), which aims to ensure that corporate greenhouse gas (GHG) reduction targets are aligned with global carbon budgets. For example, the sectoral-decarbonisation approach (SDA) involves estimating the level of future production and available carbon budgets for different sectors of the economy, and from this emissions-intensity targets are derived for companies within each sector (Science Based Targets Initiative, 2015). In theory, if the targets are met then the GHG emissions from all companies and all sectors should sum to a total that does not exceed the global carbon budget, e.g. between 550 and 1300 $GtCO_2$ between 2011 and 2050 for a 2 °C target increase in global temperature (Krabbe et al., 2015). Similar initiatives have been discussed in relation to life cycle assessment, i.e. for translating planetary boundaries into per capita allowances (Frischknecht et al., 2016). The important point is that consequential LCA cannot be used for such purposes, because the results are not additive to total aggregate impacts, and therefore Weidema et al.'s (2018) contention that only consequential LCA is essential, while attributional LCA is optional, appears problematic.

3. Second limitation: initial scope of responsibility

A second limitation with only using a consequential method is derived from the insight, originally proposed by Ekvall et al. (2005), that attributional and consequential LCA correspond to the normative ethical theories of deontology and consequentialism. Broadly, deontic ethical theories determine the rightness of an action by its conformance to a rule (Van Staveren, 2007), and consequential (or 'telic') theories determine the rightness of an action by its consequences (Scheffler, 1988). The respective emphasis on either rules or consequences parallels the UNEP/SETAC definitions of attributional and consequential LCA given above, with the attributional boundary determined by 'a normative rule' and the consequential boundary determined by the 'consequence of a change'. In addition to identifying a second limitation with only using a consequential approach, the distinction between deontic and consequential ethics is also useful for developing the proposed 'coupled accounting' solution (in Section 4), and it is therefore worth articulating these normative ethical theories in some detail.

One archetypal deontological theory is Kant's categorical imperative, which, under one formulation, states 'Act in such a way that you treat humanity, whether in your own person or in any other person, always at the same time as an end, never merely as a means' (Kant, 2002, sec. 4:429). One of the arguments in support of this rule is that all value is conferred by humanity, and therefore if humans value anything they must value humanity (Korsgaard, 1996). Such a rule creates limits to actions regardless of the consequences of the action, e.g. it is not right to harm an innocent person, even if doing so has consequences that increase total human happiness (as humans should be treated as ends and 'never merely as a means'). Deontic ethics is also often referred to as *duty* ethics, as the ethical rule binds the agent to their duty (Ransome and Sampford, 2010). A key point for the purposes of the present discussion is the primacy of *rules* within the deontic approach, rather than the consequences or outcomes from an action.

In contrast, consequential ethics holds that 'the right act in any given situation is the one that will produce the best overall outcome, as judged from an impersonal standpoint which gives equal weight to the interests of everyone' (Scheffler, 1988, p. 1). An archetypal consequential theory is utilitarianism, which states that the 'greatest happiness for the greatest number is the measure of right and wrong' (Bentham, 1948, p. 3). One of the central features of what Scheffler (1988) describes as 'pure' consequentialism is its impartiality or agent-neutrality, i.e., all interests are given equal weight. According to this view of consequentialism, 'act' consequentialism is a pure form, as individual agents are required to impartially consider the aggregate or system-wide consequences of their actions. In contrast, 'rule' consequentialism, which judges the rightness of an act based on whether it conforms to a rule which achieves the greatest beneficial consequences (Shafer-Landau, 2013), is less 'pure' as it opens to door to agent-relativism, if the rule which generates the greatest beneficial consequences involves some degree of agent-relative self-interest. An example of such a case is Adam Smith's famous justification for individual economic self-interest, which argues that it is 'not from the benevolence of the butcher, the brewer or the baker that we expect our dinner, but from their regard to their own self-interest' (Smith, 1976, pp. 26–27). Following Scheffler (1988), agent-neutrality or impartiality is treated here as a defining feature of pure consequential ethics, and this feature is also clearly present within consequential LCA. For example, Weidema et al. state that the 'socially optimal action for any organisation at any specific point in time is to change the specific activities that provide the currently most cost-efficient environmental improvement, no matter whether these activities show up as important within the specific value chains, supply chains, or product life cycles of the organisation' (2018, p. 311).

Turning now to the second limitation with taking only a consequential approach, it is possible to identify how some of the weaknesses associated with ethical consequentialism may also apply to consequential LCA. One criticism of the agent-neutrality inherent in consequentialism is that it becomes excessively demanding (Scheffler, 1988), if, for example, the universe of possible actions that are available at any point in time has to be considered. The enormity of this requirement becomes apparent when it is recognised that responsibility extends to actions that could have been taken but were not, and also to actions by others that could have been prevented, but were not. As recognised by Weidema et al. 'inaction, i.e. the omission of action in a situation where action could have been taken, has consequences' (Weidema et al., 2018, p. 311), and by Williams ' … if I am ever responsible for anything, then I must be just as much responsible for things that I allow or fail to prevent, as I am for things that I myself, in the more everyday restricted sense, bring about' (Williams, 1995, p. 95).

The problem with consequentialism is that the number of potential decisions or options may be intractably large, and being unaware of all the possible options does not limit responsibility, i.e. if it is within the power of an agent to implement an option then they are responsible for any consequences from not having done so. This problem is different from the arguments concerning the practical feasibility of identifying the consequences of decisions previously raised in the literature (Dale and Kim, 2014; Suh and Yang, 2014; Tillman, 2000), as the issue is not the feasibility of identifying the consequences of a decision relative to a business-as-usual or zero-action baseline, but rather the feasibility of identifying all the decisions or actions that could be taken at any point in time.

A further potential limitation with consequentialism is that there must always be a specific decision in question, the consequences of which constitute the responsibility of the agent. However, there are types of responsibility, notably 'role' responsibility, which are not concerned with specific actions, but instead involve on-going 'care and attention over a protracted period of time' (Hart, 1968, p. 213), e.g. the kind of responsibility parents have for their children. 'Role' responsibility appears to be highly applicable to producers (and consumers too), e.g. producers may feel they have a 'role' as stewards of their supply or value chain, and are therefore responsible for that supply/value chain, independently of any specific decision or action. An example, borrowed from Weidema et al. (2018), is a food company that uses wild fish which have been caught using high-impact bottom trawling. Regardless of any decision or whether the company's purchase of the fish causes the impact (i.e. the production system using the bottom trawling may not be the marginal system, and so would happen anyway), there is an intuitive sense of responsibility, or duty of care.

Consequentialism appears to struggle with establishing an initial scope of responsibility, either because the number of potential decisions available may be intractably large, or because some intuitively held forms of responsibility are prior to and independent of specific decisions or actions. In contrast, a deontic/attributional approach appears able to handle exactly these issues. Such an approach defines the scope of responsibility according to a normative rule, and can therefore create a clearly bounded scope that the agent in question is responsible for, e.g. 'All the impacts associated with the processes used in the physical supply chain'. Similarly, deontic/attributional approaches can define responsibility without reference to a specific decision or action, and can therefore encapsulate 'role' responsibility too. This is not to say that consequentialism is incapable of encapsulating other aspects of responsibility, e.g. 'causal' responsibility (Hart, 1968), but that there are some forms of responsibility that it cannot accommodate, and therefore consequentialism on its own, contrary to Weidema et al. (2018), is not sufficient. This suggests that both attributional and consequential approaches are necessary for adequately defining and managing responsibility.

4. Proposal for a coupled accounting approach

To briefly summarise, the two identified limitations with only using a consequential method are: first, the results from consequential LCA do not sum to approximate total aggregate environmental impacts; and, second, consequential assessments are not suitable for creating an initial scope of responsibility. Ekvall et al. (2005, p. 1232) suggest that the solution to the attributional-consequential debate is to recognise that the two approaches can 'result in complementary information', but provide relatively little detail on how this might be done. This section therefore sets out a proposal for how attributional and consequential methods should be used to complement each other within a coupled accounting approach, drawing on the respective strengths of each perspective.

The first step in the proposed coupled accounting approach is to use an attributional method to define an initial sphere of responsibility for the agent in question, in order to create a sense of ownership for a specific set of impacts. A consequential approach does not appear to be appropriate for this purpose, given the issues discussed above. A further reason for using an attributional approach to define an initial sphere of responsibility is that it is then conceptually possible to sum individual assignments of impacts to estimate total aggregate impacts to the environment, to set reduction targets based on global thresholds or budgets, and to track progress over time. This approach also helps address the point that sustainability is a system-level attribute, and that individual organisational or consumer accounting is largely meaningless if it does not reflect the overall sustainability of the system (Gray, 2010).

The second step in the proposed coupled accounting approach is to use a consequential method to assess the system-wide consequences of any decisions aimed at managing the impacts within the attributional (rule-defined) scope of responsibility, because there is no environmental benefit in reducing impacts within an attributionally defined inventory if total impacts are increased elsewhere (for example, see Searchinger et al.'s (2008) critique of US biofuel policy). This second step also embodies the strong consequentialist intuition that agents should be responsible for the system-wide consequences of their decisions (Lasswell and Kaplan, 1950; Scheffler, 1988). Fig. 2 sets out the structure of the proposed coupled accounting approach.

Returning to the example of the food company, the following is a simple illustration of how the coupled accounting approach would work. In Step 1 the food company would define its scope of responsibility by undertaking an attributional account of its impacts (e.g. based on the physical processes within its value chain). This scope of responsibility would include the high-impact bottom trawling used to catch the fish the company purchases. The company then sets a target for reducing its impacts, e.g. a 50% reduction in the impacts from bottom trawling in 2 years' time (a sophisticated target-setting approach would be to estimate the capacity of the fishing ground to sustain bottom trawling and the company's share of that sustainable 'budget'). In Step 2 the food company considers the option of switching its purchasing to farmed fish, as this would only involve a small increase in cost and would reduce bottom trawling impacts in the attributional account to zero. However (in accordance with Step 2), the company conducts a consequential assessment to estimate the system-wide change in impacts caused by the decision and finds that other food companies would most likely purchase the wild caught fish (as it is lower cost) and there would be no change in total environmental impacts. Instead, and after assessing the option with a consequential method, the food company chooses to engage with the trawler company to implement low-impact fishing gear.

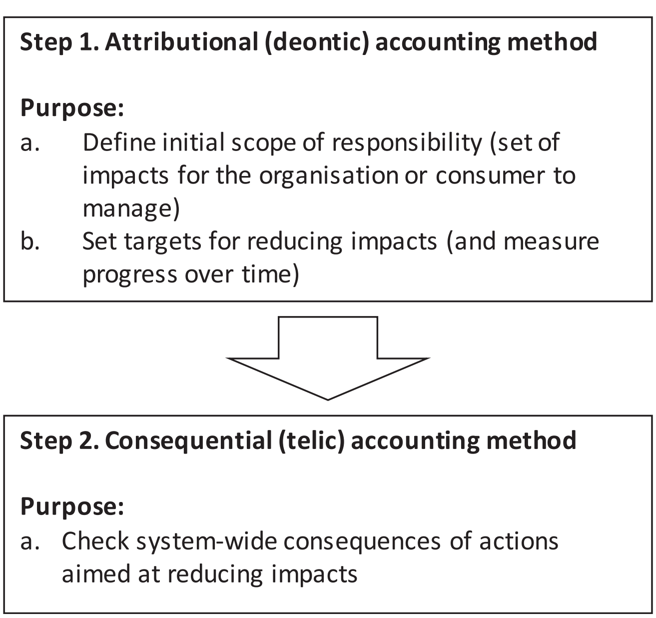

Fig. 2. Structure of coupled accounting approach.

It is worth emphasizing that this proposed approach is not suggesting that attributional and consequential elements should be blended within a single analysis, as is unintentionally done in ISO 14044 (Brander and Wylie, 2011; Weidema, 2014), but rather that separate attributional and consequential methods can be used in combination, with each applied to their appropriate purpose. It is also worth emphasizing that although the proposed approach offers a different view from that in Weidema et al. (2018), by arguing for the necessity of attributional methods, it is important to underscore that, given the widespread misuse of attributional methods for decisions about actions, only consequential methods are conceptually appropriate for this purpose.

5. Discussion

5.1. Strengths and potential objections to the proposed approach

One indication of the conceptual coherence of the proposed coupled accounting approach is that it fulfils the two criteria for assigning responsibility in Rodrigues et al. (2006) that Weidema et al. (2018) identify as being mutually exclusive if they are required of a single approach. The two conflicting criteria in Rodrigues et al. (2006) are:

> 'that environmental responsibility should verify a normalization condition, such that the sum of the environmental responsibility of all agents should equal total environmental pressure [i.e. additivity]' (2006, p. 259);

and

> 'that the indicator should not display wrong signals, only allowing for a decrease in environmental responsibility of an agent if there was a decrease in overall direct environmental pressure.' (2006, p. 259).

Weidema et al. correctly point out that these 'are mutually exclusive conditions, since the first refers to - and can only be fulfilled in - a steady-state analysis of environmental pressure and the second condition refers to a change in environmental pressure and can only be fulfilled in an analysis of changes, which is not possible in the analysed steady-state system.' (2018, p. 306). One strength of the coupled accounting approach is that it fulfils both of these criteria, but without the incoherence of attempting to fulfil them in a single method, i.e. an attributional method is necessary for the first criterion, and a consequential method is necessary for the second.

A potential objection to the necessity of the first criterion (i.e. the requirement for additivity), and by implication a partial rejection of the necessity of using an attributional method, is that 'several actors can assume full responsibility, so that responsibility is not a conserved quantity like mass' (Weidema et al., 2018, p. 308). However, there is no incompatibility between multiple actors taking responsibility for the same impact and ensuring additivity. A good example of an environmental accounting approach that achieves both is the use of 'scopes' within corporate-level GHG accounting, in which ‘scope 1’ emissions are from facilities controlled by the reporting entity, ‘scope 2’ emissions are from the generation of purchased energy (e.g. electricity), and ‘scope 3’ are from any other source within the reporting entity's value chain (WBCSD/WRI, 2004). More than one actor can take responsibility for the same emissions, e.g. scope 1 emissions for the electricity generator will be scope 2 emissions for the consumer of the electricity, but additivity is still achieved (i.e. summing only scope 1 emissions will approximate to total emissions).

A different potential objection to the proposed coupled accounting approach is a problem associated with using an attributional method to create the initial scope of responsibility, i.e. that it ‘is inconsistent for a socially responsible decision maker to exclude consequences of own actions (i.e. the consequential life cycle) while including consequences from actions of others in the value chain or supply chain’ (Weidema et al., 2018, p. 306). In other words, because the attributional boundary is defined by normative rules rather than consequences, the initial sphere of responsibility may include impacts that cannot actually be influenced by the agent, and/or may exclude impacts that can be. One partial solution to this problem is to screen the attributionally defined inventory to exclude impacts that cannot be influenced by the agent in question. An example of such an approach within corporate-level GHG accounting is the GHG Protocol *Corporate Value Chain (Scope 3) Accounting and Reporting Standard,* which states that companies ‘should prioritize activities in the value chain where the reporting company has the potential to influence GHG reductions’ (WBCSD/WRI, 2011, p. 61). It is also important to note that using a consequential method does not appear to be a viable alternative, given the potentially intractable number of possible actions that would have to be considered, and the inability to capture ‘role’ responsibility.

A further related problem is how to choose between the following two options: 1. an action which reduces the impacts within the attributionally defined inventory, and which does not increase impacts elsewhere in the system; or 2. an action which does not reduce impacts within the attributionally defined inventory, but which reduces total system-wide impacts more than the first option (T. Ekvall 2017, pers. comm., 28 November). Something akin to this issue arises with corporate-level GHG accounting and what can be termed ‘product-enabled reductions’, which are reductions enabled by a company's products (e.g. low-temperature detergents, fuel-saving tyres, or teleconferencing services etc.), but the reductions are not captured within the attributional boundary of scopes 1, 2, or 3 (World Resources Institute, 2013). One existing option for reporting reductions in impacts that occur outside the attributional inventory is a ‘gross-net’ approach, in which the gross figure is the attributional inventory and the net figure is an adjusted total reflecting the change in emissions occurring outside the attributional inventory (for an example of this approach see Defra's guidance on environmental reporting (Defra, 2013)). Such an approach would encourage, but not necessarily require, actions that maximise system-wide benefits.

5.2. Differentiation from other reconciliations

There are a number of papers in the literature that also discuss the relationship between attributional and consequential methods, and propose possible reconciliations, and it is important to highlight how the present paper supports or differs from these positions. Firstly, Sandén and Karlström (2007) suggest that a ‘constructive approach is to conduct attributional LCAs based on relevant future states or scenarios of consecutive states to guide the direction of actions. To assess decisions directly, a consequential perspective is needed’ (2007, p. 1479). This approach is similar to that of the present paper, in that it envisages using an attributional method as a starting point, and that a consequential method is required to assess specific decisions. An additional, rather than contradictory, point made in the present paper is that an attributional approach is *necessary* for establishing an initial scope of responsibility, prior to consideration of future scenarios or improvement decisions. A different area of alignment is Sandén and Karlström's intimation that attributional and consequential methods should not be mixed in a single analysis, ‘ … attributional and consequential perspectives are mixed. This opens up for somewhat misleading interpretations … ’ (2007, p. 1479).

A more recent paper, that builds on aspects of Sandén and Karlström (2007), is Arvidsson et al. (2016), which suggests that a ‘first order consequential study of the substitution [of graphene for indium tin oxide in electrodes] is methodologically identical to a comparison between two prospective attributional studies’ (2016, p. 290). This appears at odds with the contention in the present paper that consequential and attributional methods are fundamentally distinct approaches that are appropriate for different (but complementary) purposes. One caveat to Arvidsson et al.'s statement is that the equivalence holds if only ‘first order’ effects are considered, i.e. the physical flows used in the life cycle of the product(s) in question (Sandén and Karlström, 2007). With this caveat, the equivalence appears to be achieved by restricting the ‘consequential’ assessment to the normative boundary of an attributional approach, whereas a full consequential study would include second order (i.e. market-mediated) and, ideally, third order (e.g. positive feedback from learning) effects.

Arguably a more thorough-going equivalence between attributional and consequential approaches would be achieved, in principle, by describing two complete macro-level attributional scenarios (i.e. all the physical flows within each system), and then comparing those scenarios to provide an estimate of the system-wide change caused by switching from one scenario to the other. The crucial requirement for this to work is that the macro-level scenarios should include all the processes that change, i.e. there are no effects that occur outside the boundary of the macro-level scenarios. Notably, both Sandén and Karlström (2007) and Arvidsson et al. (2016) focus on large or technology-scale (rather than product-level) changes, for which macro-level scenario comparisons are highly applicable, and could (given the requirement above) capture total system-wide change. However, once the preferred macro-level scenario has been identified it is still necessary to choose specific actions and policies for achieving that scenario, which is the point made by a Sandén and Karlström (2007) quoted above (i.e. to ‘assess decisions directly, a consequential perspective is needed’ (2007, p. 1479)). A potential addition to the proposed coupled accounting approach could therefore be an optional ‘Step 1c: If considering a technology-scale change, macro-level attributional scenarios can be useful for estimating the difference in total impacts between the scenarios’.

A further recent study which has some similarities, but also some important differences, with the present paper is Yang (2016), which offers ‘a two-step approach to CLCA [consequential LCA] based on the attributional framework’ (2016, p. 277). The first step involves conducting an attributional LCA to ‘evaluate the status quo of the system under study and identify hotspots on which we can focus subsequently’ (2016, p. 277) and the second step involves modifying attributional LCA to make it more suitable for assessing change, e.g. using marginal instead of average coefficients. The first step is broadly similar to that proposed in the present paper (though, again, the additional contribution from the arguments presented above is that they establish the *necessity* of using an attributional method for setting an initial scope of responsibility). However, the second step in Yang (2016) appears to be somewhat problematic as it involves adopting some (but not all) of the features of consequential LCA, with the outcome that the results would not represent an estimation of system-wide change, and nor do they maintain the key characteristics of an attributional approach, e.g. additivity. As noted by Sandén and Karlström (2007), mixing attributional and consequential methods results in 'misleading interpretations', and rather than attempting to make attributional methods something that they are not (i.e. facsimiles of consequential methods), the present paper suggests that it is better to recognise the purposes for which they are uniquely useful.

5.3. Implications for theory and practice

The principle theoretical contribution of this paper is towards conceptualising the distinction between attributional and consequential methods, with the development of such a categorical framework (Denzin, 1970), and the formulation of normative rules (Suddaby, 2014), constituting forms of theory development. Previous studies that have sought to establish the distinction between attributional and consequential methods tend to present a negative case, i.e. by showing what attributional methods should not be used for (e.g. Plevin et al., 2014a), whereas the present paper provides a positive case for the distinction, by showing the uses for which attributional methods are uniquely appropriate.

The major implication for practice is that both attributional and consequential methods are necessary for managing social and environmental responsibility (with each method appropriate for different purposes), and that a single method is not sufficient. This idea, and the proposed way in which each method should be used, is currently not well-established or recognised within the practitioner community. For example, the recently published ISO 14067 standard for the carbon footprinting of products (ISO, 2018) does not state whether it provides an attributional or consequential method, or whether it is appropriate for determining an initial sphere of responsibility (which may be summed to approximate total global emissions), or whether it is appropriate for quantifying the system-wide change in emissions caused by a change in demand for the product. When this standard is due for review in 2023, a recommendation to ISO is to clarify which type of method the standard is intended to provide, and its appropriate use. The benefit for the field of practice would be to ensure that the correct method is used for its appropriate purpose.

6. Conclusions

The purpose of this paper is to contribute to the ongoing debate on the appropriate use of attributional and consequential LCA, and specifically to critique the recent suggestion that a consequential approach is essential, while an attributional approach is optional. The analysis presented here shows that there are limitations to consequentialism, and that these limitations can only be addressed by using an attributional approach. A coupled accounting approach is proposed which explicitly recognises and utilises the respective merits of the two methods.

The argument presented here on attributional and consequential approaches has been framed within the context of life cycle assessment, though reference is also made to lessons that can be drawn from corporate-level GHG accounting. It is worth broadening the horizons of the current debate to recognise that the attributional-consequential distinction is highly applicable as a general framework for categorising many other forms of social and environmental accounting. That is, inventories of social and environmental impacts, whether at the product, corporate, city, or national level can be categorised as 'attributional' in nature, while any assessment of system-wide change can be categorised as 'consequential'. Recognising attributional and consequential methods as categorical 'families' creates the possibility of sharing methodological lessons or innovations between methods of the same type (Brander, 2016). It also entails that the proposed coupled accounting approach is equally applicable to corporate, city, and national-level accounting, as well as product LCA. One potentially significant policy implication is from applying this to the emerging GHG accounting practices for Nationally Determined Contributions (NDCs) under the Paris Agreement (UNFCCC, 2015). At present most countries' NDCs are specified with reference to their national production-based GHG inventories (i.e. normatively defined attributional inventories), but without the requirement that any actions aimed at achieving emission reductions should be assessed using consequential methods. A strong recommendation, based on the coupled accounting approach proposed above, is that governments should use consequential as well as attributional methods.

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Six areas of methodological debate on attributional life cycle assessment

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Abstract. There is a general agreement in the LCA community that there are two types of LCAs: attributional and consequential. There have been numerous discussions about the pros and cons of the two approaches and on differences in methodology, in particular about methods that can be used in consequential LCA. There are, however, methodological aspects of attributional LCA and how it can be used that need further attention. This article discusses six areas of debate and potential misunderstandings concerning attributional LCA. These are: 1) LCA results of all the products in the world should add up to the total environmental impact of the world, sometimes referred to as the 100 % rule. 2) Attributional LCA is less relevant than consequential LCA. 3) System expansion, and/or substitution, cannot be used in attributional LCA. 4) Attributional LCA leads to more truncation errors than consequential LCA does. 5) There is a clear connection between the goal and questions of an LCA and the choice of attributional or consequential LCA. 6) There is a clear boundary between attributional and consequential LCA. In the article, these statements are discussed, and it is argued that they are either misunderstandings or sometimes incorrect.

1 Introduction

There is a general agreement in the Life Cycle Assessment (LCA) community that there are two types of LCAs: attributional (ALCA) and consequential (CLCA) [1], although the terminology has been slightly different throughout the years. There are different definitions of ALCA and CLCA in the literature, but in most cases there is a general agreement on the

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main principles. Here, ALCA is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems [2], and CLCA is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions [3].

There have been numerous discussions about the pros and cons of the two approaches and on differences in methodology [4, 5]. The focus has often been on CLCA, e.g. [6-8]. In the literature, there have been statements about ALCA (e.g. [4, 9]) that deserve further attention. The aim of this paper is to discuss six areas of debate about ALCA and to contribute to the methodological discussion on ALCA and LCA in general.

2 Six areas for discussion

2.1 The 100 % rule

It is often suggested that if one would add up all the ALCAs of all the products in the world, it should add up to the total environmental impact in the world [9]. This is sometimes referred to as "the 100 % rule".

The 100 % rule interpreted in this way, however, does not seem to be correct, which can be illustrated by a simple example. One could do an ALCA of diesel fuel. One would then include the production of the fuel and the use of it. One could also do an ALCA of a truck. One would then again include the production and use of the fuel. One could also do an ALCA of a waste management system and if that system would include a diesel truck, one would again include the production and use of the fuel. Additionally, different allocation methods applied by different ALCA practitioners to the multifunctional refinery process co-producing diesel among 10-15 other refinery products, could result in another breach of this 100% rule.

This first simple example shows that the production and use of the diesel fuel would be included in many ALCAs, and thus the sum of all ALCAs would be much larger than the total environmental impact of the world. The second example can lead to over- or underestimates. The 100 % rule is thus not correct if defined as above. It could, in principle, be correct if there were rules on which products could be included, e.g., only certain types of consumer end products, the system boundaries for the assessment, which allocation methods to apply, and how to deal with long-living products. One example of this could be when the 100 % rule is explicitly only for "final products" where "final products" are defined "as a product that is directly consumed by humans and not used in the life cycle of another product" [4]. Interpreted in this way, the 100 % rule is only relevant for "final products" and not necessarily for other types of products. In practice, it may turn out to be difficult to identify final products based on this definition. An LCA practitioner is free to choose the product under study, and most LCAs are not of final products, making the 100 % rule less relevant. Another example where the 100 % rule could be valid is for a company that may want to define a 100 % rule to account for ALCAs of all products that they sell and make sure that the sum amounts to the total impact of the company. In this case, they have defined which products to study, and they can define rules for system boundaries, allocation, etc. so that the 100 % rule can be valid. This is, however, a special case and not a general aspect of ALCA.

2.2 The relevance of ALCA

It has been suggested that the sole reason to perform LCA is to use it to support decision making, and that the key requirement for that purpose is that LCA should reflect the environmental consequences caused by the decision, and therefore only CLCA would be relevant, c.f. [10].

This is, however, a simplified view. There are many other uses for LCA apart from using it as a basis for a decision, e.g. learning [11] and finding out which life cycle phase or which product in a consumption pattern contributes most [12]. In these cases, ALCA may be very relevant. Also, much of the arguments against ALCA are based on the assumption that consequentialism is the only appropriate basis for decision-making [13]. However, there may be other relevant bases, e.g. to choose the product with the lowest environmental impacts, which also can be valid [14] and which would instead suggest an ALCA. Therefore, ALCA can also be an appropriate method in many applications.

2.3 System expansion and substitution

A classic problem in LCA is how to handle situations where a process includes several functions, e.g. [1, 2, 4, 9, 15]. An example may be a waste treatment process, e.g. waste incineration, which as one function treats the waste but as a second function also produces electricity. This process can be difficult to compare with a process that only takes care of the waste, but does not produce electricity. This situation can be solved in different ways, and here three approaches will be discussed, c.f. [4]. The first is to allocate (also called partition) between the two functions taking care of the waste and producing electricity. This can be done employing several different methods for the allocation, e.g. [1, 2]. When that has been done, the waste treatment part of the first system can be compared with the second system. A second approach is to conduct a so-called “system expansion” and include two functions: taking care of waste and producing electricity. The process that only takes care of the waste must then be complemented with a process that only produces electricity. The two alternatives are now possible to compare since they fulfil the same functions. A third approach is to credit or subtract the process that only produces electricity from the process that takes care of both the waste and produces electricity. The two systems can now be compared since the first system only has one functional unit, i.e. taking care of the waste. This third option is sometimes called “substitution” or the “avoided burdens” approach [4], but often, the term “system expansion” is used in a broad sense to cover also substitution, e.g. [16, 17], which, however, is debatable [18].

The ISO standard for LCA [19] includes a hierarchy for handling multifunctional processes where system expansion (often interpreted in a broad sense [17, 20]) is presented as a first step to avoid allocation, i.e. before performing different forms of allocation. The ISO standard does not differentiate between ALCA and CLCA. System expansion (in the broad sense) is often used and has, for example, become the dominating approach in waste management LCAs [21]. For comparative studies, system expansion in the narrow sense and substitution gives qualitatively the same result, although the specific numbers will be different [22]. However, this is not the case for non-comparative stand-alone LCAs [23].

It is often claimed that system expansion in a broad sense should not be used for ALCA, e.g. [9]. Others would suggest that system expansion, in a narrow sense, can be used for ALCA, but substitution should not be used [4].

It seems strange not to accept system expansion in the narrow sense in ALCA since a functional unit can cover several functions, c.f. [4]. Unless restrictions on the possibilities of choosing the functional unit are introduced, it seems clear that system expansion in the narrow sense should be compatible with ALCA.

One suggested reason for not accepting substitution in ALCA is that it would be in conflict with the 100 % rule [4]. However, as suggested in section 2.1, if the 100 % rule is not valid, then this argument would not be relevant. Since the results for comparative studies are qualitatively the same for system expansion in a narrow sense and substitution, and they include the same components, it also seems reasonable to also accept substitution in ALCA when substitution in the narrow sense is accepted. System expansion in the narrow sense may be preferable over substitution for increased transparency [22], especially for stand-alone studies. It can also be noted that the data used when substitution is used would probably be different in ALCA and CLCA [2, 24], with e.g. average data being used for substitutions in ALCAs and marginal data for substitutions in CLCAs. It should also be noted that in the special cases where the 100 % rule could be valid, as discussed in section 2.1, then substitution could be in conflict with this rule.

2.4 Truncation and aggregation errors

When conducting an LCA, it is difficult to include all relevant processes, which can lead to truncation errors [25]. It can be argued that this will be a greater problem for ALCA than CLCA since the latter, in theory, only needs to include processes that are changing due to the decision [6].

It may, however, be the other way around, that this is a larger problem for CLCA. One reason for this is related to difficulties in predicting the future and the consequences of decisions [26]. Furthermore, data for consequential assessments are not widely available, and the assessment may require a more comprehensive understanding of the changes to other systems. Another reason is that Input-Output Analysis (IOA) and Environmentally Extended Input-Output Analysis (EEIOA) can be used for providing missing data [27]. This is useful since IOA/EEIOA covers the whole economy of a country and thus does not include the same type of truncation errors as process-based LCA. IOA is by its nature, an accounting tool and therefore more easily applicable in an ALCA context. However, the use of IOA may lead to more aggregation errors since the product groups are broadly defined [28]. Overall, there are different types of calculation errors for both ALCA and CLCA and it is not possible to make a general conclusion on which type may have the largest errors.

2.5 The connection between goal and scope and choice of methodology

It is sometimes expected that there should be a clear connection between the goal of an LCA on the one hand and the choice between ALCA and CLCA on the other. There have been projects where the aim has been to establish this connection. In practice, however, this is not easy to reach consensus on, e.g. [29]. For example, whereas some would argue that LCAs done for different types of environmental labelling are typical ALCA applications, others would argue that since environmental labelling is intended to support decisions, and decision-support should be based on consequences of the decisions, this is instead a typical CLCA application. However, the type of decision is rarely addressed, e.g. consumer, company-level, or political. It, therefore, seems more difficult than expected to establish clear connections between goals of LCAs and the choice between ALCA and CLCA and reach a consensus around these.

2.6 The boundary between ALCA and CLCA

The literature typically suggests two distinctive methodological differences between ALCA and CLCA: 1) The way to handle multifunctional processes where it is sometimes suggested that ALCA should use allocation and CLCA system expansion in a broad sense, e.g. [9], and 2) The choice of data where it is typically suggested that ALCA should use average data and CLCA should use marginal data, e.g. [2, 9]. Sometimes it has also been suggested that the scale of the decision should influence the choice, c.f. [4, 29].

As discussed above, it seems reasonable that system expansion and substitution can be compatible with ALCA. It has also been suggested that there is no strict delimitation of average data for ALCA and marginal for CLCA [4]. Additionaly, based on common definitions of ALCA and CLCA, the scale of the decisions should not influence the choice [4]. Furthermore, in practice, there are a number of studies in between ALCA and CLCA [30, 31] and beyond both [32]. As such, the distinctions are sometimes ambiguous and have not added to the clarity on the differences between the approaches. Therefore, the difference between ALCA and CLCA may be less clear than what is often assumed, and ALCA and CLCA may not be the only two modes of LCA. On the other hand, the possibility of a difference in computational set-up has hardly been addressed [33].

3 Final reflections

When the difference between ALCA and CLCA became established in the LCA community at the end of the 1990s, it was for many somewhat of a revelation. It seemed that many of the classical methodological discussions related to the choice of data, system boundaries, and allocation could now be resolved by making a connection to the goal of the study. In practice, this has turned out to be more difficult than what was expected and discussions are still ongoing. This paper suggests that the differences between ALCA and CLCA are not so clear-cut, neither in theory nor in practice, and that many of the claims of the two methods may need to be revisited. This may require that also some of the basic definitions and methodological aspects of LCA in general and different versions of it be revisited.

More specifically, this paper suggests that the 100 % rule, stating that if one would do an ALCA of all the products in the world, the sum would be the total environmental impacts of the world, is not correct. Also, this paper suggests that system expansion and substitution can be used in both CLCA and ALCA studies.

The discussions here have focused on Environmental LCA, but are equally valid for broader Life Cycle Sustainability Assessments. It is also of relevance for other sustainability assessment tools where the distinction between attributional and consequential assessments may be relevant. We mention in particular IOA and EEIOA, which have been constructed – implicitly – as attributional models and are – tacitly – used to answer consequential questions [34].

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UNCERTAINTIES IN LCA

On the number of Monte Carlo runs in comparative probabilistic LCA

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Abstract

Introduction The Monte Carlo technique is widely used and recommended for including uncertainties LCA. Typically, 1000 or 10,000 runs are done, but a clear argument for that number is not available, and with the growing size of LCA databases, an excessively high number of runs may be a time-consuming thing. We therefore investigate if a large number of runs are useful, or if it might be unnecessary or even harmful.

Probability theory We review the standard theory or probability distributions for describing stochastic variables, including the combination of different stochastic variables into a calculation. We also review the standard theory of inferential statistics for estimating a probability distribution, given a sample of values. For estimating the distribution of a function of probability distributions, two major techniques are available, analytical, applying probability theory and numerical, using Monte Carlo simulation. Because the analytical technique is often unavailable, the obvious way-out is Monte Carlo. However, we demonstrate and illustrate that it leads to overly precise conclusions on the values of estimated parameters, and to incorrect hypothesis tests.

Numerical illustration We demonstrate the effect for two simple cases: one system in a stand-alone analysis and a comparative analysis of two alternative systems. Both cases illustrate that statistical hypotheses that should not be rejected in fact are rejected in a highly convincing way, thus pointing out a fundamental flaw.

Discussion and conclusions Apart form the obvious recommendation to use larger samples for estimating input distributions, we suggest to restrict the number of Monte Carlo runs to a number not greater than the sample sizes used for the input parameters. As a final note, when the input parameters are not estimated using samples, but through a procedure, such as the popular pedigree approach, the Monte Carlo approach should not be used at all.

Keywords Accuracy · Life cycle assessment · Monte Carlo · Precision · Uncertainty

1 Introduction

Uncertainty in LCA is pervasive, and it is widely acknowledged that uncertainty analyses should be carried out in LCA to grant a more rigorous status to the conclusions of a study (ISO 2006, JRC-IES 2010). The most popular approach for doing an uncertainty analysis in LCA is the Monte Carlo approach (Lloyd and Ries 2007), partly because it has been implemented in many of the major software programs for LCA, typically as the only way for carrying out uncertainty analysis (for instance, in SimaPro, GaBi, Brightway2, and in openLCA).

The Monte Carlo method is a sampling-based method, in which the calculation is repeated a number of times, in order to estimate the probability distribution of the result (see, e.g., Helton et al. 2006, Burmaster and Anderson 1994). This distribution is then typically used to inform decision-makers about characteristics, such as the mean value, the standard deviation or quantiles (such as the 2.5 and 97.5 percentiles). In LCA, the results are typically inventory results (e.g., emissions of pollutants) or characterization/normalization results (e.g., climate change, human health, etc.). In comparative LCA, such distributions form the basis of paired comparisons and tests of hypothesis (Mendoza Beltran et al. 2018). Many programs and studies offer or present visual aids for interpreting the results, including histograms and boxplots (Helton et al. 2006; McCleese and LaPuma 2002).

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A disadvantage of the Monte Carlo method is that it can be computationally expensive. Present-day LCA studies can easily include 10,000 or more unit process, and calculating such as system can take some time. Repeating this calculating for a new configuration then takes the same time, and this is repeated a large number of times. Finally, the stored results must be analyzed in terms of means, standard deviations, p values and visual representations. Altogether, if we use the symbol N_{run} to refer to the number of Monte Carlo runs, the symbol T_{cal} for the CPU time needed to do one LCA calculation, and T_{ana} for the time needed to process the Monte Carlo results, the total time needed, T_{tot}, is simply

$$T_{\text{tot}} = N_{\text{run}} \times T_{\text{cal}} + T_{\text{ana}}$$

Usually, $T_{\text{cal}} > T_{\text{ana}}$ and certainly $N_{\text{run}} \times T_{\text{cal}} \gg T_{\text{ana}}$, so that we can write

$$T_{\text{tot}} \approx N_{\text{run}} \times T_{\text{cal}}$$

and further ignore the aspect of T_{ana}.

The time needed for a Monte Carlo analysis is thus determined by two factors: T_{cal}, which is typically in the order of seconds or minutes, and N_{run}. A normal practitioner has little influence on T_{cal}, as it is dictated by the combination of algorithm, the hardware, and the size of the database. Typically, it is between 1 s and 5 min. (This is a personal guess; there is no literature on comparative timings using a standardized LCA system). A practitioner has much more influence on the number of Monte Carlo runs, N_{run}. So, the trick is often to take N_{run} not excessively high, say 100 or 1000. On the other hand, it has been claimed that this number must be large, for instance 10,000 or even 100,000. For instance, Burmaster and Anderson (1994) suggest that "the analyst should run enough iterations (commonly ≥10,000)," and the authoritative Guide to the Expression of Uncertainty in Measurement (BIPM 2008) writes that "a value of $M = 10^6$ can often be expected to deliver [a result that] is correct to one or two significant decimal digits." In the LCA literature, we find similar statements, for instance by Hongxiang and Wei (2013) ("more than 2000 simulations should be performed") and by Xin (2006) ("[it] should run at least 10,000 times"). Such claims also end up in reviewer comments: We recently received the comment "Monte Carlo experiments are normally run 5000 or 10,000 times. In the paper, Monte Carlo experiments are only run 1000 times. Explain why?". With the pessimistic $T_{\text{cal}} =$ 5 min, using $N_{\text{run}} = 100{,}000$ runs will require almost 1 year. If we take the short calculation time of $T_{\text{cal}} = 1$ s, we still need more than one full day. And, even Brightway2's (https://brightwaylca.org/) claim of "more than 100 Monte Carlo iterations/second" (of which we do not know if this also applies to today's huge systems) would take more than 16 min. Such waiting times may be acceptable for Big Science, investigating fundamental questions on the Higgs boson or the human genome. But, for a day-by-day LCA consultancy firm, even 1 h is much too long.

In this study, we investigate the role of N_{run}. We will in particular focus on the original purpose of the Monte Carlo technique vis-à-vis its use in LCA, and consider the fact that in LCA, the input probability distributions are often based on small samples, or on pedigree-style rules-of-thumb, as well as the fact that in LCA, we are in most cases interested in making comparative statements ("product A is significantly better than product B").

The next section discusses the elements of the analysis: the mathematical model and its probabilistic form, the description of probabilistic ("uncertain") data, the estimation of input data, and the estimation of output results. Section 3 provides two numerical examples. Section 4 finally discusses and concludes.

2 Probability theory

In this section, we discuss a few background topics from probability theory. The interested reader is referred to general textbooks, such as Ross (2010) and Gharamani (2005).

2.1 Mathematical models

When a model needs several input variables to compute an output variable, we can abstractly write the model relation as

$$y = f(x_1, x_2, \ldots)$$

Here, x_1, x_2, … represent the values of the input variables (the data, for instance CO_2 coefficients and electricity requirements) and y is the output (the result, for instance a carbon footprint). The function $f(\cdot)$ is a specification of the LCA algorithm (Heijungs and Suh 2002). We will assume that this algorithm is known and fixed, and that it has been implemented in software in a reliable way and therefore does not introduce any uncertainty (however, see Heijungs et al. 2015).

2.2 Probabilistic models

Uncertainty can enter the scene in different ways:

- When the input data is not exactly known (for instance, the effect of glyphosate on human health is not fully known)
- When the input data displays variability (for instance, the lifetime of identical light bulbs is not exactly equal)
- When choices must be made by the analyst (for instance, allocation factors can be based on mass or on economic value)

Sometimes, additional sources of uncertainty are mentioned (Huijbregts 1998), such as model uncertainty. Here, we restrict the discussion to those types of uncertainty that can be phrased as inputs (x_1, x_2, etc.) in the model equation ($f(\cdot)$). Our analysis can, however, easily be broadened to cover such cases. For instance, we can include allocation choices as an extra input parameter into $f(\cdot)$. (Heijungs et al. 2019).

2.3 Probability distributions of input variables

In a probabilistic model, we can specify the input data as a probability distribution (continuous or discrete). So, from now on, we will assume that x_1, x_2, … are not fixed numbers, but that they are stochastic (random) numbers, following some probability distribution. We will use the convention from probability theory to indicate stochastic variables with capital letters, like X_1, X_2, … Further, the symbol ~ indicates that a stochastic variable is distributed according to some probability distribution. For instance,

$$\begin{cases} X_1 \sim N\left(\mu_{X_1}, \sigma_{X_1}\right) \\ X_2 \sim N\left(\mu_{X_2}, \sigma_{X_2}\right) \\ \cdots \sim \cdots \end{cases}$$

where $N(\mu, \sigma)$ is the normal (Gaussian) probability distribution with parameters μ and σ. We might go for other probability distributions (uniform, log-normal, binomial, etc.) but at this stage want to keep the discussion simple. The numbers that specify the numerical details of the probability distribution (here μ and σ in general, and more specifically μ_{X_1}, μ_{X_2}, σ_{X_1}, σ_{X_2}, etc.) are referred to as parameters. So, not x_1 is a parameter (as the usual terminology in LCA goes), but rather μ_{X_1} and σ_{X_1} are parameters of the distribution of X_1. Other types of distributions are usually specified with different types of parameters (for instance, the uniform distribution with a parameter for the lower limit and a parameter the upper limit) or even with another number of parameters (for instance, the Poisson distribution requires only one parameter, while the asymmetric triangular distribution requires three parameters).

2.4 Probability distributions of output variables

Recognizing that (some of) the input parameters of the model $f(\cdot)$ are stochastic, a logical consequence is that the model output is also stochastic. Thus, we write

$$Y = f(X_1, X_2, \ldots)$$

See Heijungs et al. (2019). With this change of y into Y, our task shifts from calculating the value of y to calculating the distribution of Y. More specifically, we may want to know:

- The shape of the distribution of Y (i.e., normal, uniform, log-normal, binomial, etc.)
- The value or values of the parameter or parameters (e.g., μ_Y and σ_Y)

Probability theory offers methods to calculate the probability distribution of Y when those of X_1, X_2, … are given, but only for a few cases of $f(\cdot)$ and only for a few input distributions. For instance, when $Y = f(X_1, X_2) = X_1 + X_2$ and X_1 and X_2 are normal, every textbook shows that

$$Y \sim N\left(\mu_{X_1} + \mu_{X_2}, \sqrt{\sigma_{X_1}^2 + \sigma_{X_2}^2}\right)$$

In words, the sum of two normal variables is itself normally distributed, and the parameters μ_Y and σ_Y can easily be calculated from the parameters of the input distributions. Another case is $Y = f(X_1, X_2) = X_1^2 + X_2^2$. This is pretty complicated, but when we take the special case of $\mu_{X_1} = \mu_{X_2} = 0$ and $\sigma_{X_1} = \sigma_{X_2} = 1$, it is a well-known result:

$$Y \sim \chi^2(2)$$

where $\chi^2(\nu)$ is the chi-squared distribution with parameter ν. In general, most choices of $f(\cdot)$ with less trivial combinations of X_1, X_2, … (such as $f(X_1, X_2) = X_1 X_2^2 + \frac{\ln X_1}{4 + \sin X_2}$) are not manageable by the theory of probability. It is therefore important to have an alternative way to determine the probability function of such more complicated functions of stochastic variables. The same applies also to situations where $f(\cdot)$ is straightforward, but where the input distributions for X_1, X_2, etc. are not normal.

The Monte Carlo approach (Metropolis and Ulam 1949; Shonkwiler and Mendivil 2009) can be used as an alternative way for constructing the probability distribution of Y in case the mathematical approach is too hard. It is based on artificially sampling values from Y, and using this sample for reconstructing (the technical term is estimating) the shape and the parameter values of Y. We will spend the next section on the topic of estimating a probability distribution from a sample of values. This is a topic of more general interest than Monte Carlo simulations, so we will keep the discussion quite general, also covering the case of estimating the distribution of input variables like X_1 and X_2.

2.5 Estimating a probability distribution in general

We will discuss the question of estimating a probability distribution Z (including its parameters), given a sample of data, z_1, z_2, …, z_n. This task is known as the estimation problem, and it is one of the central topics of inferential statistics. See, for instance, Rice (2007) and Casella and Berger (2002) for general textbooks.

Suppose we have a sample of data from an unknown stochastic process, Z. Let the sampled values be indicated by z_i, for $i = 1, \ldots, n$. If we want to estimate the probability distribution belonging to the stochastic process that generated this sample, we must first make an assumption about the type of distribution. Is it a normal distribution, a uniform distribution, a log-normal distribution, a Weibull distribution? This choice is one of the trickiest parts of the entire estimation process, because there is no clear guidance. Different aspects can play a role here:

- Evidence: the data (e.g., a histogram or a boxplot) may suggest a certain distribution.
- Conventions and compatibility with software: the log-normal distribution has a longer and more widespread history in LCA than the Erlang distribution.
- Familiarity and simplicity: if the histogram looks approximately bell-shaped, a normal distribution is more natural than the Cauchy distribution.
- Statistical criteria: we can use statistical tests (such as those by Kolmogorov-Smirnov and Anderson-Darling) to assess the goodness-of-fit with a number of probability distributions.

Clearly, there are also cases where none of the conventional model distributions provides a satisfactory fit with the empirical data. We will not further discuss such cases, because the usual procedure in LCA is to model input uncertainties in terms of just a few distributions: lognormal, normal, uniform, or triangular (Frischknecht et al. 2004) or perhaps a few more (gamma and beta PERT; see Muller et al. 2016).

Once we have selected a probability distribution, the next task is to estimate the parameter value or values of that distribution. Suppose we have selected a normal distribution, so

$$Z \sim N(\mu_Z, \sigma_Z)$$

where μ_Z and σ_Z are the distribution's parameters, which are still unknown at this stage of the analysis. Then, our task is to estimate the values of μ_Z and σ_Z that correspond best with the sampled data. Different estimation principles are available in the statistical literature to do this. Two widely used principles are the method of moments and the method of maximum likelihood. For the case of a normal distribution, these two principles yield the same estimate of μ_Z and σ_Z, but for some distributions, there is a difference in the outcome of the estimation procedure. Anyhow, the theory of statistics offers formulas for estimators, which are functions of the observations. We can use the symbol of the parameter to be estimated with a hat on top of it to indicate the estimator: $\hat{\mu}$ is an estimator of μ and $\hat{\sigma}$ is an estimator of σ. In the case of a normal distributions, both estimation principles (method of moments and method of maximum likelihood) suggest

$$\hat{\mu}_Z = \frac{1}{n}\sum_{i=1}^{n} Z_i$$

and

$$\hat{\sigma}_Z = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(Z_i - \hat{\mu}_Z\right)^2}$$

as estimators for μ_Z and σ_Z. When applied to a concrete data set, $z_1, z_2, \ldots, z_n$, these estimators produce a concrete value, because we insert the observed values of z_i at the place of the stochastic variable Z_i. These concrete values are the estimates, which we will indicate hereafter as $\bar{z}$ and s_Z.

Of course, we cannot expect that the estimates will be fully accurate if the sample size is finite. The estimate $\bar{z}$ will be hopefully close to the true value μ_Z, but probably it will be a little bit off (that is also why we distinguish the symbols: in general $\bar{z} \neq \mu_Z$, but $\bar{z} \approx \mu_Z$). The same applies to the estimate s_Z of σ_Z.

The theory of inferential statistics not only allows to estimate the values, but it also allows us to say something about the level of precision of such estimates. This is done through the theory of sampling distributions, standard errors, and confidence intervals.

A sampling distribution is the probability distribution of an estimator. Let us suppose we have a probability distribution $Z \sim N(\mu_Z, \sigma_Z)$, with unknown parameter μ_Z and known parameter σ_Z, from which we sample n observations, and use the estimator $\hat{\mu}_Z$ to estimate μ_Z by the value $\bar{z}$. If we would take another sample of size n, we can use the same estimator to again estimate μ_Z, but we will find a slightly different value $\bar{z}$, because the sample will contain different values. Repeating and repeating, always with the same sample size n, we will end up with a distribution of $\bar{z}$ values. This distribution will be referred to as $\overline{Z}$.

The famous central limit theorem states that the distribution of the estimates of the mean, $\overline{Z}$, is normally distributed and that there is a simple relation between its parameters ($\mu_{\overline{Z}}$ and $\sigma_{\overline{Z}}$) and the parameters of the parent distribution Z (μ_Z and σ_Z):

$$\overline{Z} \sim N\left(\mu_Z, \frac{\sigma_Z}{\sqrt{n}}\right)$$

So, $\mu_{\overline{Z}} = \mu_Z$ and $\sigma_{\overline{Z}} = \frac{\sigma_Z}{\sqrt{n}}$. This first fact signifies that the mean of the sample means corresponds to the mean of the parent distributions. This is a convenient property, because it allows to use the sample mean ($\bar{z}$) as the best guess of μ_Z. The second fact tells us that the width of the distribution of $\overline{Z}$ (so $\sigma_{\overline{Z}}$) depends on the width of the distribution of Z (so on σ_Z) and on the size of the sample (so on n). In fact, $\sigma_{\overline{Z}}$ decreases without limits when n increases. The important consequence is that the estimate of μ_Z, $\bar{z}$, is more precise when n is large and that we can determine its value as precisely as we want by just increasing sample size. The larger the sample, the more precise the estimate.

The quantity $\sigma_{\overline{Z}} = \frac{\sigma_Z}{\sqrt{n}}$ is known as the standard error of the mean, also known as "the" standard error. For a precise estimation of μ_Z, we want this $\sigma_{\overline{Z}}$ to be small. The only way to do so is to use a large sample size n, because σ_Z is fixed. The standard error is related to the concept of a confidence interval. For the case of estimating μ_Z, the 95% confidence interval is given by

$$CI_{\mu_Z;0.95} = \left[\overline{z} - 1.96\sigma_{\overline{Z}}, \overline{z} + 1.96\sigma_{\overline{Z}}\right]$$

This means that with 95% confidence, the interval CI will contain the true value μ_Z that we are supposed to estimate by $\overline{z}$. Observe that the confidence interval has a width of $2 \times 1.96\sigma_{\overline{Z}} = 3.92\sigma_{\overline{Z}} = 3.92\frac{\sigma_Z}{\sqrt{n}}$. If we want this interval to be smaller, we need to increase sample size n.

Above, we discussed how to estimate the parameter μ when the parameter σ is known. Estimation of σ and other parameters, and estimation of μ when σ is unlnown, are technically more difficult, but conceptually the idea is the same.

2.6 Estimating the probability distribution of input variables

When we want to estimate the probability distribution of an input variable (X_1, etc.), we carry out the following steps:

- We sample data ($x_{11}, x_{12}, \ldots, x_{1n}$) from the phenomenon (e.g., unit process).
- We choose a convenient probability distribution shape (e.g., normal).
- We use the formulas for the estimators ($\hat{\mu}_{X_1}$, $\hat{\sigma}_{X_1}$, etc.) to find estimates ($\overline{x_1}$, s_{X_1}, etc.).

The estimated parameter values ($\overline{x_1}$, s_{X_1}, etc.) are "best guesses" given the available data. However, we cannot expect that they are perfect estimates, because the confidence interval of these parameters decreases with $\frac{1}{\sqrt{n}}$, and n is usually limited. Of course, we can increase n by collecting more primary data, but site visits and measurements are usually expensive and time-consuming. For that reason, in LCA, as in most other fields of science, n is usually quite limited. The price we pay for that is a larger standard error and a wider confidence interval.

2.7 Estimating the probability distribution of output variables, given perfectly known inputs

Next, we move to the topic of estimating the probability distribution of an output variable (Y, etc.). Suppose, for simplicity, we have one stochastic input variable, X, normally distributed, with known parameters:

$$X \sim N(\mu_X, \sigma_X)$$

Next, we define a very simple function of that variable:

$$Y = f(X) = X$$

Of course, the distribution of the output variable Y is trivial:

$$Y \sim N(\mu_X, \sigma_X)$$

and in particular, $\mu_Y = \mu_X$. But, let us pretend we are bad in probability theory and prefer to use a Monte Carlo approach. We simulate N_{run} instances of X (namely $x_1, x_2, \ldots, x_{N_{run}}$) and use that to calculate N_{run} instances of Y (namely $y_1 = x_1$, $y_2 = x_2$, etc.). These values of y are used to estimate μ_Y as follows:

$$\overline{y} = \frac{1}{N_{run}} \sum_{i=1}^{N_{run}} y_i$$

When the sample has been obtained in a random way, we can also be sure that the estimate will converge to the correct value:

$$\lim_{N_{run} \to \infty} \overline{y} = \mu_Y = \mu_X$$

Likewise, we can estimate the standard deviation of Y, σ_Y. This can be used to find the standard error of the mean

$$s_{\overline{Y}} = \frac{s_Y}{\sqrt{N_{run}}}$$

The noteworthy aspect of this standard error is that it will go to zero when N_{run} grows very large:

$$\lim_{N_{run} \to \infty} s_{\overline{Y}} = 0$$

As a consequence, the estimate of μ_Y will become arbitrarily precise, if we have enough computer time:

$$\lim_{N_{run} \to \infty} CI_{\mu_Y;0.95} = [\mu_Y, \mu_Y] = [\mu_X, \mu_X]$$

That is not surprising. If we would have been more thoughtful, we could have saved the computer expenses and directly deduce that $\mu_Y = \mu_X$, with infinite precision. The situation is comparable to computing $\frac{1}{2} + \frac{1}{4} + \frac{1}{8} + \frac{1}{16} + \ldots$, for a large number of terms, or being more thoughtful and directly writing this as $\frac{\frac{1}{2}}{1-\frac{1}{2}} = 1$. Both approaches yield approximately the same result. So, when we want to use a Monte Carlo approach to estimate the parameters of a probability distribution, we must use a large sample size N_{run} to find a reliable answer. The recommendations quoted in the introduction (1000, 10,000, 100,000) are based on the situation described here: accurately estimating an output distribution on the basis of perfect knowledge of the input distributions.

2.8 Estimating the probability distribution of output variables, given imperfectly known inputs

But now, take the next case, a normal distribution with parameters μ_X and σ_X, but under the provision that μ_X itself is slightly off, because we did not know μ_X but used its imperfect estimate $\overline{x}$. So, we consider

$$X \sim N\left(\overline{x}, \sigma_X\right)$$

Next, we again study the trivial function

$$Y = f(X) = X$$

first analytically, using probability theory, and then through a Monte Carlo simulation.

Analytically, we find

$$Y \sim N\left(\overline{x}, \sigma_X\right)$$

The essential point to observe is that the mean of Y is not μ_X but $\overline{x}$, which is likely to be somewhat wrong.

Next, let us try this by a Monte Carlo simulation. We use $\overline{y}$ to estimate μ_Y. It will be close to $\overline{x}$, rather than close to μ_X. Moreover, the standard error of this estimate is still $s_{\overline{Y}} = \frac{s_Y}{\sqrt{N_{\text{run}}}}$, so as close to 0 as we like. In fact,

$$\lim_{N_{\text{run}} \to \infty} CI_{\mu_Y;0.95} = [x, x]$$

Summarizing, using probability theory and using the Monte Carlo approach, both will give you the wrong value ($\overline{x}$ instead of μ_X) when estimating μ_Y, and the Monte Carlo approach will in addition suggest that this estimate is very precise due to a vanishing standard error, at least when N_{run}.is very large.

Observe that this is not a mistake or limitation of the Monte Carlo approach. In fact, it performs very well. The mistake is entirely due to the analyst, who uses an imperfectly estimated input parameter ($\overline{x}$ instead of μ_X) to run an infinite-precision method. Also, observe that this is a very ubiquitous situation in LCA: Most LCA data on unit processes is obtained from limited samples. Even a sample size of 1 is not uncommon. There is even a widely used approach, referred to as the pedigree approach and popularized by the ecoinvent database, of which the purpose is to estimate a probability distribution on limited data (Frischknecht et al. 2004; Weidema et al. 2013). We devote a longer discussion to this problem toward the end of this paper.

3 Numerical illustration

To test and illustrate these ideas, we did two simulation experiments, first for one stand-alone system, and then for two systems in a comparative analysis.

To illustrate the situation for one system, we made a small code in R (Fig. 1) and used it to simulate the following case:

- The parent distribution is $X \sim N(10, 1)$.
- We sample $n = 16$ observations, and estimate μ_X by $\overline{x}$.
- We draw from $Y \sim N(\overline{x}, \sigma_X)$ a Monte Carlo sample of size $N_{\text{run}} = 100{,}000$.
- From this sample, we estimate μ_Y by $\overline{y}$.

In our simulation, the results were as follows:

- $\overline{x} = 10.31$, $\sigma_{\overline{X}} = 0.25$, so the 95% confidence interval for μ_X is [9.819, 10.799].
- $\overline{y} = 10.31$, $\sigma_{\overline{Y}} = 0.0031$, so the 95% confidence interval for μ_Y is [10.305, 10.318].

The interpretation of these results are as follows:

- We misestimate μ_X (10.31 instead of 10.00).
- But, we acknowledge that it may be wrong, and in fact, our 95% confidence interval contains the correct value (it suggests a value somewhere between 9.8 and 10.8).
- We misestimate μ_Y (10.31 instead of 10.00).
- But, we deny that it may be wrong, because our 95% confidence interval is pretty sure about a value somewhere 10.30 and 10.32.

In conclusion, the Monte Carlo approach will yield a very precise, but inaccurate, result.

Fig. 1 R code for generating a large Monte Carlo sample (N_{run} = 100,000) from an input distribution with limited precision

```
mu_x <- 10 # define population mean for X
sigma_x <- 1 # define population standard deviation for X
n <- 16 # define sample size for X
x <- rnorm(n, mu_x, sigma_x) # generate vector of random numbers for X
x_bar <- mean(x) # estimate mean of X
N_run <- 100000 # define number of Monte Carlo runs
y_mc <- rnorm(N_run, x_bar, sigma_x) # generate the Monte Carlo result
# compute 95% confidence interval
c(mean(y_mc)-1.96*sd(y_mc)/sqrt(N_run), mean(y_mc)+1.96*sd(y_mc)/sqrt(N_run))
```

The precision of an estimate plays an important role in testing statistical hypotheses. When we would like to test a statement like $\mu_X = 10$, the null hypothesis significance testing procedure would not reject the null hypothesis, because the hypothesized value of 10 is in the 95% confidence interval [9.819, 10.799]. On the other hand, the same procedure when applied to the null hypothesis $\mu_Y = 10$ would lead to a rejection, because 10 is not in the 95% confidence interval [10.305, 10.318].

The second example is about two systems, A and B, in a comparative LCA: Seemingly precise estimates of the impact of products A and B can lead to the conclusion that A is better than B, while the real situation is that B is better than A. Or we find that A is better than B, although they do not differ. To test and illustrate this phenomenon, we made another computer experiment (Fig. 2). We generate $n = 16$ samples from $X_A \sim N(10, 1)$ and $X_B \sim N(10, 1)$. From these two samples, we estimate μ_{X_A} through $\overline{x}_A$ and μ_{X_B} through $\overline{x}_B$ and do a two-sample t test to test the hypothesis $\mu_{X_A} = \mu_{X_B}$. Next, we use $Y_1 = f(X_1) = X_1$ and $Y_2 = f(X_2) = X_2$, and sample $N_{run} = 100{,}000$ values from Y_A and Y_B. From this Monte Carlo sample, we test the null hypothesis $\mu_{Y_A} = \mu_{Y_B}$. The p value of the first test was 0.67 providing strong evidence of equality of μ_{X_A} and μ_{X_B}. The second test yielded a p value around 10^{-16}, pointing to overwhelming evidence that $\mu_{Y_A} \neq \mu_{Y_B}$.

This comparative case is even more interesting than the first example, because decisions about purchases, ecolabels, etc. are often taken on the basis of comparative assessments: Is there evidence that one product is significantly better than another product? Statistical hypothesis testing can provide an answer to such questions, but the example shows that inaccurately specified parameters of the parent distributions may give a seemingly convincing wrong answer, because an excessive number of Monte Carlo runs will optimize precision, ignoring inaccurate inputs.

4 Discussion and conclusions

Let us be a bit more explicit on the terminology: An estimate can be imprecise or it can be inaccurate. The two have been illustrated in various ways (Fig. 3). In our analysis of example 1, we have an inaccurate estimate ($\overline{y}$ can be off quite a bit due to small n in determining $\overline{x}$) with arbitrary high precision ($\sigma_{\overline{Y}}$ is almost zero due to very large N_{run}). By reporting a very small standard error of the mean, we suggest to have done a high-quality calculation.

The discussion above took a very trivial function, namely $Y = f(X) = X$ as starting point. The storyline is no different for more complicated cases, such as $Y = f(X_1, X_2) = X_1 X_2^2 + \frac{\ln X_1}{4 + \sin X_2}$ or for functions of hundreds of input distributions $Y = f(X_1, X_2, \ldots)$. Likewise, we used a normal distribution with known standard deviation to start with. If the standard deviation is unknown, or if the parent distribution is of a different type (log-normal, binomial, ...), the mathematics is more difficult, but the take home message remains the same: with an imprecise estimate of the input parameters, we can make a very precise but probably inaccurate estimate of the output parameters. Garbage in, garbage out, but the type of garbage has changed: from imprecise to inaccurate. That is a problem, because imprecision is visible through a large standard error of the mean ($\overline{x} = 10.31 \pm 0.25$), while inaccuracy is not visible ($\overline{y} = 10.31 \pm 0.0031$). As a result, the estimate will suggest to be of high quality where it is not.

Superficially, it sounds better to make precise statements than imprecise statements. But, when the statements are on inaccurate values, this is not necessarily true.

In a statistical analysis, we can always draw wrong conclusions (type I errors: not rejecting an incorrect null hypothesis, type II errors: rejecting a correct null hypothesis), but this is a completely different type of error: rejecting a null hypothesis for which we have no appropriate data. The root of the problem is that we sample from inaccurately specified distributions. While we would naively expect that this leads to inaccurate results, the statistical analysis neglects the inaccuracy and concentrates on the precision. The imprecision declines with the number of Monte Carlo runs, but the inaccuracy does not. And, imprecision is visible, while inaccuracy is invisible.

The remedy is to maintain the imprecision in the estimate of the input parameters. As long as the parameters of the input distributions are imprecise, we should not be allowed to decrease the precision of the output distribution estimates

Fig. 2 R code for testing the hypothesis of equality of means in the input data X_1 and X_2, generated from small samples ($n_{X_A} = n_{X_B} = 16$), and of equality of means in the output results Y_1 and Y_2, generated with a large Monte Carlo sample (N_{run} = 100,000)

```
mu_xA <- 10 # define population mean for XA
sigma_xA <- 1 # define population standard deviation for XA
n_xA <- 16 # define sample size for XA
xA <- rnorm(n_xA, mu_xA, sigma_xA) # generate vector of random numbers for XA
mu_xB <- 10 # define population mean for XB
sigma_xB <- 1 # define population standard deviation for XB
n_xB <- 16 # define sample size for XB
xB <- rnorm(n_xB, mu_xB, sigma_xB) # generate vector of random numbers for XB
xA_bar <- mean(xA) # estimate mean of XA
xB_bar <- mean(xB) # estimate mean of XB
t.test(xA, xB) # testing equality of means of XA and XB
N_run <- 100000 # define number of Monte Carlo runs
yA <- rnorm(N_run, xA_bar, sigma_xA) # generate the Monte Carlo result for YA
yB <- rnorm(N_run, xB_bar, sigma_xB) # generate the Monte Carlo result for YB
t.test(yA, yB) # testing equality of means of YA and YB
```

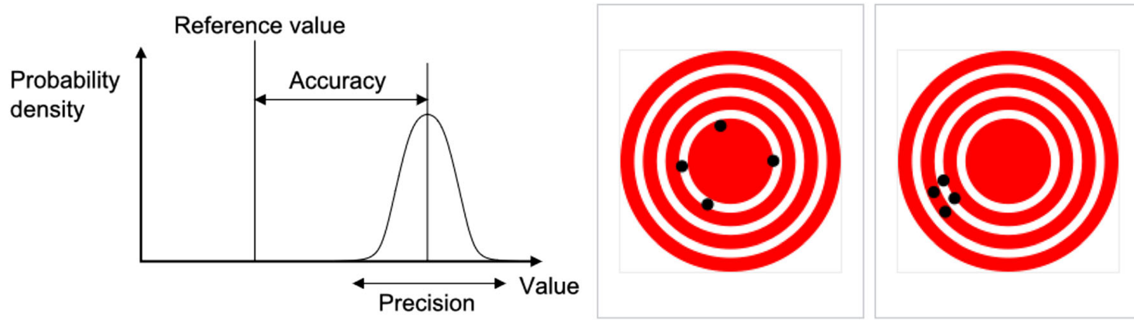

Fig. 3 Illustration of the difference between precision and accuracy. The left figure illustrates both; the middle one is an example of low precision and the right one is an example of low accuracy. Source: https://en.wikipedia.org/wiki/Accuracy_and_precision

without limits. How can this be done? One simple way is to put an upper limit to the number of Monte Carlo runs. If the estimate of the input parameter μ_X is based on a sample of n = 16 data points, perhaps we should not do more than N_{run} = 16 Monte Carlo runs. While this sounds fair, a complication is that we need more guidance on the case of more complicated functions than just $Y = X$, for instance $Y = X_1X_2^2 + \frac{\ln X_1}{4+\sin X_2}$. If X_1 has been sampled with $n_{X_1} = 16$ and X_2 with $n_{X_2} = 9$, what should we take for the number of Monte Carlo runs, N_{run}? Perhaps the weakest link defines our maximum quality, so our Monte Carlo run could do with just 9 runs in this case. The result is a very imprecise estimate of μ_Y, but visibly imprecise. The solution of taking a small number of Monte Carlo runs by the way also solves the problem of overly significant results (Heijungs et al. 2016).

Another remedy is of course to determine the parameters of the input distributions with more precision, so using a larger sample size n_{X_1}, n_{X_2}, etc. In practice, this is, however, not easy. Many of the millions of data in the LCA model come from general purpose generic databases, and recollecting these data from multiple sites and at multiple days would be a horrendous task.

A final point is the case of probability distributions with parameters that have not been estimated from data, but for which a procedural estimation has been used. An important example is the earlier-mentioned pedigree approach, where data quality indicators, for instance for representativeness and age, define default standard deviations. The popular ecoinvent database is a major example here (Frischknecht et al. 2004; Weidema et al. 2013), but the approach is also becoming popular in other areas (Laner et al. 2016). For such data, it is often unclear what the sample size of the data is, so it is not possible to estimate the precision of the mean in terms of a standard error. But, it will be clear that the parameters of the input distribution are not at all accurate, so a propagation into almost infinitely precise Monte Carlo output results is as misleading as the parameter-based procedure on which our main argument was based. An ultimate consequence is that such pedigree-based probability distributions are incompatible with large-scale Monte Carlo simulations. This is an important take-home message of our analysis, because the pedigree approach has grown into a major paradigm for estimating standard deviations of LCA data, and Monte Carlo has become the default procedure for propagating uncertainties in LCA. The incompatibility of the two has, as far we know, not been recognized before, and our analysis does not suggest any way out. This suggests a major area of research in dealing with uncertainty in LCA.

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Comparison of Asian Aquaculture Products by Use of Statistically Supported Life Cycle Assessment

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Supporting Information

ABSTRACT: We investigated aquaculture production of Asian tiger shrimp, whiteleg shrimp, giant river prawn, tilapia, and pangasius catfish in Bangladesh, China, Thailand, and Vietnam by using life cycle assessments (LCAs), with the purpose of evaluating the comparative eco-efficiency of producing different aquatic food products. Our starting hypothesis was that different production systems are associated with significantly different environmental impacts, as the production of these aquatic species differs in intensity and management practices. In order to test this hypothesis, we estimated each system's global warming, eutrophication, and freshwater ecotoxicity impacts. The contribution to these impacts and the overall dispersions relative to results were propagated by Monte Carlo simulations and dependent sampling. Paired testing showed significant ($p < 0.05$) differences between the median impacts of most production systems in the intraspecies comparisons, even after a Bonferroni correction. For the full distributions instead of only the median, only for Asian tiger shrimp did more than 95% of the propagated Monte Carlo results favor certain farming systems. The major environmental hot-spots driving the differences in environmental performance among systems were fishmeal from mixed fisheries for global warming, pond runoff and sediment discards for eutrophication, and agricultural pesticides, metals, benzalkonium chloride, and other chlorine-releasing compounds for freshwater ecotoxicity. The Asian aquaculture industry should therefore strive toward farming systems relying upon pelleted species-specific feeds, where the fishmeal inclusion is limited and sourced sustainably. Also, excessive nutrients should be recycled in integrated organic agriculture together with efficient aeration solutions powered by renewable energy sources.

1. INTRODUCTION

Aquaculture is the only solution for meeting the growing demand for aquatic products in a world where capture fishery catches have stagnated.[1,2] Asia is the main producing region with 88% of global aquaculture production by volume, and the European Union (EU) is the largest single market with 36% of total world imports by value.[1] However, while consumption trends have rapidly increased in the European Union, concerns have been raised regarding the environmental sustainability of fish and crustacean products imported from Asia. These concerns are

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associated with detrimental environmental consequences such as global warming, eutrophication, ecotoxicity, land-use and land-use change (LULUC), excessive energy use, and freshwater use.[3−5]

Environmental impacts related to aquaculture commodities have been quantified in various life cycle assessment (LCA) studies.[3] However, only a handful of these have focused on Asian aquaculture. Four LCA studies have evaluated Vietnamese pangasius catfish,[6−9] three have studied shrimp farming,[4,10,11] two have focused on Indonesian finfish,[12,13] and one has studied Thai finfish.[14] Only three of these quantified the uncertainties related to results.[4,11,15] Little is therefore known about the level of confidence behind conclusions made in previous studies, despite the increasing importance of LCA results in policy contexts.[9] Seafood standards are, for example, starting to incorporate carbon footprints into their recommendations,[16] and a PAS2050 (publicly available specification) standard has been developed for seafood and other aquatic food products.[17] For such standards to be realistic and effective, differences in impact need to be statistically substantiated.

In the present study, we performed LCAs and statistically evaluated the environmental impacts for some of the most common Asian aquaculture commodities found on European markets[15] (see Table 1). From this selection, the most important producing regions and production systems were identified and evaluated.[15,18,19] Noteworthy is that some of these production systems currently are not eligible for export due to existing import regulations into the European Union (e.g., tilapia integrated with pigs in China). System characterization was based on farm scale, pond type, species combination, and other features of the production systems.[15,19]

The present study builds upon the final LCA case study report[15] (available at: http://media.leidenuniv.nl/legacy/d35-annexreport.pdf) of the Sustaining Ethical Aquaculture Trade project (www.seatglobal.eu) but also includes calculated freshwater aquatic ecotoxicity potential characterization factors (FAETPs) for a number of aquaculture-related chemicals by use of the USEtox model, including uncertainty estimates for characterization factors.[20]

In order to provide a level of confidence behind conclusions, the hypothesis "different production systems providing the same aquaculture commodity to European consumers are associated with different environmental impacts" was tested statistically. The null hypothesis tested assumed that the environmental life-cycle impacts of commodities originating from different aquaculture system were equal (e.g., system A = system B).

Two approaches were used for testing the differences between paired results as obtained in dependent sampling:[9] one used significance tests (H_0: $m_A = m_B$ at $\alpha = 0.05$), and the other analyzed the percentage of Monte Carlo (MC) runs in which the difference was lower or higher than 0 [$p(x_A - x_B < 0)$ or $p(x_A - x_B > 0)$ at $p = 0.95$]. This dual approach was chosen as each answers different questions. Significance tests for the median analyze whether the distribution of differences has a median that deviates significantly from zero, while MC frequencies indicate how often a type of farming system is expected to perform better than another. Given the large differences in nutritional, culinary, and monetary value of the different species,[21] comparisons were made only across countries and systems, not across species.

2. MATERIALS AND METHODS

2.1. Goal and Scope. The study aimed to evaluate the comparative eco-efficiency per functional unit of 1 tonne of frozen product for some selected aquaculture commodities commonly imported to Europe from Bangladesh, China, Thailand, and Vietnam. The products surveyed were frozen peeled tail-on (PTO) whiteleg shrimp (*Litopenaeus vannamei*), PTO Asian tiger shrimp (*Penaeus monodon*), headless shell-on (HLSO) giant river prawn (*Macrobrachium rosenbergii*), tilapia fillets (mainly *Oreochromis niloticus*), and pangasius catfish fillets (*Pangasianodon hypophthalmus*). The production chains were modeled up to European ports, assuming that any processes (e.g., retailing, cooking, and composting) downstream of this system boundary would be equivalent.

Three impact categories were evaluated: global warming, eutrophication, and freshwater toxicity. The selection of these represents a trade-off among access to good quality data (e.g., important emissions driving some impact categories could not be specified for Asian processes, such as halon causing ozone layer depletion or palladium resulting in abiotic resource depletion), avoidance of extensive multiple comparison problems, diversity of inventory flows and impacts (e.g., acidification gave similar

Table 1. Farming Systems Evaluated in This Study[a]

code	species	region	key characteristics
		Bangladesh	
BD K	giant river prawn	**Khulna**	avg 2 kg of fish coproduced/kg of prawn
BD B	giant river prawn	**Bagerhat**	avg 3.3 kg of fish coproduced/kg of prawn
BD S&P	giant river prawn and Asian tiger shrimp	both	**integrated** with Asian tiger shrimp
BD W	Asian tiger shrimp	**West**	lower stocking density, not always fed, with fish
BD E	Asian tiger shrimp	**East**	higher stocking density, no fish
BD S&P	Asian tiger shrimp and giant river prawn	West	**integrated** with giant river prawn
		China	
CN HL	whiteleg shrimp	Guangdong	lined **high-level** ponds with pumped water exchange
CN LL	whiteleg shrimp	Guangdong	**low-level** earthen ponds with tidal water exchange
CN GD	tilapia	**Guangdong**	intensive to semi-intensive farms, <30 postlarvae·m^2
CN HI	tilapia	**Hainan**	intensive to semi-intensive farms, <30 postlarvae·m^2
CN R	tilapia	both	farmed in freshwater **reservoirs**
CN IG	tilapia	Guangdong	ponds fertilized by **integrated** pigs on dikes
		Thailand	
TH E	whiteleg shrimp	**East**	electricity as main energy source on farm
TH S	whiteleg shrimp	**South**	LPG as main energy source on farm
		Vietnam	
VN SI	Asian tiger shrimp	Soc Trang and Bac Lieu	**semi-intensive** with <30 shrimp postlarvae·m^2
VN I	Asian tiger shrimp	Soc Trang	**intensive** with >30 shrimp postlarvae·m^2
VN I	whiteleg shrimp	Ben Tre	**intensive** with >30 shrimp postlarvae·^{2}
VN S	pangasius catfish	An Giang and Can Tho	**small** farms with no full-time labor
VN M	pangasius catfish	An Giang and Can Tho	**medium** farms, privately owned with full-time labor
VN L	pangasius catfish	An Giang and Can Tho	**large** corporate farms

[a]Systems will hereafter be referred to by the code or characteristic shown in boldface type.

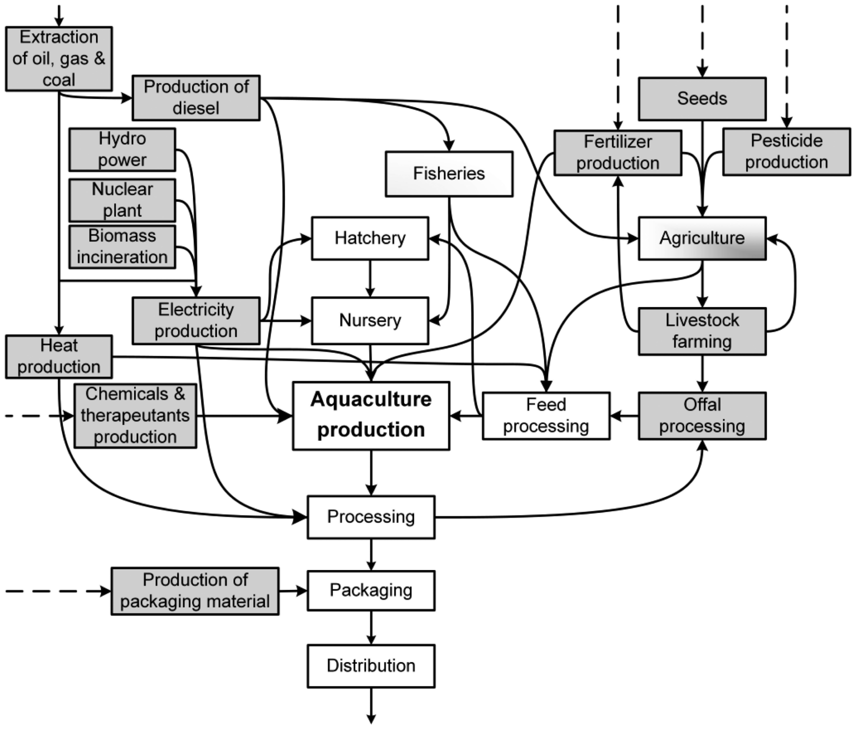

Figure 1. Simplified flowchart of the processes included in this LCA, where arrows symbolize transportation, dashed lines indicate upstream processes, unshaded boxes indicate processes modeled from primary data, and shaded boxes indicate processes modeled from secondary data.

outcomes to global warming[15]), and the different uncertainties they are subject to. Impacts were allocated among multiple coproducts originating from the same process (e.g., fillets and heads from fish processing) based upon mass and economic proceeds (monetary value times mass), in order to evaluate the sensitivity of this highly influential methodological choice[3] and to strengthen conclusions. These two allocation methods were chosen as they generally constitute two extreme outcomes and since they can be consistently applied to all allocation situations. Sensitivity in many other pivotal parameters of aquaculture LCAs (amount of feed used, emissions from agricultural fields and aquatic systems, characterization factors, etc.)[3] was accounted for as part of the variable distribution and therefore considered in the statistical evaluation. Other modeling decisions that could influence outcomes (e.g., cutoff) were not evaluated in the present research, as they were deemed to be of only limited importance to our comparative setup. For a more complete set of impact categories and methodological choices, please see Henriksson et al.[15,18] and Supporting Information.

The data sourcing procedure was based upon the protocol presented in Henriksson et al.[22] Following this protocol, secondary data were weighted (in this study based upon the squared coefficient of variation, wt = $1/CV^2$) according to their inherent uncertainty (inaccuracies in measurements and models) and unrepresentativeness (mismatch between representativeness and use of data), defined by the numerical unit spread assessment pedigree and quantitative uncertainty factors in Frischknecht et al.[23] Overall dispersions were quantified as the sum of inherent uncertainty, spread (variability resulting from averaging), and unrepresentativeness, in accordance with the protocol.[24] Life cycle inventory (LCI) models were constructed, propagated, and characterized by use of CMLCA 5.2 software (www.cmlca.eu) and subsequently aggregated toward the functional unit over 1000 MC simulations with dependent sampling.[9] Covariance was not accounted for in the current models because of methodological limitations. Distributions were tested by the Anderson−Darling goodness-of-fit test in EasyFit v5.5 software (www.mathwave.com), and significance tests were conducted in SPSS v21 (for a more detailed description of the statistical approach, see Supporting Information).

The median impact of each system was pairwise-tested against all other systems used to produce the same commodity, for all three impact categories. Since the distributions were quite skewed, we decided to test equality of medians with the nonparametric Wilcoxon signed-rank test rather than equality of means with a paired *t*-test. Significant differences were considered as $\alpha = 0.05$. However, since 216 comparisons were made among the five species and 20 systems, for two allocation factors and three impact categories, there is over 99.99% probability that at least one of our hypothesis would be a false positive $[1 - (1 - 0.05)$^(36 comparisons $\times$ 2 allocation factors $\times$ 3 impact categories)]. A Bonferroni correction was therefore implemented, adjusting the α level to $\alpha_b = 0.05/216 = 0.00023$.

The alternative approach, looking at the cumulative frequency of one alternative to be favorable to another according to the MC runs, was assumed to hold if cumulative frequencies were higher than 95%, as described by Heijungs and Kleijn[25] and Huijbregts et al.[26]

2.2. Life Cycle Inventory Data Collection. Primary data for the current study involved several actors in the aquaculture value chains (Figure 1). Initial data collection on basic farming practices was conducted between October 2010 and February 2011 for approximately 200 farmers for each species in each of the four countries (in total, about 1400 farmers were interviewed). Farm selection was performed by a random sampling design of farm clusters representing the most important production methods.[19] From this data set, 20 production systems were identified as systematically different based upon basic parameters such as feed used, energy sources, and integrated species[18] (Table 1). A follow-up in-depth survey was then

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Environ. Sci. Technol. 2015, 49, 14176−14183

conducted between 2011 and 2013 with focus on more LCI-specific data and other actors in the aquaculture value chain, including feed mills, capture fisheries, and agricultural producers. A complete set of data is available as Supporting Information and as an annex to SEAT deliverable D3.5[18] (available at http://media.leidenuniv.nl/legacy/d35-annexreport.pdf).

2.3. Life Cycle Impact Assessment Data. Eutrophying emissions were characterized on the basis of the Redfield ratio, with the assumption of an average phytoplankton biomass composition of 106 carbon atoms, 16 nitrogen atoms, and 1 phosphorus atom, as suggested by Heijungs et al.[27] and neglect of any uncertainty. Emissions resulting in global warming were characterized by use of the characterization factors and uncertainty estimates presented in the fifth IPCC report (Table S1).[28,29] Characterization factors for freshwater ecosystem impacts were derived from Rosenbaum et al.[20] or, for noncharacterized chemicals used in aquaculture farming, calculated via the USEtox model (Tables S2−S4). Ecotoxicity data for potentially toxic chemicals applied in aquaculture farms used in the model were sourced primarily from Rico et al.[30] and Van den Brink,[31] and secondarily from the U.S. Environmental Protection Agency's (EPA) ECOTOX database (cfpub.epa.gov; accessed 25 May 2014) (Tables S3 and S4). For chemical characteristics, measured data were prioritized (primarily from sitem.herts.ac.uk/aeru/vsdb/atoz.htm; accessed 25 May 2014) before quantitative structure−activity relationships (QSARs) were used (toxnet.nlm.nih.gov, accessed 25 May 2014; Episuite v4.11 from U.S. EPA). All chemicals applied to agricultural fields and ponds were assumed to be lost to the environment, in consistency with ecoinvent v2.2. For acute exposure, EC_{50} and LC_{50} values were considered, and for chronic exposure, no observed effects concentration (NOEC) and lowest observed effects concentration (LOEC) values were used (see Table S2). Dispersions around the FAETPs were calculated as the sum of dispersions around acute and chronic effect concentrations within and among genera, and the unrepresentativeness of these data. No dispersions were available, however, for the FAETPs readily available in Rosenbaum et al.[20]

3. RESULTS AND INTERPRETATION

Significant conclusions among systems for each species are summarized below. Only conclusions that held for both allocation factors were considered. Relative differences as percentages and contribution analyses are available in Supporting Information (Tables S5−S34 and Figures S1−S3). Dispersions related to the contribution analysis could unfortunately not be quantified by the present approach. These values are instead based upon the so-called baselines (point-value estimates), which in the current study were defined by arithmetic means, in line with the arithmetical structure of CMLCA.[32]

3.1. Asian Tiger Shrimp. Asian tiger shrimp farming in Western Bangladesh was related to significantly lower median global warming and eutrophication impacts than all other systems and also had the lowest median freshwater ecotoxic emissions alongside intensive farming in Vietnam. This is explained by the fact that many Asian tiger shrimp farms in Western Bangladesh use limited feed and/or fertilizer inputs, resulting in a net sink for nutrients. The median eutrophying impacts of Bangladeshi farms in the east were, in the meantime, comparable with those from either of the Vietnamese shrimp farming systems but worse with regard to freshwater ecotoxicity. Asian tiger shrimp integrated with prawn performed the worst for all impact categories except global warming. The poorer performance of the Bangladeshi systems with regard to toxicity was largely due to more extensive use of agricultural products as feed, for which pesticides are used. In Vietnam, intensive production of Asian tiger shrimp had significantly lower ecotoxicological and eutrophying impacts, as compared to semi-intensive production, but similar global warming impacts (Table 2).

Table 2. Ranking of Relative Environmental Performance Related to Asian Tiger Shrimp Provided to European Consumers[a]

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	BD W[a]	BD W[a]	BD W[a]	BD W[a]	BD W[a]	BD W[a]
	BD E[b]	BD E[b]	BD E[b]	VN I[b]	VN I[b]	VN I[a]
	BD S&P[c]	VN SI[c]	VN I[c]	VN SI[c]	VN SI[c]	VN SI[b]
	VN I[d]	VN I[d]	VN SI[d]	BD E[c]	BD E[d]	BD E[c]
Worst	VN SI[d]	BD S&P[e]	BD S&P[e]	BD S&P[d]	BD S&P[e]	BD S&P[d]

[a]VN = Vietnam; BD = Bangladesh; I = intensive; SI = semi-intensive; W = west; E = east; S&P = shrimp and prawn. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test, and different colors indicate ranges where more than 95% of the runs favored the green alternative over the red.

3.2. Whiteleg Shrimp. For all three impacts, the median related to the production of frozen peeled whiteleg shrimp was significantly larger for the Thai farms compared to the Vietnamese farms. Farming in low-level ponds in China was also related to lower median environmental impacts compared to farming in eastern Thailand. Chinese high- and low-level farms (Table 3), however, had similar global warming and eutrophication impacts, while low-level farms had lower freshwater ecotoxicity impacts. The environmental impacts of whiteleg shrimp farming in China were also similar to those of farming in Vietnam, while the allocation factor used greatly influenced results due to more extensive use of fishmeal from mixed fisheries and livestock byproducts in feeds. None of the impacts was significantly different when analyzing the entire distribution of differences between systems.

3.3. Giant River Prawn. Allocation also had a large influence on the outcomes of the Bangladeshi giant river prawn systems (Table 4). Farms where such prawn were polycultured with Asian tiger shrimp had more favorable median outcomes than prawn from Khulna province farmed without shrimp with regard to global warming and eutrophication, while the situation was the opposite in terms of freshwater ecotoxicity impacts. Distributions of differences did not differ among systems.

3.4. Tilapia. Among the Chinese tilapia systems, fillets from ponds in Guangdong were associated with significantly lower median impacts compared to fillets from Hainan (Table 5). The Hainan farms were also related to larger median eutrophication and ecotoxicity impacts than farms integrated with pigs and reservoir systems. Distributions of differences did not differ among systems.

3.5. Pangasius Catfish. All evaluated environmental median impacts caused by the production of pangasius catfish fillets were found to be significantly lower in the studied large-scale farms as compared to those calculated for small- and medium-scale farms. (Table 6). Small-scale farms also resulted in significantly lower median eutrophication impacts than medium-scale farms. Distributions of differences did not differ among systems.

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Environ. Sci. Technol. 2015, 49, 14176−14183

Table 3. Relative Environmental Performance of Whiteleg Shrimp Provided to European Consumers[a]

	global warming		eutrophication		ecotoxicology	
rank	mass	economic	mass	economic	mass	economic
best	CN HL a	VN I a	VN I a	VN I a	CN LL a	VN I a
	CN LL a	CN LL b	CN LL a	CN LL b	CN HL b	CN LL b
	VN I b	CN HL bc	CN HL a	CN HL b	VN I b	CN HL c
	TH S c	TH S bc	TH S b	TH S c	TH S c	TH S d
worst	TH E d	TH E c	TH E b	TH E d	TH E d	TH E d

[a]VN = Vietnam; TH = Thailand; CN = China; I = intensive; E = east; S = south; LL = low-level; HL = high-level. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 4. Relative Environmental Performance of Giant River Prawn Provided to European Consumers[a]

	global warming		eutrophication		ecotoxicology	
rank	mass	economic	mass	economic	mass	economic
best	BD B a	BD S&P a	BD S&P a	BD S&P a	BD B a	BD S&P a
	BD S&P a	BD B b	BD B b	BD K b	BD S&P b	BD B b
worst	BD K b	BD K b	BD K c	BD B c	BD K c	BD K b

[a]BD = Bangladesh; B = Bagerhat; K = Khulna; S&P = shrimp and prawn. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 5. Relative Environmental Performance of Tilapia Fillets Provided to European Consumers[a]

	global warming		eutrophication		ecotoxicology	
rank	mass	economic	mass	economic	mass	economic
best	CN GD a	CN GD a	CN GD a	CN GD a	CN GD a	CN GD a
	CN R b	CN R a	CN INT b	CN INT b	CN INT b	CN R a
	CN INT c	CN INT b	CN R c	CN R b	CN R b	CN INT b
worst	CN HI d	CN HI b	CN HI d	CN HI c	CN HI c	CN HI c

[a]CN = China; GD = Guangdong; HI = Hainan; I = integrated; R = reservoir. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 6. Relative Environmental Performance of Pangasius Catfish Fillets Provided to European Consumers[a]

	global warming		eutrophication		ecotoxicology	
rank	mass	economic	mass	economic	mass	economic
best	VN LG a	VN LG a	VN LG a	VN LG a	VN LG a	VN LG a
	VN SL b	VN SL b	VN SL b	VN SL b	VN SL b	VN SL b
worst	VN MD b	VN MD b	VN MD c	VN MD c	VN MD b	VN MD b

[a]VN = Vietnam; SL = small; MD = medium; LG = large. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

4. DISCUSSION

4.1. Analytical Approach. Unlike previous comparisons of point values, the current approach offered a level of confidence to support conclusions; and unlike previous comparisons of ranges,[11] consideration of only relative uncertainties reduced type II statistical errors (incorrectly accepting the null hypothesis). Of the systems tested, most differed significantly, despite the conservative Bonferroni correction.[33] This is largely due to the large sample size used (n = 1000), a sample size deemed as sufficient but not excessive. Historically, the number of MC iterations has been limited by computing power, and mathematical solutions for calculating the number of iterations needed to achieve a desired confidence level have even been proposed (so-called sequential stopping boundaries).[34] One could therefore argue that, by increasing the number of MC runs, any hypothesis test on means or medians will always produce significant results. This, by the way, not only is true for Monte Carlo but also is a danger of large real samples, and it is an inherent characteristic of classical hypothesis testing.[35] Using the alternative to significance tests showed that only the comparison of Asian tiger shrimp systems deviated in more than 95% of the MC runs in their environmental impacts.

From a naive point of view, the two statistical approaches give contradictory answers, but in reality they answer different questions. The more suitable of the two approaches therefore depends upon the question that needs answering: is the median of A significantly different from the median of B, or is a random pick of A demonstrably better than a random pick of B? Thus, while significance tests provide a conventional answer with respect to the median (or mean) impact, the proportional outcomes favoring a certain type of farming system might be more informative for a policy decision. In alternative words, statistical tests are about comparing distribution parameters, while the other approach is about a random pick from a distribution. While our belief is that operating within the paradigm of statistical hypotheses testing is too valuable to discard,[9] statistical significance should not always be taken at face

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Environ. Sci. Technol. 2015, 49, 14176−14183

value.[35−37] However, differences that are proclaimed to be "significant" should be supported by statistical tests.

4.2. Aquaculture Findings. Reflecting on previous aquaculture LCAs, many of the conclusions in the current research confirm the general outcomes of LCAs of fed aquaculture systems worldwide. Like tilapia and African catfish farming in Cameroon, eutrophication was mainly related to farm effluent,[38] and like most salmon farming, the provision of feed (including fisheries, agriculture, and livestock) was related to most greenhouse gas emissions[39] (see Figures S1−S3). Lowering the feed conversion ratio would consequently offer environmental improvements, where formulated feeds tailored to the nutritional needs of each species served in portions ensuring high availability (e.g., floating pellets) should be promoted. Reductions in aquaculture impacts, moreover, require agriculture to switch to less toxic pesticides or adopt organic farming practices to the extent possible. Developing models for reusing pond effluents and sediments locally as fertilizers, as already practiced in traditional Chinese aquaculture, would also reduce the impacts of both agri- and aquaculture, as nutrients in modern aquaculture systems are largely lost to adjacent water bodies where they result in eutrophication. Production systems with limited environmental interactions that allow for nutrients to be captured, and the influence by external parasites and bacterial diseases to be reduced (thus reducing the reliance on and discharge of therapeutants), should therefore also be favored.

Use of wild fish in aqua-feeds is one of the major critiques of the aquaculture sector, based on both environmental and socioeconomic arguments.[40,41] In the present research this also stood out as one of the major causes for global warming and eutrophication for many systems (see Figures S1 and S2). Limiting the inclusion and choosing more sustainable sources of fishmeal in feeds therefore need to be priorities for reducing the environmental impacts of farmed aquatic products, especially for shrimp. This goal can be achieved only if both feed producers and farmers, who often believe that larger fishmeal inclusions result in faster growth, recognize advancements in dietary substitution and supplements. A more sustainable source could be derived from processing byproducts, as many of these are still discarded (e.g., shrimp byproducts in Bangladesh). This would not only reduce pressure on wild fish stocks[41,42] but also reduce eutrophying emissions at landfills and recycle nutrients.[6] Finally, it is important to always favor feed ingredients, terrestrial or aquatic, that do not compete with their direct use as human food, as malnutrition still is widespread in some regions of Asia and elsewhere.

Intensity of systems had no clear correlation with the impacts evaluated in the present study. Paddle-wheel aerators were, however, more intensively used in ponds with higher stocking densities, with consequent global warming impacts. Monitoring oxygen levels in ponds could therefore help to optimize the use of paddle wheels, and more energy-efficient forms of aeration should be developed and promoted. The use of coal to generate the electricity that powers aerators and other activities also needs to be curbed or improved, as does the electricity efficiency of freezers.

On-farm chemical use made only small contributions to the overall life-cycle freshwater ecotoxicity impacts, with the exception of benzalkonium chloride and other chlorine-releasing compounds used as disinfectants. Chlorine is volatile and therefore used in large quantities, but the presence of organic matter leads to chlorinated compounds (e.g., halogenated hydrocharbons) that are more stable and induce long-term toxicity. The use of alternative, less toxic, biocidal or disinfection methods is therefore promoted.

4.3. Limitations and Future Research Needs. When chemical and other emissions are considered, it is important to acknowledge that LCA has limited capacity to account for spatiotemporal aspects in both LCI and life cycle impact assessment (LCIA) phases.[43,44] Thus, even if many of the local impacts related to the grow-out sites appeared not to exceed the buffering capacity of local ecosystems, they cannot be discounted as inconsequential. For example, with regards to therapeutant use in the present study, the peak predicted environmental concentrations for 61% of the treatments applied by grow-out farmers resulted in a risk quotient higher than 1, implying a potential risk to important structural end points of aquatic ecosystems not accounted for in the LCAs.[31] Similarly, for eutrophication, discharge of sediments and/or sludge from postharvested ponds could have severe ecological consequences through peaks in turbidity, oxygen depletion, or ammonia toxicity. Neither are additive and synergistic effects of different stressors accounted for in current LCA methodology, highlighting the added value of adopting the refined spatiotemporal windows and mixture toxicity approaches currently used in risk assessment alongside LCA.[31] A risk assessment approach could also provide better insights into other impacts that have been deemed as relevant for aquaculture LCAs,[45] such as reduced dissolved oxygen levels, introduction of nonindigenous species, and spread of disease and parasites.

The large dispersions around the characterization factors for freshwater ecotoxicity originated partially from ecotoxicological effect factors, with large discrepancies in experimental acute and chronic effect concentrations and within and among genus. Chronic effects on different types of algae often expressed the largest irregularities. Many additional assumptions exist around the chemical properties, some of which had to be resolved by use of QSARs. Given that these values are purely based upon the theoretical properties of molecules, QSAR estimates can differ greatly from reality.[37] Many other parameters related to inventory and impact assessment models also lack confidence estimates,[46,47] which in some cases were almost impossible to quantify.[48,49] For example, in the present research no uncertainty estimates were assigned to eutrophication potentials, as the uncertainty around the actual environmental consequences are hard to quantify given their complex nature and geographically specific context, with discrepancies induced by factors such as planktonic species assemblage, bioavailability of nutrients, fate of emissions, abiotic factors, and nutrient compositions in receiving environments.[50] More recent impact assessment methods that address these challenges by presenting country- or even region-specific characterization factors[51,52] can, in the meantime, induce new uncertainty in the form of unknown locations of emissions.

In addition to this, uncertainties also arise from the limited number of distributions available to represent data in LCA at present and the general negligence of covariance.[48] Still, these are only some of the many assumptions made over the different phases of an LCA, where quantitative uncertainty estimates remain incomplete or undefined, resulting in a fragile pyramid where the ranges of results capture only part of the underlying uncertainty. Significant differences thus consider only the dispersions quantified, confirming the strict relative meaning of comparative LCAs.[9] Other types of uncertainties, including several methodological choices, may also be more easily illustrated by performing sensitivity analyses[49] until more sophisticated approaches become available.[53,54]

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Environ. Sci. Technol. 2015, 49, 14176−14183

More extensive data on emissions related to LULUC are warranted, as the removal of mangrove for pond constructs is known to greatly influence both global warming and eutrophication results.[4] More inventory and characterization data related to freshwater ecotoxicity are also invited, as many emissions with possible environmental effects had to be excluded from the present study due to resource constraints. The inclusions of infrastructure, its maintenance, and waste disposal might, for example, alter the conclusions made related to freshwater ecotoxicity, as metals were a major driver for this impact category. Moreover, it is important to acknowledge that the data in the present research represents farming practices between 2010 and 2011, while aquaculture practices are notable for changing rapidly. For example, an outbreak of early mortality syndrome led to a rapid shift from Asian tiger shrimp to whiteleg shrimp for many Vietnamese farmers during the period of this research. Wild fish stocks, agricultural yields, and monetary values are also variable over time. More extensive databases and better software that allow for more rapid data processing and invite practitioners to utilize methodological advancements are therefore desired, in order to promote more scientifically robust conclusions in future LCA studies.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.5b04634.

Additional text, three figures, and 34 tables showing characteristic factors and chemical properties, comparative analyses, and contribution analysis (PDF)

Excel file with detailed contribution analyses for all systems and impact categories (ZIP)

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Notes

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SUSTAINABLE DEVELOPMENT

Lights and shadows in consequential LCA

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Abstract

Purpose Consequential LCA (CLCA) is becoming widely used in the scientific community as a modelling technique which describes the consequences of a decision. However, despite the increasing number of case studies published, a proper systematization of the approach has not yet been achieved. This paper investigates the methodological implications of CLCA and the extent to which the applications are in line with the theoretical dictates. Moreover, the predictive and explorative nature of CLCA is discussed, highlighting the role of scenario modelling in further structuring the methodology.

Methods An extensive literature review was performed, involving around 60 articles published over a period of approximately 18 years, and addressing both methodological issues and applications. The information was elaborated according to two main aspects: *what for* (questions and modes of LCA) and *what* (methodological implications of CLCA), with focus on the nature of modelling and on the identification of the affected processes.

Results and discussion The analysis points out that since the modelling principles of attributional LCA (ALCA) and CLCA are the same, what distinguishes the two modes of LCA is the choice of the processes to be included in the system (i.e. in CLCA, those that are affected by the market dynamics). However, the identification of those processes is often done inconsistently, using different arguments, which leads to different results. We suggest the use of scenario modelling as a way to support CLCA in providing a scientifically sound basis to model specific product-related futures with respect to technology development, market shift, and other variables.

Conclusions The CLCA is a sophisticated modelling technique that provides a way to assess the environmental consequences of an action/decision by including market mechanisms into the analysis. There is still room for improvements of the method and for further research, especially in relation to the following aspects: clarifying when and which market information is important and necessary; understanding the role of scenario modelling within CLCA; and developing a procedure to support the framing of questions to better link questions to models. Moreover, we suggest that the logic of mechanisms could be the reading guide for overcoming the dispute between ALCA and CLCA. Going further, this logic could also be extended, considering CLCA as an approach—rather than as a modelling principle with defined rules—to deepen LCA,

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providing the conceptual basis for including more mechanisms than just the market ones.

Keywords Affected processes · Ceteris paribus assumption · CLCA · Consequential LCA · Framing the question · Market mechanisms · Scenario modelling

1 Introduction

Consequential LCA (CLCA) is a modelling technique whose applications have boomed in the last 4 years in the scientific community (Fig. 1). Introduced in the 1990s (Weidema 1993), the topic has been elaborated mainly in the last decade (Curran et al. 2005; Ekvall 2002; Ekvall and Weidema 2004; Ekvall et al. 2005; Tillman 2000; Weidema 2003). An important reference in this field is the work by Ekvall (2002), according to which CLCA was defined as aiming "at describing the effects of changes within the life cycle" (p. 403), where "changes" to some parts of the life cycle inventory system led to a series of consequences through chains of cause–effect relationships (Curran et al. 2005). Subsequently, more complete definitions were formulated, according to which "the consequential approach to life cycle inventory attempts to estimate how flows to and from the environment will change as a result of different potential decisions" (Curran et al. 2005, p. 856). On the same line, CLCA was defined as aiming at "describing how the environmentally relevant physical flows to and from the technological system will change in response to possible changes in the life cycle" (Ekvall and Weidema 2004, p. 161). Many other definitions were given, enriched over the years to highlight the market-oriented nature of the model. In fact, the term "marginal and market-oriented" was used by Nielsen and co-workers (Nielsen et al. 2007; Nielsen et al. 2008; Nielsen and Hoier 2009) to denote an approach in which "environmental profiles are compiled by addressing changes induced by a change in demand for the company's products, and co-product issues are handled by system expansion" (Nielsen et al. 2007, p. 433). Schmidt and Weidema (2008) and Schmidt (2010) further refined the definition, introducing the concept of a consequential approach to system delimitation as that in which "the actual affected suppliers and technologies are modelled instead of averaged. In addition, co-product allocation is avoided by system expansion" (Schmidt 2010, p. 183).

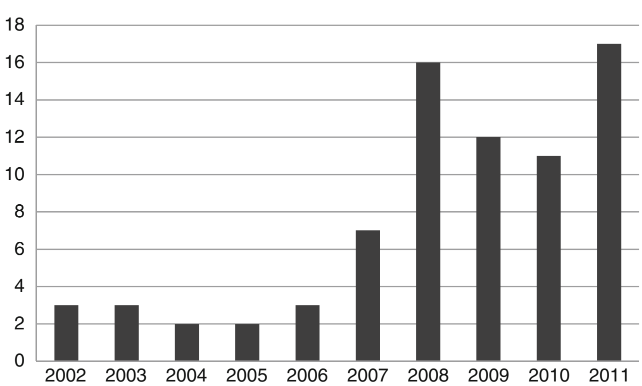

Fig. 1 Histogram of the number of articles published from 2002 to 2011 which contain the word "consequential" AND "LCA" in their abstract, title or keywords. The *diagram* shows in particular an increase in the number of publications on CLCA from 2007 on

Each definition built on a previous one, trying to add new elements to clarify the concept at a methodological level, and preparing the road for case studies. These started almost in parallel with the methodological development, but gained their momentum around the middle of what Guinée et al. (2011) defined as the decade of elaboration in LCA (2000–2010).[1] CLCA has been applied to a wide range of products (e.g. Astrup and Fruergaard 2011; Geyer 2008; Kimming et al. 2011a, 2011b; Nguyen et al. 2010; Schmidt et al. 2007; Thrane 2006), and also outside LCA, with the introduction of the concept of consequential social life cycle assessment[2] (Jörgensen et al. 2010). A clear example of a CLCA study is provided by Ekvall and Andrae (2006) who analyse what would happen if, after the ban on lead in solder pastes used in the electronics industry, the production would shift from SnPb to Pb-free solder. The consequences analysed refer to the fact that the solder shift increases electricity use, especially in the solder application. This means that less electricity will be available for other purposes. These dynamics require the use of partial equilibrium models to be solved, and thus the information about the sensitivity of the supply and demand of lead and scrap lead to price fluctuations.

There are also plenty of examples of CLCA applied to biofuels (e.g. Hedegaard et al. 2008; Melamu and Blottnitz 2011; Reinhard and Zah 2009, 2011; Silalertruksa et al. 2009). In that context, CLCA was used to address problems like the environmental consequences of the production of second-generation biofuels compared to current palm oil biodiesel production (Lim and Lee 2011), and to investigate the likely indirect effects of the development of a grass biomethane industry in Ireland, such as a reduction in beef exports to the UK (Smyth and Murphy 2011). Indeed, the debate on biofuels and on other bio-based products contributed to the speeding up of the methodology, particularly in relation to the effects of land use change. The seminal study by Searchinger et al. (2008)

[1] The decade of elaboration makes reference to a period characterised by the development of multiple different approaches, ranging from dynamic LCA to spatially differentiated LCA, and environmentally extended input–output based LCA, to mention just a few.

[2] A consequential social life cycle assessment would take into account not only social impacts on the stakeholders in the life cycle, but also those related to the non-implemented product life cycle (i.e. when not carrying out a process or using a product).

found that most of the previous LCA studies provided only a limited answer to the problem. In fact, by excluding emissions from land use change, they failed to account for the indirect effects (i.e. those taking place outside the biofuel value chain). Indirect effects, which may be quite relevant, are those that result, for example, from the competition between food and fuel (the land currently used for food can be allocated to fuel production), or from the competition for limited biomass in general, including competition between alternative energy uses for biomass (Hedegaard et al. 2008). Thus, replacement/displacement mechanisms started to be analysed and approaches to include them in LCA have been proposed.

Considering the increasing interest in CLCA within and outside the LCA community, and the potential of the approach, we have analysed the methodological implications of CLCA. Our research questions were: what is CLCA exactly? For what is it useful? How is CLCA perceived and used among practitioners? To what extent have the applications carried out so far been in line with the theoretical dictates, in particular, with reference to the way in which affected processes are identified and multi-functionality is dealt with?

The analysis builds upon previous reviews (Earles and Halog 2011; Finnveden et al. 2009; Reap et al. 2008; Zamagni et al. 2008), further broadened and updated, and addresses both methodological papers (22 articles) and applications (38 articles) that have been published over a period of 18 years. While the details of the literature review are available in the Electronic Supplementary Material, this article discusses two key issues derived from the review: "what for" and "what". In other words, for which applications has CLCA been used (and thus, which questions have been answered in the scientific literature by CLCA), and what is CLCA? A direct comparison between CLCA and attributional LCA (ALCA) is out of the scope of this article, as this issue has been widely debated in the scientific community (see for example Ekvall et al. 2005) and such a discussion would just perpetuate the myth that only one way to conduct LCA is correct. The analysis conducted in this paper is centred on CLCA.

Starting from the debate occurred in the past and elaborating on it, Section 2 focuses on the "what for" ("Questions and modes of LCA"), while Section 3 discusses the "what" ("Methodological content of CLCA"), with focus on the nature of modelling and on the identification of affected processes. Finally, a discussion and structure of the main findings is provided in Section 4, and conclusions are summarized in Section 5, where recommendations for further research are also provided.

2 "What for"—questions and modes of LCA

Which type of life cycle inventory modelling to adopt and for which purpose is a question that has been debated for 20 years (Tillman 2000) but the dispute is not yet solved. This is a critical aspect, because a correct application of the method is strictly related to the identification of the problems for which it provides the best answer(s), as also pointed out by several authors (Frischknecht and Stucki 2010; Guinée et al. 2002; Tillman 2000), and recently in other fields, such as Risk Assessment (Kapustka et al. 2010). However, its importance continues to be neglected.

Different modes or types of LCA have been identified, which represent attempts to capture the notion that LCA can be conducted in a variety of ways. Guinée et al. (2002) introduced two modes of LCA, descriptive and change-oriented (which echo attributional and consequential), and three types of questions, related to three main types of decisions: occasional (one-off fulfilment of a function), structural (related to a function regularly supplied), and strategic (related to function supplied for a long or even indefinite period). Four decision-context situations are identified in the ILCD Handbook (EC JRC-IES 2010), namely micro-, meso/macro-levels, and two modes, ALCA and CLCA. Their application is guided by specific LCI method approaches to allocation or system expansion/substitution. Together with ALCA and CLCA, Frischknecht and Stucki (2010) introduced decisional LCA (DLCA), based on future actual or anticipated economic and/or contractual relation. Moreover, the authors also propose the (economic) size of the object of investigation as a criterion for choosing among the different modes of LCA.

On top of this, Guinée and Heijungs (2011) recently suggested that another mode of LCA might be relevant to explore: back-casting LCA (BLCA), a scenario-based way to model specific product systems to normative future targets. In CLCA, global consequences in terms of CO_2, land, water, and resource uses, among others, are all modelled from a single product's perspective. Indeed, although CLCA is particularly suitable for mapping the impacts of processes indirectly affected by a decision, simply summing up all single product CLCAs does not necessarily result in sensible estimations of global consequences. Guinée and Heijungs (2011) therefore argue that we may consider exploring ways to back-cast normative targets for CO_2, land, water, and resource uses to global scenarios for agriculture, energy production, and transport (cfr. Graedel and van der Voet 2010), and to relevant LCA scenarios (e.g. from the global 2050 IPCC target of 450 ppm CO_2-eq. to global energy and transport scenarios—being the highest contributors—to single product's consequences).

However, the distinction between A (attributional), B (back-casting), C (consequential) and D (decisional) LCA does not seem sufficient for properly supporting practitioners in understanding which questions could be better answered by which mode of LCA. Over the last 15 years, many authors have been discussing the knowledge generated by LCA, especially with reference to CLCA and ALCA. Ekvall et al. (2005), and Sandén and Kalström (2007) state that both ALCA and CLCA can be applied for modelling future, past, or current systems. Others (e.g. Lundie et al. 2008; Tillman 2000) recommend the use of CLCA for decision-making, while others consider CLCA as the method able to generate the most relevant information, independently of the application. The reason is that LCA is considered interesting and relevant only if it affects decisions and, in turn, a rational decision-making requires information about the consequences of the decision (Wenzel 1998). Along the same line of thought, Weidema (2003) recognises the relevance of CLCA in most of the decision-context situations, with the exception of a few cases (e.g. studies on environmental taxation) in which the ALCA could be considered more appropriate. Moreover, he considers ALCA relevant also when no specific decision is at hand for increasing the understanding of the causal relations within the product chain, and between this chain and the surrounding technological system. Finally, Lundie et al. (2008) suggest avoiding CLCA when the difference between the CLCA and ALCA results is small, and when the uncertainties in modelling outweigh the insights gained from it. This suggestion is questionable since the difference in results and the degree of uncertainty are not generally known until both methods are applied.

Overall, what emerges from the discussion above is that there is not a right or wrong mode in which LCA can be carried out, both ALCA and CLCA have utility. The difference consists in how boundaries are defined, which stems from a clear and unambiguous definition of the goal of the study. In fact, as already recognised by other authors (e.g. Curran et al. 2005; Ekvall et al. 2005; Tillman 2000), confusion arises in understanding the usefulness of CLCA because the questions to be answered are not formulated in a clear way.

Usually, the purpose of the study is defined, but this is not enough to decide whether or not to use CLCA. Table 1 and Table 2 report some examples of questions formulated in the literature for which a CLCA modelling was applied, together with the identification of the consequences addressed and the target audience of the study.

The examples in Table 1 refer mainly to case studies carried out in the "LCA context", i.e. either published in journals in which LCA has a prominent role or conducted by practitioners with notable experience in LCA.

Simply stating that the purpose is to evaluate the potential environmental impacts associated with the production of X (e.g. Gamage et al. 2008; Lesage et al. 2007b; Nielsen et al. 2007; Nielsen and Hoier 2009), or to investigate whether a technology is a more environmentally sound alternative than a conventional way of producing a particular product (e.g. Skals et al. 2008) still leaves the modelling choices open to a number of interpretations.[3] Moreover, the concept of consequences is never mentioned in the purpose of the study and the authors seldom explain why a consequential modelling has been selected instead of an attributional or decisional one. A reason could be that in many cases the application of CLCA was just the purpose and not the means of the study. An explanation for that could be that since CLCA is a new modelling approach, most applications have focused on testing the method in order to understand how it works and whether it gives different results compared to ALCA (e.g. Ekvall and Andrae 2006; Gaudreault et al. 2010; Thomassen et al. 2008).

Table 2 focuses on examples concerning biofuels and it shows a different situation with respect to Table 1. In fact, the questions of the studies are formulated in a clearer way, and refer mainly to research problems analysing decisions which have consequences at the macro level (whole economy or country level). It seems that the consequential approach is better understood when applied at a higher level of the analysis than the product one, like the examples in Table 1. This would be in line with the provisions of the ILCD Handbook, according to which a consequential modelling is suggested when large-scale consequences are involved. The typical decision context is that of policy development/information, in which the decision and the related changes "will affect the rest of the economy by having large-scale structural effects" (EC JRC-IES 2010, p.40).

Reinhard and Zah (2009) for example examine the consequences expected in Switzerland if 1 % of the current diesel consumption were replaced with imports of soybean methyl ester or palm methyl ester. Silalertruska et al. (2009) analyse the effects that would occur at the global level if cassava demand increased in Thailand, while Smyth and Murphy focus on the effect at country level. In both Table 1 and Table 2, the target audience is not always made clear, even if in most cases it can be easily identified. This is another aspect that hampers the correct identification of the most appropriate modelling approach to the research questions at hand. In fact, a complete and detailed definition of the decision context is fundamental in determining the most appropriate methods for the modelling, besides affecting also other key aspects of the scope definition (EC JRC-IES 2010).

This lack of clarity about CLCA is also due to the ambiguous interpretation of the definition and the concept given by practitioners. Starting from the most quoted definitions (Curran et al. 2005; Ekvall and Weidema 2004;

[3] It is worthy to note that stating the purpose of the assessment is perfectly in line with what required by the ISO standard in the goal and scope phase (ISO 2006). However, this proved not to be sufficient to properly address and model the problem.

Table 1 Examples of questions addressed by means of CLCA

Reference	Subject	Question addressed/purpose of the study	Consequence(s) addressed	Target audience
Dalgaard et al. (2008)	Soybean meal	To establish a reliable representation of soybean meal production for use in LCAs of European livestock production chains To estimate the environmental consequences of soybean meal consumption using a consequential LCA approach To identify the environmental hotspots in the product chain of soybean meal	The increased production of soybean meal affects the palm oil and rapeseed oil production, respectively (soybean-rapeseed loop)[a]	Not explicitly addressed. The results have two main potential users: Scientific community (availability of data about for LCA on livestock products) Decision makers at national level, since the information of the study can be used to set policies for the sector
Gamage et al. (2008)	Furniture (chair)	To determine the environmental hotspots in the life cycle of two chairs To compare the life cycle impacts of the two chairs To compare alternative potential waste management scenarios	None. The study is indeed an attributional one, even if the authors classify it as consequential	Producers and designers working in the furniture sectors
Gaudreault et al. (2010)	Pulp and paper mill	To compare the information provided by ALCA and CLCA approaches for decision-making regarding the selection of process options aiming at reducing the dependency of an integrated newsprint mill to purchased electrical power	Consequences identified for each of the four options considered (increase in cogeneration capacity and of the DIP [b] content of the paper) Reduction of recycled pulp production in other systems due to an increased use of recycled paper is compensated by an increase in a mixture of virgin kraft and TMP[c] pulp New consumption of virgin fibre results in additional extracted material Decreased use of wood chips (due to the implementation of the four options) leads to less material to be disposed of Reduction of electricity consumption (due to the implementation of the options): coal fuelled power plant is identified as marginal technology for electricity	North America pulp and paper mill managers
Lesage et al. (2007a, b)	Brownfield rehabilitation	To evaluate the potential environmental impact associated with a brownfield rehabilitation project aiming at residential redevelopment	Three categories of consequences are analysed: Primary: changes in the site's environmental quality Secondary: changes due to the rehabilitation service system (supply of housing services)	Not explicitly defined. However, the optimal target audience is represented by decision makers operating at a territorial level (those who are in charge of the management of brownfield)

Table 1 (continued)

Reference	Subject	Question addressed/purpose of the study	Consequence(s) addressed	Target audience
			Tertiary: effects of the reoccupation of the site on the life cycle of other regional sites (inclusion in the analysis of other site occupations marginally affected by the increase in supply of housing services)	
Nielsen et al. (2007)	Enzyme products	To address the environmental impact potentials associated with enzyme production in a cradle-to-gate perspective	Displacement of alternative sources of N and P fertilisers as a result of the enzyme application in agriculture Starch applied in enzyme production has protein (gluten) as co-product. It is assumed that the marginal protein displaces other types of protein for animal feed Natural gas fired power plant is the marginal source of electricity identified in the study	The company that produces the product investigated in the study
Nielsen and Hoier (2009)	Mozzarella cheese production	To assess the environmental impacts that come with the use of industrial phospholipase in mozzarella production and compare these with the savings that come with the avoided milk production	Increased demand for vegetable oil (palm oil) as a result of the reduced output of fat from the mozzarella production of the edible fat market Increased demand for alternative protein sources in animal breeding (soybean meal) as a result of the reduced protein output from the cheese factory	The company that produces the product investigated in the study
Skals et al. (2008)	Enzymes	To investigate whether the enzyme technology is a more environmentally sound alternative than the conventional ways of producing paper	Not discussed in detail. Reference is made to the inclusion of changes occurred when enzymatic solutions displace conventional solutions but evidence is not given. Only the following aspects have been included: Natural gas and coal were selected as marginal sources of electricity	The company that produces the product investigated in the study
Thomassen et al. (2008)	Milk	To demonstrate and compare the ALCA and CLCA of an average conventional milk production system in the Netherlands	Natural gas power plant is identified as the marginal source of electricity Soybean meal was identified as the marginal fodder protein (being the feed ingredient that will meet the increased protein demand due to increased milk production)	Policy makers

[a] Soybean oil is the co-product of soybean meal. The avoided production of other vegetable oils (rapeseed, palm), caused by the production of soybean oil, is included. Because vegetable oil is co-produced with protein, this introduces another need for system expansion (which again includes a co-production of protein).

[b] DIP is the short for deinked pulp. It is recycled paper which has been processed by chemicals, thus removing printing inks and other unwanted elements and freeing the paper fibres.

[c] TMP stands for thermomechanical pulping

Table 2 Application of CLCA to case studies about biofuels/bioenergy

Reference	Subject	Question addressed/purpose of the study	Consequence(s) addressed	Target audience
Lemoine et al. (2010)	Bioenergy	To examine the relative climatic merits of combusting corn grain or switchgrass for electricity generation versus refining corn grain or switchgrass	Detailed discussion about the displaced source of electricity when an increase in bioelectricity occur. Parameters analysed: type of electricity replaced and structure of the regional electricity market	Policy makers
		To produce liquid fuels for use in the current vehicle fleet		
Lim and Lee (2011)	Palm oil biofuel	To determine the environmental consequences of the inclusion of second-generation biofuels (bioethanol from palm oil biomass) towards current palm oil biodiesel production	Not clearly identified and discussed Increased use of inorganic fertilisers to replace the removal of palm oil fronds Replacement of palm oil fibre and shell by fossil fuel as team-boiler fuel	Not clearly defined
Reinhard and Zah (2009)	Cassava	To assess the direct and indirect environmental impacts to be expected if Switzerland should replace 1% of its current diesel consumption with imports of soybean methyl ester (SME) from Brazil, or palm methyl ester (PME) from Malaysia	Several mechanisms and consequences taken into account: The effects of an increased demand for a specific crop on the agricultural stage are assumed to be met by expansion The identification of the increased production of the marginal vegetable oil on the world market when the use of soybean oil for biofuel production increases The identification of the substituted for the protein source (for animal fodder) when palm kernel meal increases as a consequence of increased production of palm oil for biofuels	Not specified, but it is assumed to be represented by policy makers and implementers of biofuels policy
Silalertruksa et al. (2009)	Cassava	To identify the environmental consequences of possible (future) changes in agricultural production systems and determine their effects on land use change and GHG implications when cassava demand in Thailand increases	Changes in the region where energy crop demand increased: Consequences due to the conversion of unoccupied land to cropland Increased production by yield improvements in the country Displacement of cultivated area of other crop (sugarcane) in the country and reduced sugar production is compensated by yield improve or increased cultivation in other countries Changes in other regions:	Policy makers and implementers of biofuels policy

Table 2 (continued)

Reference	Subject	Question addressed/purpose of the study	Consequence(s) addressed	Target audience
			Increased demand is met by import from other countries Reduced sugar products in the market are compensated by increased production in another country	
Smyth and Murphy (2011)	Grass biomethane	Identification of the likely indirect effects of a grass biomethane industry in Ireland as a reduction in beef exports to UK	A detailed analysis of consequences is provided: UK demand for beef is unchanged and reduction in Irish import is met with an increased UK beef production Unchanged UK beef demand but increase in beef imports from other countries to meet the demand Decline in UK consumption met by the consumption of poultry	Not discussed, but the target is represented byPolicy makers and implementers of biofuels policy

Examples of questions addressed and consequences analysed

Tillman 2000; Weidema 2003; Weidema et al. 2009), some authors further elaborated on the concept, giving interpretations that do not always reflect the expressed intentions of the primary authors. Some authors associate the concept of consequential only to the way in which multi-functionality is dealt with (Gamage et al. 2008; Gaudreault et al. 2010), and thus system expansion is considered a synonym of CLCA. One may refer to these studies as semi-CLCA (cf. Schmidt 2010). In a similar way, others make reference to a consequential approach to system delimitation (Lim and Lee 2011), but the study they carried out is actually an attributional one, in which no consequences are addressed. Overall, interpretations on CLCA range from considering it as the state-of-the-art methodology, to just a complication, and finally, to a model that avoids allocation by means of system expansion.

These misalignments point out that the choice of CLCA versus other modes of LCA is not always done consciously, and the consequences are propagated at the level of modelling, as described in Section 3. This in turn reaffirms the fundamental importance of the goal and scope phase in unambiguously defining what the LCA study is about and for whom it is intended. In fact, the depth and the breadth of LCA can differ considerably depending on the goal and scope of a particular LCA, and errors made in this phase may have strong consequences for the results (Fullana et al. 2011).

3 "What"—methodological content of CLCA

After the analysis of the CLCA applications, the review focused on the modelling technique. The purpose was to identify the main characteristics of the approach, possible shortcomings in the current applications, and areas for improvements. We started out by examining the nature of modelling and then focused our attention on one of the most characteristic aspects of CLCA, namely the choice of affected processes.

3.1 The nature of modelling

Discussing the nature of modelling means analysing principles, analytical techniques used and their limitations, all characteristics that define what the model can (and cannot) do, and thus contribute to the identification of how to solve the problem at hand. To our knowledge, these aspects have not been dealt with in detail in the literature, with the exception of Weidema et al. (2009, p. 7) who define CLCA as a "steady-state, linear, homogeneous model, with each unit process fixed at a specific point in time". From this point of view, no differences exist with respect to ALCA. The application of CLCA is not time-related, as already discussed in previous publications (e.g. Curran et al. 2005; EC JRC-IES 2010; Sandén and Karlström 2007) and thus both the adjective “retrospective” and “prospective” apply. Time is a parameter outside the models in both cases, but this simplification seems to be less acceptable, at least at a conceptual level for CLCA. In fact, the term “consequential” suggests that, since the primary object of the model is the analysis of consequences, there is a sequence in time of events that gives rise to a propagation effect, according to which the consequence at time t_1 is different from that at time t_2.

However, in CLCA we are interested in two main aspects from the temporal point of view: the end time of consequences (t_1, t_2, …, t_n) and the storyline on how to arrive at that point. Then, CLCA is defined for a given point in time, and at that point, the changes occurred are modelled in a steady-state way, using the information given by the storyline.

According to Weidema et al. (1999), this is still done under the ceteris paribus assumption (CPA), because the changes analysed are usually small compared to the production in society, which can be assumed to be unaffected.

Considering that ALCA and CLCA share the same modelling principles, what distinguishes CLCA from ALCA? We tried to answer this question by making reference to the general modelling principles discussed by Heijungs et al. (2007), namely the distinction between a system and its environment (*where boundaries are set*), the internal structure of a system in terms of its components (*unit processes and environmental compartments*), the relationships among the components, and the relationship between the system and its environment (*open* versus *closed* versus *isolated system*). With respect to the principles introduced above, the main distinctive characteristics of CLCA compared to ALCA are (Weidema 2003):

(a) Processes are included to the extent of their expected change caused by a demand (affected processes).
(b) Co-products are handled by system expansion.

Regarding the first aspect, the inclusion of affected processes represents an attempt to include market mechanisms into the analysis. In fact, processes are selected on the basis of considerations on how the market of the system investigated might look under the hypothesis of the change analysed, introducing insights into the future. We should be aware that these “future” considerations are not a peculiarity of CLCA but can be dealt with also in other LCA modes, making the assessment prospective (Spielmann et al. 2005), or elaborating scenarios (Sandén and Karlström 2007), trying to understand what the situation in the future will look like, with or without taking the change in demand into account. However, since the main difference between CLCA and other LCAs relies on the choice of processes to be included in the system, their representativeness and relevance with respect to the changes that occur is a daunting question. Presently, the underlying assumption in CLCA studies is that technological data are the same: some new/future technologies are assumed, but current processes are used to represent them. This assumption can be less relevant for short-term changes, but when long-term changes are involved, data projections are probably needed.

As far as the other distinctive characteristic of CLCA is concerned, we did not analyse how system expansion has been applied in the several case studies, since the question of multi-functionality has been the object of several publications. We would like to point out only the connection between the boundary expansion and how the functional unit (FU) is dealt with in CLCA. According to the several definitions provided in the literature in CLCA, the boundaries of the system are expanded so as to include those processes which are affected by the consequences of the decision at hand. In doing so, the resulting FU of the whole system would consist of multiple functions, including the main system and those added by the processes included in the boundaries. When a comparative analysis has to be conducted, it might be difficult to guarantee the functional equivalency between the systems compared, since the processes included in the two situations might serve different functions. Moreover, such a resulting *multi* FU raises some concerns about whether it can still be considered a FU. The question is not addressed in the literature, with the exception of Smyth and Murphy (2009) in relation to the comparability problem due to the different boundary conditions. Even Weidema et al. (2009) state that when small changes and consequences are at the core of CLCA, further requirements on FU are not necessary. Moreover, the authors pointed out the relevance of the FU only in terms of “quantity” (it should “reflect the extent of the consequences of the decisions studied”, when decisions “involve the entire market of a major product or process”, as stated by Weidema et al. (2009), p. 22) and not “number of functions”.

How to deal with a *multi* FU and how this affects modelling is an aspect which requires more understanding and research, which could also help in the development of CLCA modelling.

3.2 Identification of affected processes: where do we set boundaries?

As already pointed out above, one of the main issues of CLCA is the identification of the processes to be included in the analysed system, which implies the way in which boundaries are set. In this regard, the methodology is clear in stating that only (but all relevant) affected processes need to be included, defined as those that respond to changes in demand and/or supply driven by the decision at hand. However, a decision can affect processes through a wide range of mechanisms, which cause different consequences. For example, a change in demand and/or supply may influence prices that determine what is produced (substitution mechanisms) and who can afford to consume it (income effects). Price changes in turn affect income to an extent that depends on how large the cost of the item is relative to the consumer's budget. Rebound effects or ripple effects, as defined by Hertwich (2005), might then occur when, for example, the increased real income due to the reduced price of a good causes consumers to increase their demand for other goods (see also Girod et al. 2010). The chain of consequences that can be analysed does not seem to have an end. However, not all of these consequences are presently taken into account in CLCA (for example, a few approaches to rebound effects still exist—see Thiesen et al. 2008), and simplifications are adopted, for example, in relation to the number of markets dealt with simultaneously, to the scale of the consequences,[4] or to the complexity of substitution mechanisms, as can be seen in the examples of Tables 1 and 2. In fact, usually only one market situation is modelled and only one affected process/technology is identified. Other approximations refer to the investigation of a stand-alone increase in demand, assuming that substitution occurs within the same type of products (Schmidt 2008). This is clearly a limitation, because products and markets are connected, and an increased supply of some products implies increased demand for upstream intermediate products (Lundie et al. 2008). In some cases, as extensively discussed by Earles and Halog (2011), attempts to link several markets simultaneously have been made using partial equilibrium models (PEM) or general equilibrium models (e.g. Dandres et al. 2011; Ekvall and Andrae 2006; Kløverpris et al. 2008; Kløverpris et al. 2010; Lesage et al. 2007a, b). The studies carried out demonstrate that the use of PEM is limited by the knowledge of the price elasticities, adding great uncertainty to the model. There is also the problem that PEM can be modelled only for a few markets, and the decision regarding which markets to include needs accumulated experience. Moreover, PEMs ignore interactions between markets, while in real world, substitution involves cross-elasticities, being the change in demand/supply for one good in response to a change in the price of another good.

A procedure to support the identification of the affected processes, which had been developed since 1999 (Weidema et al. 1999), was updated and refined in 2009 (Weidema et al. 2009). That procedure consists of five steps, during which the following elements are analysed: (a) the time horizon of the study; (b) the analysis of the extent to which the changes in production volume only affect specific processes or a market; (c) the trend in the volume of the affected market; (d) the analysis of the extent to which the technology has the potential to provide the required capacity adjustment; (e) the analysis of whether the identified technology is the most/least preferred. The procedure is schematised into a decision tree which guides the practitioners throughout all the steps.

Moreover, recently Schmidt (2008) also developed a procedure for system delimitation in agricultural CLCA. However, the identification of affected processes in case studies (e.g. Lemoine et al. 2010; Nielsen et al. 2007; Skals et al. 2008) is still often done without adopting the procedures developed, using various arguments not always supported by the evidence of market information. Moreover, even when the procedure is applied, evidence is not often given in the papers. Some authors simply use the results of previous studies without providing arguments in support. For example, in several studies involving vegetable oils (e.g. Dalgaard et al. 2008; Nguyen et al. 2010; Nielsen et al. 2008; Nielsen and Hoier 2009; Thomassen et al. 2008), the results obtained by Schmidt and Weidema (2008) have largely been adopted, without justifying the full consistency with and appropriateness for the specific case at hand. This attitude has been pointed out recently by Reinhard (2011) who stressed how results cannot be generalised or used for similar or relating goals and scope. In other cases, dynamic optimising models have been applied (Eriksson et al. 2007; Mattsson et al. 2003), together with energy system analysis simulation tools (Lund et al. 2010; Mathiesen et al. 2009), which demonstrate that a set of affected technologies, instead of just one, can provide a more complete description of the consequences.

Mathiesen et al. (2009) tested the ability of identifying marginal electricity technologies in CLCA by analysing the statistics of historical developments of the energy system. The results show a discrepancy between the energy technologies identified as marginal in the LCA studies and the actual marginal electricity technologies. The authors demonstrated that marginal energy technologies are identified inconsistently, using different arguments, and not always mentioning market trends.

These discrepancies between the actual and the foreseen marginal energy technologies, which could probably be identified also for other technologies, are partly unavoidable when modelling the future, but might also be partly due to erroneous assumptions and simplifications adopted in modelling the market consequences. Moreover, it is not clear if the wrong identification of the affected processes should be ascribed to an inappropriate application of Weidema's recommended procedure or to the individual choices made by

[4] Typically the scale of the potential changes is small (for non-marginal variations see the work of Dandres et al. 2011), which means the direction of the trend in market volume and the constraints on and production costs of involved products and technologies are not affected (Weidema et al. 2009).

practitioners, seldom transparently documented. There is a high degree of uncertainty and a wide range of possible results, depending on the system enlargements (and thus on the affected processes taken into account), on the type of indirect effects included, and on the assumptions and scenarios made (Eriksson et al. 2007; Nielsen and Hoier 2009; Reinhard and Zah 2009; Schmidt 2010; Smyth and Murphy 2011). Thus, where boundaries of the system should be properly set is a tricky question. Without making the definition of the system boundary a rigid procedure but flexible enough to account for the potential impacts of a change in production demand, the need exist to define procedures and rules to increase the robustness of the studies. In this regard, sensitivity analysis and the use of scenarios are unavoidable (Mathiesen et al. 2009). The two techniques together can contribute to considering how the consequences might change under several market situations, taking into account relevant parameters to calculate substitution, the possible marginal products on the world market, and the feedback mechanisms (Reinhard and Zah 2009).

4 Discussion

In the previous sections, it has been clarified that the introduction of (some) market mechanisms in the analysis, throughout the inclusion of the affected processes in the product system, is what primarily distinguishes CLCA from other modes of LCA.[5] The logic behind this is that all products are included in markets, with price mechanisms, among other factors, regulating their production, development, and consumption. Thus, the common supply and demand mechanisms introduce perturbations in the system that reacts accordingly, giving rise to a chain of cause–effect relationships. These market relations or mechanisms are presently not endogenised in the model (Lundie et al. 2008), but are derived from economic models (e.g. Dalgaard et al. 2008; Eriksson et al. 2007; Lemoine et al. 2010; Lesage et al. 2007a, b; Lund et al. 2010; Pehnt et al. 2008) or outlooks in specific sectors (e.g. Schmidt 2010; Schmidt and Weidema 2008), and then included as input into LCA.

One could argue that even market mechanisms are not a key characteristic of CLCA because the substitution or avoided-burden method for handling multi-functionality already adopts inclusion of market mechanisms as basis for “avoiding” allocation in ALCA, and prices can be the basis of allocation in ALCA. However, the way in which multi-functionality is handled is just one component of CLCA (semi-CLCA), which also entails the methodological implications discussed above.

The inclusion of market mechanisms and information into LCA has been strongly criticised in a recent paper by Pelletier and Tyedmers (2011), who consider market signals inadequate in managing the environmental dimensions of activities. With reference to CLCA, they state that since the purpose of CLCA is “to model impacts that result as a consequence of decisions, as mediated through biophysically myopic market transactions, the utility of resultant research outcomes is similarly questionable” (Pelletier and Tyedmers 2011, p. 9). However, this conservative position neglects the economic and ecological interdependencies, which have already been identified as one of the causes of policy failure in both environmental and economic development (Bartelmus 2008). If from the one side we could agree on market aspects being inadequate to catch environmental issues, on the other side we have to recognise that if we want the foreseen system changes to be reliable, we should base them on mechanisms of actual markets. Moreover, the inclusion of market mechanism would add realism (i.e. accuracy) to the analysis, which is the ultimate goal of any model. Clearly, the system complexity will be higher because CLCA includes additional economic concepts such as marginal production costs, elasticity of supply and demand, and so forth. Some models used in the analysis are also much less transparent than the linear and static/steady-state model of a traditional LCA. Their results can also be very sensitive to assumptions. This all adds to the risk that inadequate assumptions or other errors significantly affect the final results (Ekvall and Andrae 2006; Gaudreault et al. 2010; Reinhard and Zah 2009; Schmidt 2010; Smyth and Murphy 2011). However, the complexity and uncertainty, on the one side, can be balanced with the dramatic increase in knowledge about the system on the other (Lesage et al. 2007b; Schmidt 2010; Silalertruska et al. 2009), a compromise that can be accepted if the analysis is transparent in the assumptions and choices made regarding the different consequences considered.

The logic of mechanisms could be the reading guide for better understanding and further developing CLCA. The modelling specification in terms of which market mechanisms are included in the analysis can provide better indications than the term “consequential” itself.

Going further, this logic could be extended, considering CLCA and the other modes which address changes (BLCA, DLCA) as approaches—and not modelling principles with defined rules—to deepen LCA, to include mechanisms that result from an initial change to some parts of the product system. Which mechanisms to include is a tricky question, since they can show up everywhere, involving a variety of domains. Market mechanisms are part of broader economic mechanisms, which recall concepts like employment and

[5] Presently LCA includes only technological and environmental relations in the inventory and impact assessment phase, respectively (Heijungs et al. 2010).

growth. These in turn function within a cultural, social, political and regulatory context. This process of adding mechanisms would require that those constraints typically assumed as fixed entities[6] under the present modelling are taken into considerations, relaxing the ceteris paribus assumption related to both temporal and causal aspects.[7]

Specifying which constraints to impose and why, which to relax (e.g. change in functional unit, change in technology, constrained capacity, market reactions, production volume developments, etc.), whether all together or just one by one, and which (market) mechanisms to include and how to report them, would support the introduction of more mechanisms into the analysis and contribute to making the assessment more consistent and robust.

Someone could argue that the distinction among the different modes of LCA is in contrast with the suggestion of considering CLCA as an approach rather than a modelling technique. In fact, if a practitioner is asking the right question, the resulting LCA should be designed to answer that question, regardless of how we call it (ALCA, CLCA, DLCA, BLCA). However, CLCA at this stage of development still does not have all the capabilities for addressing all the mechanisms, and thus it distinguishes itself mainly for being a modelling technique with defined rules. In this context, it is important to provide practitioners with detailed guidance on a number of issues, including: how to select the most appropriate data; how to deal with multi-functionality; how to identify the mechanisms to be modelled and the affected processes to be included in the system boundaries. In this regard, an interesting support to address both present mechanisms and future developments could be provided by scenario modelling, as suggested also by Mathiesen et al. (2009). Frameworks for scenario development in LCA have already been developed, and can be found in Pesonen et al. (2000), Fukushima and Hirao (2002) and Spielmann et al. (2005). Börjeson et al. (2006) further detailed the approach of Pesonen et al. (2000), by proposing a scheme in which three main categories of scenarios are distinguished, namely predictive (what will happen?), explorative (what can happen?), and normative (how can a specific target be reached?). Each contains two scenario types respectively: Forecast and What-if, External and Strategic, and Preserving and Transforming. A clear identification of which type of scenarios is most appropriate for CLCA is difficult since there is a grey area between the different categories, especially between what-if scenarios and explorative scenarios, as pointed out by Börjeson et al. (2006). On the one hand, what-if scenarios would support the investigation of what will happen on the condition of some specified events in the future. Some case studies on CLCA reflect this logic (see, for example, Schmidt 2010). On the other hand, explorative scenarios, "staying to the facts" (Heijungs and Guinée 2007), explore situations and developments that are considered possible to happen. Moreover, they are elaborated with a long time horizon to allow for more profound changes, and accordingly, the long-term is considered the typical dominant effect in CLCA (Weidema et al. 2009), even if evidence is not always given in the case studies (for example, Schmidt (2010) considers only 5–10 years ahead, while most of the authors do not specify the time perspective adopted).

Thus, both scenarios can support CLCA in providing a more scientifically sound basis to model specific product-related futures with respect to technology development, market shifts, and so forth. In fact, scenarios could be used to frame the questions, before the system is modelled by means of the inclusion of mechanisms, and a representation of possible future alternatives is obtained. Such a structured and extended analysis would make it possible to address the several dimensions involved in any assessment, like time, the size of change, and the size of consequences, just to cite a few.

5 Conclusions and recommendations

CLCA has been applied for many years, but has only recently gained momentum in being applied for evaluating the environmental consequences of an action/decision into consideration in different disciplinary fields, also outside the domain of LCA. However, despite the numerous methodological papers published on the topic and the increasing number of applications, we have come to the conclusion that the CLCA is still far from a proper systematization. In fact, our analysis points out that the application of the consequential modelling is often done in a non-systematic and inconsistent way, suggesting that CLCA is still not fully understood either at a conceptual or at a modelling level. CLCA seems to be considered as a macro container in which different concepts flow in, not always mutually consistent in terms of the perspective adopted (prospective/retrospective), the direction in time (future/past), the typology of consequences analysed, and the effects included (marginal versus average). Overall, considering how CLCA is perceived and applied by practitioners, it appears that there are more

[6] Constraints can be technical, political, natural, and marked-related. A common simplification adopted in CLCA is to consider them as fixed entities due to the difficulties in modelling them. Thus, it is assumed that they do not change, even when the long-term horizon is analysed.
[7] The CPA, assuming away any interference from the system, works under the hypothesis of isolation. The temporal isolation assumes that those factors we consider frozen under the CPA move so slowly relative to the others that can be considered constant at any point in time. The causal isolation considers that the same factors are not significantly affected by the processes under study. Thus, the more the mechanisms are added, the more the CPA and its related constraints are relaxed, the more the steady-state and static models become tight because causes and effects are always intertwined in time.

shadows than lights in CLCA. The result is a lack of clarity about *what* CLCA is, *for what* it is useful, and *what it entails* from the methodological point of view.

Regarding the *what* question, our analysis points out that CLCA stands out as a modelling approach that takes (some) market mechanisms into account. We propose to use the logic of mechanisms as a good criterion for distinguishing between CLCA and the other modes of LCA. On the *modelling* side, while ALCA and CLCA share the same modelling principles, the main difference between them relies on the way in which the boundaries of the system are set and the processes to be included are selected. Which types of processes, and thus, which type of causal chains should be included and how to identify them are the main questions, which are dealt with differently and not always transparently by practitioners, leading to different results.

We suggest that scenario modelling could provide an important contribution to better structuring CLCA. In fact, it can increase the robustness of studies by providing an approach to think about plausible future developments in a structured manner (Zurek and Henrichs 2007). Moreover, in doing so, they also support the identification of relevant mechanisms to be included in the analysis. However, as data uncertainty can be very large in scenarios that refer to technologies not yet in use, accounting for future developments increases the need for methods to manage great uncertainties (Höjer et al. 2008).

As far as the *what for* is concerned, the analysis highlights that this aspect is not well addressed at the methodological or practical level, despite its relevance. The correct formulation of a question is central in every evaluation, but its importance continues to be neglected. We believe that the lack of attention to this important aspect is at the origin of the dispute between CLCA and the other modes or types of LCA over the last 20 years, and consequently, of the lack of clarity about the contexts in which CLCA could be applied. How to better link questions and models is an important field of research, not only for CLCA, and it requires the development of practical guidelines on how to frame questions, to identify what the problem to be tackled is exactly, what the derived questions are, what the technological options are, what the scale of the expected changes is, what the time frame of the question is, if a ceteris paribus assumption may hold, if the system analysed is replacing another system at a small scale, or if the technology used in the new system is expected to extend to many more applications on a larger scale (Guinée et al. 2009). It is the sum of all these answers that determines which methodological choices are relevant, and thus, which mode is most appropriate.

Besides framing the question, further efforts should be devoted to improve the existing approach, which is considered to still be in its earliest stages (Anex and Lifset 2009), and making it a robust support to decisions with long-term consequences (see, for example, the Renewable Fuel Standard programme developed by US-EPA, in which the use of CLCA for GHG emissions accounting is required). Efforts are necessary to clarify when and which market information is important, to improve long-term forecasting techniques, and to identify the affected processes.

However, together with shadows we believe that there are also many lights in CLCA, three in particular which we would like to point out. The first refers to the fact that CLCA forces practitioners to think about the correct formulation of a question. Since consequences are at the core of CLCA, we need to explicitly state them in the research question in order to identify which processes to include in the analysis and thus how to set boundary. Thus, it is thanks to CLCA that the topic of framing the question gained again its importance. The second aspect is the ability of CLCA to model some economic mechanisms. Despite the several methodological developments still needed and highlighted above, CLCA provides an important sophistication of the LCA methodology. It adds the capability of accounting for those linkages, pointed out by Graedel and van der Voet (2010), which are inherent in any sustainability evaluations. Building upon this concept, we arrive at the third positive aspect, that is the perspective offered by CLCA. We suggest that the concept of CLCA could be broadened (extended CLCA) so as to be considered as an approach—rather than a modelling principle with defined rules—to deepen LCA, providing the conceptual basis for including more mechanisms than the market ones.

In conclusion, scenario and modelling of mechanisms, together with framing the question, are three important research fields for CLCA that deserve attention, since they could be seen as the backbone of any assessment.

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Global Sensitivity Analysis: An Introduction

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Abstract: This presentation aims to introduce global sensitivity analysis (SA), targeting an audience unfamiliar with the topic, and to give practical hints about the associated advantages and the effort needed. To this effect, we shall review some techniques for sensitivity analysis, including those that are not global, by applying them to a simple example. This will give the audience a chance to contrast each method's result against the audience's own expectation of what the sensitivity pattern for the simple model should be. We shall also try to relate the discourse on the relative importance of model input factors to specific questions, such as "Which of the uncertain input factor(s) is so non-influential that we can safely fix it/them?" or "If we could eliminate the uncertainty in one of the input factors, which factor should we choose to reduce the most the variance of the output?" In this way, the selection of the method for sensitivity analysis will be put in relation to the framing of the analysis and to the interpretation and presentation of the results. The choice of the output of interest will be discussed in relation to the purpose of the model based analysis. The main methods that we present in this lecture are all related with one another, and are the method of Morris for factors' screening and the variance-based measures. All are model-free, in the sense that their application does not rely on special assumptions on the behaviour of the model (such as linearity, monotonicity and additivity of the relationship between input factor and model output). Monte Carlo filtering will be also be discussed to demonstrate the usefulness of global sensitivity analysis in relation to estimation.

Keywords: global sensitivity analysis, factor prioritisation, main effects, second-order interaction effects, nonlinear models

INTRODUCTION

The material in this presentation is taken from a primer on global sensitivity analysis entitled "*Sensitivity Analysis in Practice: A Guide to Assessing Scientific Models*" by Andrea Saltelli, Stefano Tarantola, Francesca Campolongo and Marco Ratto. This will appear with John Wiley & Sons by early 2004, and we shall refer to it as to Saltelli et al., 2004 in the following. The primer aims at guiding a non-expert user in the choice of the method to adopt for the user own problem. The methods recommended include the variance based measures, the method of Morris, and Monte Carlo filtering, e.g. some effective methods for global sensitivity analysis.

Global sensitivity analysis is the study of how the uncertainty in the output of a model (numerical or otherwise) can be apportioned to different sources of uncertainty in the model input”. Global could be an unnecessary specification here, were it not for the fact that most analysis met in the literature are local or one-factor-at-a-time.

All models have use for sensitivity analysis. Applications worked by the Joint Research Centre group for Applied Statistics include: Atmospheric chemistry (Campolongo et al., 1999a), transport emission modelling, fish population dynamics (Campolongo et al. 1999b), composite indicators (Tarantola et al. 2002), portfolios, oil basins models (Saltelli, 2002), capital adequacy modelling (for Basle II), macroeconomic modelling, radioactive waste management (Saltelli and Tarantola, 2002). Applications from several practitioners can be found in Saltelli et al. Eds. 2000, a multi-author book.

Prescriptions have been issued for sensitivity analysis of models when these used for policy analysis.
In Europe, the European Commission recommends sensitivity analysis in the context of the extended impact assessment guidelines and handbook (2002). Similar recommendation in the United States EPA’s White Paper on model use acceptability (1999)

The EC handbook for extended impact assessment, a working document by the European Commission, 2002, states: “A good sensitivity analysis should conduct analyses over the full range of plausible values of key parameters and their interactions, to assess how impacts change in response to changes in key parameters”. The EPA paper (1999) is less prescriptive, but insists on the need for uncertainty and sensitivity analysis.

Even leaving prescriptions aside, one cannot ignore that models have not escaped the post-modern critique of the role of science in society. Specific critiques of simulation modelling and model validation have been frequent in recent years. One example: <<*…most simulation models will be complex, with many parameters, state-variables and non linear relations. Under the best circumstances, such models have many degrees of freedom and, with judicious fiddling, can be made to produce virtually any desired behaviour, often with both plausible structure and parameter values.*>>, Hornberger and Spear 1981.

Also, from within the modelling community reminders of the problem were frequent: Konikov and Bredehoeft, 1992, proclaims: "*Groundwater models cannot be validated*". This cry of alarm was taken up by Oreskes et al. 1994, in an article on Science entitled "*Verification, Validation and Confirmation of numerical models in the earth sciences*", both works focusing on the impossibility of model validation. Two established laboratory, IIASA and RIVM, had considerable trouble with the perceived quality of their models, see Mac Lane 1989, and van der Sluijs 2002 respectively. The post-modern French thinker Jean Baudrillard (1990) presents 'simulation models' as unverifiable artefact which, used in the context of mass communication, produce a fictitious hyper realities that annihilate truth. Science for the post modern age is discussed in Funtowicz and Ravetz 1990, 1993, 1999, mostly in relation to Science for policy use, a settings which Gibbons (1994) calls “mode 2” scientific production.

Faced with these critiques, the modelling community may consider that a quality check as that which is provided by a careful sensitivity analysis is worth its effort.

Before we discuss the methods for sensitivity analysis, we would like to say a few words about the output Y of interest. In our experience, the target of interest should not be the model output per se, but the question that the model has been called to answer. To make an example, if a model predicts contaminant distribution over space and time, it is the total area where a given threshold is exceeded at a given time which would play as output of interest, or the total health effects per time unit.

One should seek from the analyses conclusions of relevance to the question put to the model, as opposed to relevant to the model, e.g.

- Uncertainty in emission inventories [in transport] are driven by variability in driving habits more than from uncertainty in engine emission data.
- In transport with chemical reaction problems, uncertainty in the chemistry dominates over uncertainty in the inventories.
- Engineered barrier count less than geological barriers in radioactive waste migration.

This remark on the output of interest clearly applies to model use, not to model building, where the analyst might have interest in studying a variety of intermediate outputs.

FIRST EXAMPLE: THE OBVIOUS TEST CASE

We move now to a self-evident problem, to understand the methods as applied to it. This is a simple linear form:

$$Y = \sum_{i=1}^{r} \Omega_i Z_i$$

Y is the output of interest (a scalar), Ω_i are fixed coefficients, Z_i are uncertain input factors distributed as

$$Z_i \sim N\left(\bar{z}_i, \sigma_{Z_i}\right), \quad \bar{z}_i = 0, \quad i = 1,2,\ldots r \;.$$

Y will also be normally distributed with parameters:

$$\sigma_Y = \sqrt{\sum_{i=1}^{r} \Omega_i^2 \sigma_{Z_i}^2}$$

$$\bar{y} = \sum_{i=1}^{r} \Omega_i \bar{z}_i$$

To make our point we stipulate as additional assumptions:

$$\sigma_{Z_1} < \sigma_{Z_2} < \ldots. < \sigma_{Z_r}$$

$$\Omega_1 > \Omega_2 > \ldots. > \Omega_r$$

According to most of the existing literature, SA should be done by taking derivatives, such as: $S_{Z_i}^d = \dfrac{\partial Y}{\partial Z_i}$, which would give for our model of Y: $S_i^d = \dfrac{\partial Y}{\partial Z_i} = \Omega_i$.

Hence the factors' ordering by importance would be $Z_1 > Z_2 > ... > Z_r$, based on our previous assumption that $\Omega_1 > \Omega_2 > .... > \Omega_r$, and this in spite of the fact that $\sigma_{Z_1} < \sigma_{Z_2} < .... < \sigma_{Z_r}$. This would seem to suggest that if our purpose is to rank input factors in terms to their contribution to the variability of the output, then simple derivatives such as $S^d_{Z_i} = \frac{\partial Y}{\partial Z_i}$ are not the best instrument to use.

A better measure could a normalised derivative of the type: $S^\sigma_{Z_i} = \frac{\sigma_{Z_i}}{\sigma_Y}\frac{\partial Y}{\partial Z_i}$, which, applied to our model, gives $S^\sigma_{Z_i} = \Omega_i \frac{\sigma_{Z_i}}{\sigma_Y}$

Comparing this with our previous expression $\sigma_Y = \sqrt{\sum_{i=1}^r \Omega_i^2 \sigma_{Z_i}^2}$, we obtain $\sum_{j=1}^r \left(S^\sigma_{Z_i}\right)^2 = 1$.

This is a nice result: the terms add to 1, and each of them gives the fractional contribution of the factor to the variance of the output. Unfortunately this only works for linear models.

If we want to tackle nonlinear models as well, we have to abandon derivatives and move into "exploration" of the input factors space, e.g. via Monte Carlo.

We generate a sample

$$\mathbf{M} = \begin{matrix} z_1^{(1)} & z_2^{(1)} & ... \; z_r^{(1)} \\ z_1^{(2)} & z_2^{(2)} & ... \; z_r^{(2)} \\ ... & ... & ... \; ... \\ z_1^{(N)} & z_2^{(N)} & ... z_r^{(N)} \end{matrix}$$

and run our computer program estimating the corresponding model output

$$\mathbf{y} = \begin{matrix} y^{(1)} \\ y^{(2)} \\ ... \\ y^{(N)} \end{matrix}$$

A natural thing to do at this point is to regress the y' s on the z_i's to obtain a regression model $y^{(i)} = b_0 + \sum_{i=1}^r b_{Z_i} z_i^{(i)}$, where asymptotically $\hat{b}_0 \cong 0, \hat{b}_{Z_i} \cong \Omega_i$, $i = 1,2,...r$. Most regression packages will already provide the regression in terms of standardised regression coefficients $\hat{\beta}_{Z_i} = \hat{b}_{Z_i}\sigma_{Z_i} / \sigma_Y \cong \Omega_i \sigma_{Z_i} / \sigma_Y$. Comparing $\hat{\beta}_{Z_i} \cong \Omega_i \sigma_{Z_i} / \sigma_Y$ with

$$S^{\sigma}_{Z_i} = \Omega_i \frac{\sigma_{Z_i}}{\sigma_Y},$$

it is easy to conclude that for linear models $\beta_{Z_i} = S^{\sigma}_{Z_i}$.

In summary, $\sum_{j=1}^{r}\left(S^{\sigma}_{Z_i}\right)^2 = \sum_{j=1}^{r}\left(\beta_{Z_i}\right)^2 = 1$, but only for linear models. Yet the regression coefficients are better than the derivatives in several respects.

Although for nonlinear models $\sum_{j=1}^{r}\left(\beta_{Z_i}\right)^2 \leq 1$, at least we now know how much linear the model is. This is given by the model coefficient of determination $R_y^2 = \frac{\sum_{i=1}^{N}\left(\hat{y}^{(i)} - \bar{y}\right)^2}{\sum_{i=1}^{N}\left(y^{(i)} - \bar{y}\right)^2}$.

We now know that we can decompose a fraction R_y^2 of the model variance using the β_{Z_i}. Furthermore the coefficients β_{Z_i} offer a measure of sensitivity that is multi-dimensionally averaged, unlike the $S^{\sigma}_{Z_i}$. For linear model this does not matter but it does, and a lot, for nonlinear ones. The drawback is when $R_y^2 << 1$; typically R_y^2 can be zero or near it for non-monotonic models.

In summary, we like the idea of decomposing the variance of the model output according to source (the input factors), but would like to do this for all models, independently from their degree of linearity or monotonicity. We would like a model-free approach.

In order to get there, we take a somehow twisted path and start asking ourselves the question: If I could determine the value of an uncertain factor, e.g. one of our Z_i and thus fix it, how much would the
variance of the output decrease? E.g. imagine the true value is z_i^* and hence we fix Z_i to it obtaining a "reduced" conditional variance: $V\left(Y \middle| Z_i = z_i^*\right)$. There are two problems with this quantity being a good measure of sensitivity. First I do not know where to fix the factor, and secondly for nonlinear model one could have $V\left(Y \middle| Z_i = z_i^*\right) \geq V(Y)$.

This difficulty can be overcome by averaging this measure over the distribution of the uncertain factors obtaining $E\left(V\left(Y|Z_i\right)\right)$, or $E_{Z_i}\left(V_{\mathbf{Z}_{-i}}\left(Y|Z_i\right)\right)$ where we have made explicit the variables over which mean and variance operators are applied. This measure has the property that $E\left(V\left(Y|Z_i\right)\right) \leq V(Y)$ always, and in particular
$E\left(V\left(Y|Z_i\right)\right) + V\left(E\left(Y|Z_i\right)\right) = V(Y)$, where the term $E\left(V\left(Y|Z_i\right)\right)$ is called a residual, and the term $V\left(E\left(Y|Z_i\right)\right)$ is known as the first order effect of Z_i on Y. A nice property of the main

effect is that it is large when a factor is influential. Furthermore it is easy to verify that for linear models $S_{Z_i} = \dfrac{V\left(E\left(Y|Z_i\right)\right)}{V(Y)} = \beta_{Z_i}^2$.

We have made a real progress, as while $\sum_{j=1}^{r}\left(\beta_{Z_i}\right)^2 = 1$ only holds for linear models, $\sum_{j=1}^{r}\left(S_{Z_i}\right) = 1$ holds for a much larger class of models: that of the additive models. For non-additive models, $\sum_{j=1}^{r}\left(S_{Z_i}\right) \leq 1$, which is also a way to define non-additive models. Yet the measure S_{Z_i} is very useful for <u>all</u> models, as it provides a rigorous answer to a precise sensitivity analysis setting: <u>setting FP</u>, for factors prioritisation. Let us then make a digression here, and describe this setting.

FACTORS' PRIORITISATION (FP) SETTING

Imagine that I must bet on a factor that, once "discovered" in its true value and fixed, would reduce the most V(Y). Of course I do not know where the true values are for the factors, hence I cannot compare the $V\left(Y \middle| Z_i = z_i^*\right)$ for the various factors. Hence the best choice I can make is, by definition, to choose the factor with the highest $V\left(E\left(Y|Z_i\right)\right)$ or , which is the same, the highest $S_{Z_i} = \dfrac{V\left(E\left(Y|Z_i\right)\right)}{V(Y)}$, whether the model is additive or not (Saltelli and Tarantola, 2002).

To complete all this, we must say something about non-additive model treatment, so let us complicate our model $Y = \sum_{i=1}^{r}\Omega_i Z_i$ by allowing both the Ω_i and Z_i to be uncertain, i.e. $Z_i \sim N\left(\bar{z}_i, \sigma_{Z_i}\right), \quad \bar{z}_i = 0, \quad i = 1,2,...r$ as before and $\Omega_i \sim N\left(\bar{\omega}_i, \sigma_{\omega_i}\right), \ \bar{\omega}_i = ci, i = 1,2,...r$, where c is a constant greater than zero (note: if the mean of the Ω_i were also null as that of the Z_i , then the model would be fully non-additive, as we shall see in a moment).

Our set of uncertain input factors is now $\mathbf{X} \equiv (\Omega_1, \Omega_2, ...\Omega_r, Z_1, Z_2, ...Z_r)$. We start crunching number estimating the sensitivity measures and we obtain the following results:

All S_{Ω_i} are zero.

All S_{Z_i} are > zero.

S_{Ω_i} is zero because the distribution of Z_i is centred in zero, and hence for any fixed value ω_i^* of Ω_i

$E\left(Y \middle| \Omega_i = \omega_i^*\right) = 0$, and *a fortiori* $V\left(E\left(Y|\Omega_i\right)\right) = 0$.

Given that $\sum_{j=1}^{r}\left(S_{Z_i}\right) \le 1$ where is the remaining variance? To find it out we must compute sensitivity indices on more than one factor. If we do that, we find that $\frac{V\left(E\left(Y|Z_i,Z_j\right)\right)}{V_Y}=S_{Z_i}+S_{Z_j}$, while, instead: $\frac{V\left(E\left(Y|\Omega_i,Z_i\right)\right)}{V_Y}>S_{\Omega_i}+S_{Z_i}$. The difference $S_{\Omega_i Z_i}=\frac{V\left(E\left(Y|\Omega_i,Z_i\right)\right)}{V_Y}-S_{\Omega_i}-S_{Z_i}$ is the second order (or two-way) effect of the two factors. We have discovered that our model is additive with respect to S_{Z_i}, S_{Z_j}, and non-additive with respect to S_{Ω_i}, S_{Z_i}.

Adding all the non-zero first order terms and all the non-zero second order terms gives back 1, i.e. 100% of the variance of Y is accounted for.
I.e. $\sum_{i=1}^{r} S_{Z_i}+S_{\Omega_i Z_i}=1$

For our model, all other terms of whatever order (1,2,3…2r) is zero. In general, if k is the total number of independent factors, then $\sum_i S_i+\sum_i\sum_{j>i}S_{ij}+\sum_i\sum_{j>i}\sum_{l>j}S_{ijl}+\ldots.S_{12\ldots k}=1$ (Sobol', 1993).

It is quite rare that in practical applications one computes all terms in the development above. The number of terms grows exponentially with k.

We are customarily happy with computing all the S_i plus a full set of synthetic terms called S_{Ti} which give for each factor X_i, the effect of all terms including that factor.

What are the total effect terms S_{Ti} and why do we need them? Let us compute one of them, by starting with the measure $\frac{V\left(E\left(Y|\mathbf{X}_{-\Omega_i}\right)\right)}{V_Y}=\frac{V\left(E\left(Y|\Omega_1,\Omega_2,\ldots\Omega_{i-1}\Omega_{i+1},\ldots\Omega_r Z_1,Z_2,\ldots Z_r\right)\right)}{V_Y}$. We have taken factor Ω_i as an example. Analogy with previous formulae should suggest that, by definition, this is the [first order] effect of all-but- Ω_i. Hence $S_{T\Omega_i}\equiv 1-\frac{V\left(E\left(Y|\mathbf{X}_{-\Omega_i}\right)\right)}{V_Y}$ will be the effect of all terms [any order] that include Ω_i ; for our model this is simply $S_{T\Omega_i}=S_{\Omega_i}+S_{\Omega_i Z_i}$, provided we remember that the S_{Ω_i} are zero as well, so that $S_{T\Omega_i}=S_{\Omega_i Z_i}$. Note that because of an algebraic relation already mentioned

$1-\frac{V\left(E\left(Y|\mathbf{X}_{-\Omega_i}\right)\right)}{V_Y}=\frac{E\left(V\left(Y|\mathbf{X}_{-\Omega_i}\right)\right)}{V_Y}$., so that the right hand expression is often used for the S_{Ti}.

There is a considerable symmetry between the S_i and S_{Ti}. Both indices can be computed in a single
shot at the cost of about N(k+2) simulations, where N is between 100 and 1000, to give an idea. In Saltelli, 2002, we use an extension of the method of Sobol', 1993. Both indices can also be computed using the Fourier based FAST method, as extended in Saltelli et al., 1999.

Furthermore S_i is ideal for factor prioritisation setting, already described, while S_{Ti} is ideal for the "factors fixing" setting (of which more in a moment).

A nice property of S_{Ti} is that if one is desperate for less expensive simulations, a rough estimate of these can be obtained via the method of Morris, at less than 1/10 of the cost, see Morris 1991. (We prefer to compute a "modulus" version of the test statistics, as described in Chapter 4, Campolongo et al., in Saltelli et al. Eds., 2000).

Finally one last useful property of variance based methods is their application "by groups", e.g.
$S_{\mathbf{\Omega}}+S_{\mathbf{Z}}+S_{\mathbf{\Omega},\mathbf{Z}}=1$, where $\mathbf{\Omega}=\Omega_1,\Omega_2,...\Omega_r$. The computational cost of this is just 3N. Or I can regroup as $\sum_{i=1}^{r}S_{\mathbf{A}_i}=1$, where $A_i=(\Omega_i,Z_i)$. The computational cost of this is kN. Note that in this latter expression all higher order terms are zero because there are interactions only within $A_i=(\Omega_i,Z_i)$.

Although in the first regrouping we save a lot in terms of model execution, and in the second we don't, there might be reasons other than economy to regroup factors. I might want to groups factors in different submodels. In this way, if I can fix all factors in the submodels may be I can skip the submodel altogether. I might want to separate controllable factors from uncontrollable ones, and so on.

A SECOND EXAMPLE: WHAT CAN SENSITIVITY OFFER FOR PARAMETER ESTIMATION

Let us now move to an estimation/calibration problem for a computational model with six parameters. We do not know how the model is done – imagine it is a computer code. The output of interest Y is a measure of likelihood is obtained after comparing the model prediction Y' with data, e.g.

Y=exp(-[sum of squared residuals of the predicted Y' versus the data]).

How can we characterise the good parameter set for calibration? A scatter plots of log-likelihood (e.g. of the sum of scores) vs. parameters is not very informative (Figure 1). Even "filtering", e.g. taking the best outcomes, those with the highest log-likelihood, leaves us in the dark (Figure 2). Plotting the factors value for the input (Figure 3) as well as for the input corresponding to the best values (Figure 4) is likewise noninformative. Note that if we computed on the filtered input factors (Figure 4) the pairwise correlation coefficients we would obtain zeros. Also Principal Component Analysis would not be informative as applied to the filtered input sample, as there are no correlations among the filtered factors. Computing the first order sensitivity indices for the log-likelihood and the second order ones (Figure 5), a story starts to emerge; there are non-zero second order effects, but only within the closed groups involving factors (1,2,3) and (4,5,6). Computing the third order effect (Figure 6) again only those pertaining to (1,2,3) and (4,5,6) are non-zero. Regrouping and adding the terms up gives an interesting result:

$$S_{123}^{c} = S_1 + S_2 + S_3 + S_{12} + S_{13} + S_{23} + S_{123} = 0.5$$
$$S_{456}^{c} = S_4 + S_5 + S_6 + S_{45} + S_{46} + S_{56} + S_{456} = 0.5$$

where we have used the supescript c symbol to denote the effects closed within the indices. The variance of the problem is characterised by two groups of three factors. Higher term orders are zero.

This leads the investigator to conclude that what could be reasonably estimated are two unknown functions of two parameter sub-sets. We can now reveal that the unknown function, our computer program, was the sum of two speres.

$$f(X_1,\ldots,X_6) =$$
$$= -\left(\sqrt{X_1^2 + X_2^2 + X_3^2} - R_1\right)^2 / A_1 - \left(\sqrt{X_4^2 + X_5^2 + X_6^2} - R_2\right)^2 / A_2$$

Were the investigator to identify this structure, by trial and error, he/she would conclude that all that estimation can provide are the two radiuses.

This concludes our illustration of sensitivity analysis as applied to a diagnostic setting, and we would now like to come back to our discussion of the settings for sensitivity analysis.

MORE ON THE SETTINGS FOR SENSITIVITY ANALYSIS

We have already mentioned that the sensitivity measure of the first order, $S_i \equiv \frac{V\left(E\left(Y|X_i\right)\right)}{V_Y}$ is the ideal measure for factor prioritisation. It is also easy to see that the total effect measure $S_{Ti} \equiv \frac{E\left(V\left(Y|\mathbf{X}_{-i}\right)\right)}{V_Y}$ is appropriate for a setting that we could call

“Factors Fixing”: Can I fix a factor [or a subset of input factors] at any given value over their range of uncertainty without reducing significantly the output variance? If factor X_i is totally non-influential, then all the variance is due to $\mathbf{X}_{-i}$, and fixing this vector results in $V(Y|\mathbf{X}_{-i}) = 0$. It is easy to see that the reverse is also true so that necessary and sufficient condition for X_i to be totally non-influential is $S_{Ti} \equiv 0$.

Other settings that we have found useful are the following.

Factors mapping: Which factor is mostly responsible for producing realisations of Y in the region of interest? This can be treated with Monte Carlo Filtering and related tools (described elsewhere at this workshop).

Variance cutting: Reducing the variance of the output of a prescribed amount fixing the smallest number of factors. This setting can be dealt with using a combination of the S_i and S_{Ti} measures (Saltelli and Tarantola, 2002).

Why do we need settings? One way in which a sensitivity analysis can go wrong is because its purpose is left unspecified or vague (e.g. “find the most important factors”). One throws different statistical tests and measures to the problem and obtains different factors rankings. What can then be concluded? Models can be audited and settings for sensitivity analysis can be audited as well. For this reason we believe that importance must be defined beforehand.

A FEW MORE COMMENTS ON PRACTICES

What else can go wrong in a sensitivity analysis? Two instances come to mind:

There are too many outputs of interest, as we discussed at the beginning. What is the question asked from the model? Is the model relevant to the question? The optimality of a model must be weighted with respect to the task, according to a current mode of thinking. According to Beck et al. 1997, a model is “relevant” when its input factors actually cause variation in the model response that is the object of the analysis. Model “non-relevance” could flag a bad model, or a model used out of context (e.g. a gun to kill a fly). Excess complexity could also be used to silence or to fend off criticism from stakeholders, e.g. in environmental assessment studies.

Patchy or piecewise sensitivity (performed by sub-model, or one possible model at a time, or one factor at a time): Not only conflicts with the requirement of focus just mentioned, but leads to a dangerously incomplete exploration of the uncertainties; interactions are overlooked. All uncertainties should be explored simultaneously. Also the procedure of fixing non-influential factors should be conducted in this way, as fixing factors based on their first order effect can be dangerous as discussed above. The Ω_i of our initial example all have first order equal zero.

A posteriori sensitivity: Once an analysis has been produced, its revision via sensitivity analysis by a third party is not something most modellers will willingly submit to. Sensitivity analysis should be used in the process of model development, prior and within model use in analysis.

One should never forget that an unpleasant (or pleasant, depending from the viewpoint) feature of sensitivity analysis is that it might falsify the analysis altogether, e.g. by showing that the model cannot answer the question given the uncertainties, or that the model is irrelevant, or that the variation in the output of interest (e.g. a contamination level in an estuary) is insensitive to the available policy options given the uncertainties. A nice example that shows how SA can falsify a model as applied to a policy issue is described in Chapter 20, Tarantola et al., of Saltelli et al., Eds. 2000.

CONCLUSIONS

We can itemise our main conclusions as follows. There is an increased need, scope and prescription for quantitative uncertainty and sensitivity analyses. Methods are mature for use, e.g. in terms of literature, software, computational cost, tested practice, ease of communication.

In spite of this one observes a “slow start” of quantitative methods in practical analyses

Variance based measure are concise, easy to understand and to communicate, reduce to the elementary test (the standardised regression coefficients $\beta_{i_i}^2$) for linear model, relate to the popular method of Morris.

We also like and use methods in the MC filtering family.

Whatever the method one uses, we think it important that the framing of the analysis be defensible and meaningful to its users.

Figure 1. Log-likelihood for the six input factors.

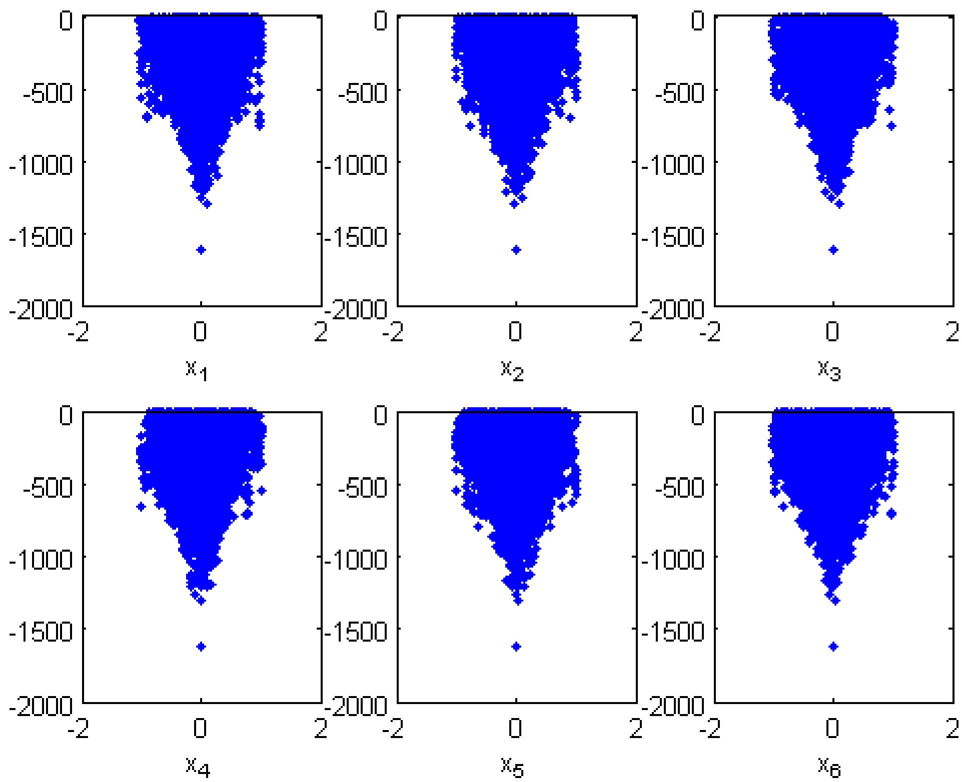


Figure 2. Same as Figure 1, for values of log-likelihood > -200.

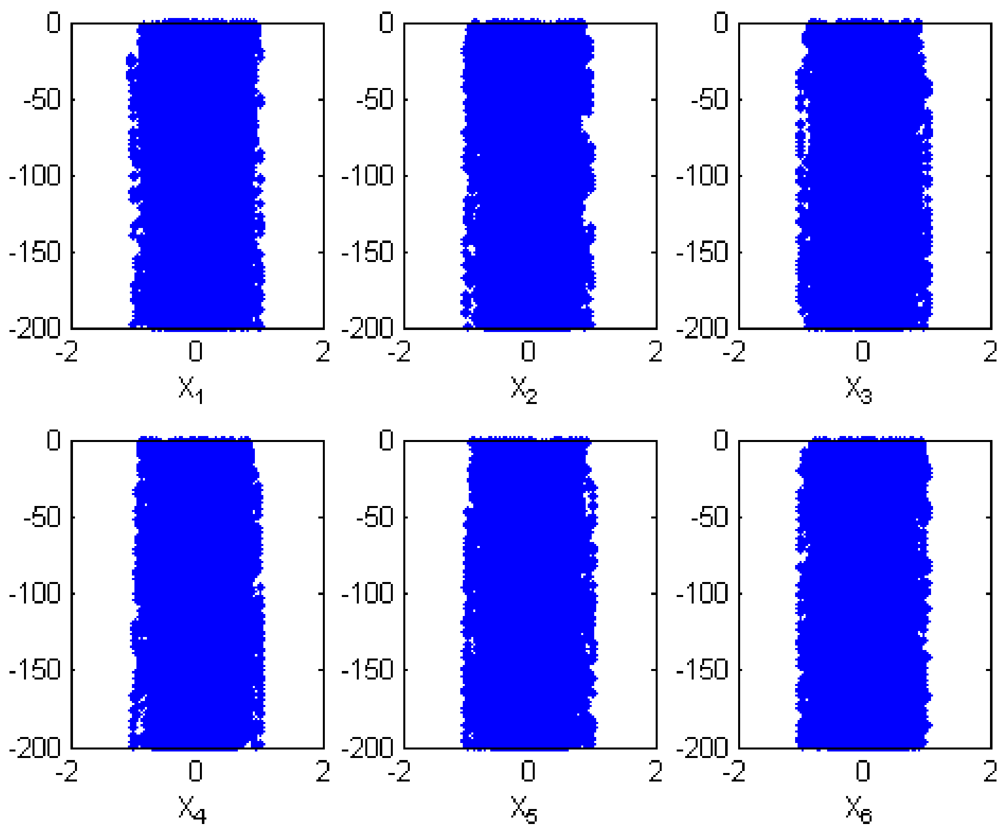


Figure 3. Pair-wise scatter plots of input factors.

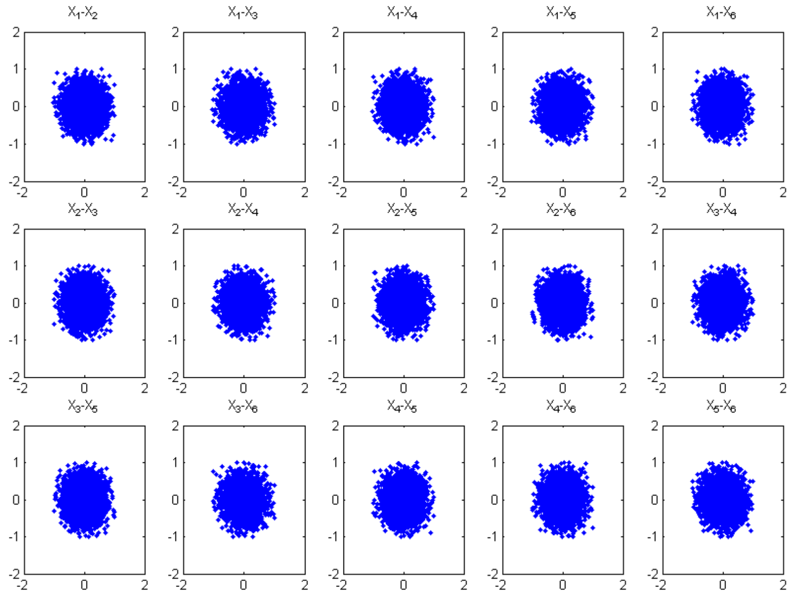


Figure 4. Same as the previous figure, for values of log-likelihood > -200.

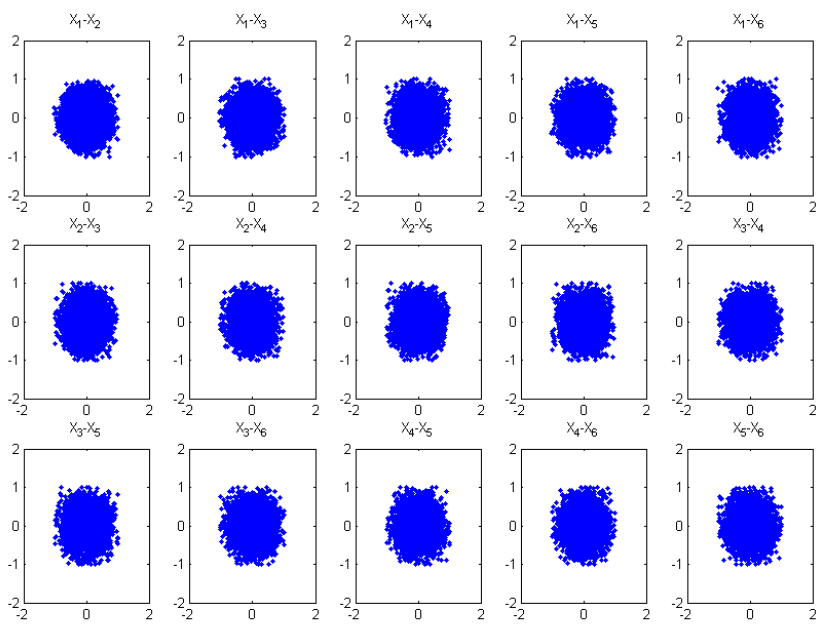


Figure 5. First- and second-order sensitivity indices for the log-likelihood.

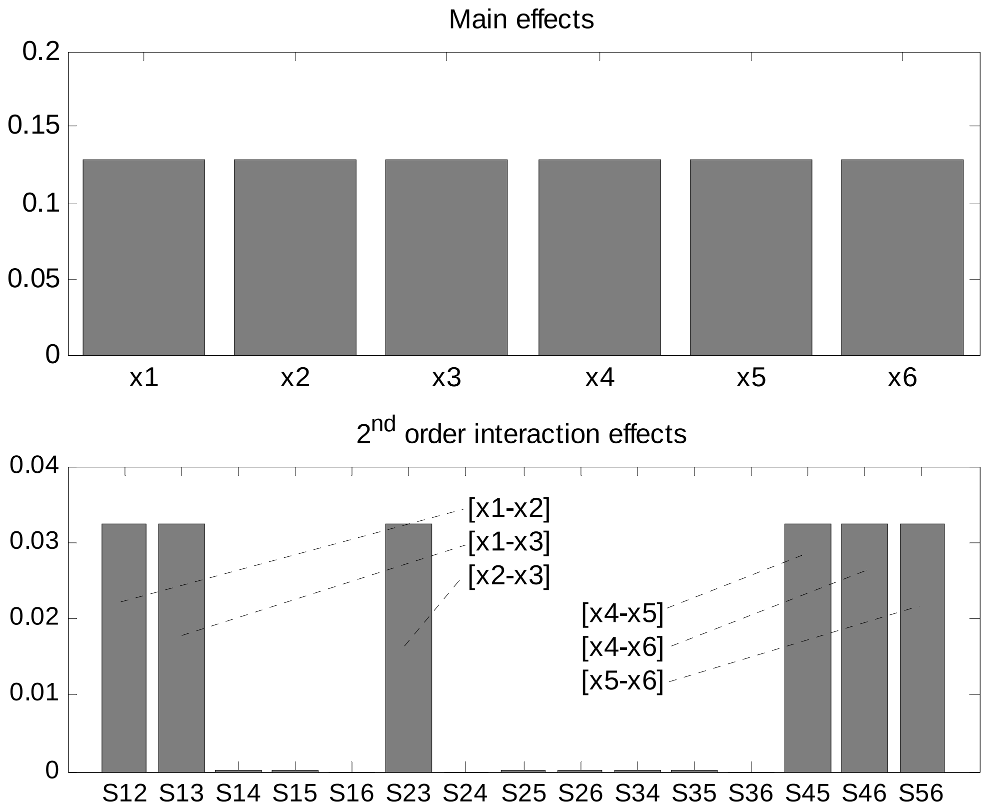
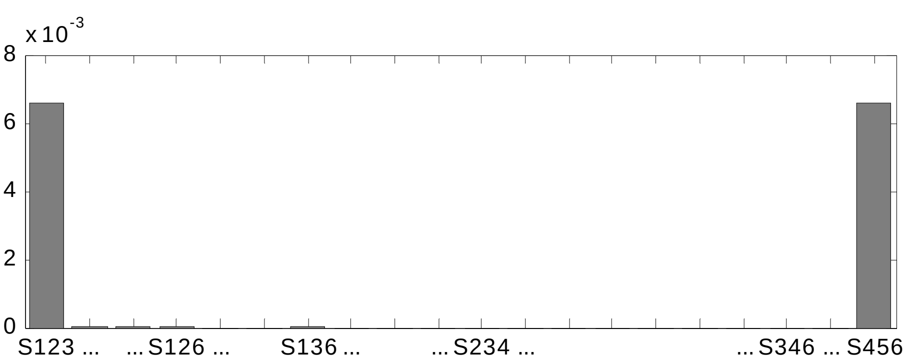


Figure 6. Third-order sensitivity indices for the log-likelihood.



NOTES

The Joint Research Centre distributes freely the software SIMLAB for uncertainty and sensitivity analysis. More information from stefano.tarantola@jrc.it. Marco Ratto (marco.ratto @jrc.it) has developed a set of scripts in Matlab to run global sensitivity analysis in diagnostic settings (e.g. with filtering plus variance based methods, see our two-sphere example). This is also available.

A forum to discuss sensitivity analysis issues is available at http://sensitivity-analysis.jrc.cec.eu.int/.
It includes a FAQ section, introduction to the main methods and a bibliography.

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Attributional or consequential Life Cycle Assessment: A matter of social responsibility

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ABSTRACT

Results of Life Cycle Assessment (LCA) are critically dependent on the system boundaries, notably the choice of attributional or consequential modelling. Published LCA studies rarely specify and justify their modelling choices. Since LCA studies are typically performed within the context of social responsibility and product life cycle management, this article investigates the relationship between social responsibility paradigms and the system modelling choices in LCA. We identify three different social responsibility paradigms: Value chain responsibility, Supply chain responsibility and Consequential responsibility. We point out that while there is no generally right or wrong choice of system model, each responsibility paradigm implies a specific matching system model. We then argue that all responsibility paradigms ultimately imply a consequential perspective, namely that of responding to the concerns of the system stakeholders. Although value or supply chains are systems defined without concern for consequences, and thus may include activities that the decision maker cannot influence, the chosen system is still analysed and assessed by accounting for its social consequences, and it is for these consequences that social responsibility is then taken. We argue that it is inconsistent to exclude consequences of own actions (i.e. the consequential system) while including consequences from actions of others in value chain or supply chain. We thus conclude that a consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle and *may* additionally take responsibility for the consequences of other activities in the value chain or supply chain. We end the article with recommendations on reporting on LCA system models that are more specific than those of the current LCA standards.

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1. Introduction

This article presents the results of an investigation into the relationship between social responsibility paradigms and the system modelling choices in LCA. This investigation was motivated by the widespread lack of specification and justification of the modelling choices made in published LCA studies, which leads to confusion when LCAs on the same product arrive at different results and conclusions. Since LCA studies are typically performed within the context of social responsibility and product life cycle management, the results of our investigation may contribute to improved clarity on and justifications of the modelling choices in LCA. The research methods applied are critical review and deductive logic from the first principles of formal ethics.

The current literature, which we review in Section 2, discuss different principles for allocation of responsibility within a value chain, but do this with the implicit premise that the relevant system for which responsibility is taken is the value chain, i.e. the economically allocated fraction of either purchases or sales. However, delimiting the system to that of value chain already implies an allocation of responsibility. Based on the definition of social responsibility and its concept of "sphere of influence" as well as the literature on system delimitation in LCA, we point out that any choice of system boundary necessarily involves the application of a normative rule.

In Section 3, we identify three different system types: value chains, supply chains and consequential product life cycles, each representing a specific social responsibility paradigm: value chain responsibility, supply chain responsibility and consequential

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responsibility, respectively. We describe and exemplify each of these system types and their rules for system boundary closure.

In our discussion, Section 4, we argue that it is possible to take responsibility for more than one system perspective at a time, thus combining systems additively, while the selective use or mixing of different system modelling rules in the assessment of a single system leads to obscuring and shifting of responsibility. We then argue that all responsibility paradigms ultimately imply a consequential perspective, namely that of responding to the concerns of the system stakeholders. We argue that it is inconsistent for a socially responsible decision maker to exclude consequences of own actions (i.e. the consequential life cycle) while including consequences from actions of others in the value chain or supply chain.

We thus conclude, in Section 5, that assessment of social responsibility *must* always include the consequential product life cycle and *may* additionally include consequences of other activities in the value chain or supply chain, and call for increased transparency in reporting of modelling choices and system boundaries for LCA studies.

This paper does not investigate the initial motivations for taking social responsibility, but exclusively investigates the consequences of a consistent application of social responsibility to LCA modelling.

2. Review of the current literature

2.1. Principles for allocation of responsibility within a value chain

In the current literature, allocation of responsibility for environmental impacts has mainly been discussed in the context of the global regulation of greenhouse gas emissions. In this literature, it has been pointed out that the allocation of responsibility to nations exclusively for their direct territorial emissions ignores the important role of final consumption as the main driver for the emissions (Kondo et al., 1998; Eder and Narodoslawsky, 1999; Munksgaard and Pedersen, 2001; Peters and Hertwich, 2008). This *territorial responsibility* (often referred to using the more ambiguous term "producer responsibility") creates perverse incentives for "carbon leakage", including the outsourcing of emissions to countries with lower commitments (Pedersen and de Haan, 2006), and thus for an inefficient allocation of resources. As an alternative to territorial responsibility, Life Cycle Assessment (LCA) and the closely related Environmentally Extended Input-Output Analysis (EEIOA) provide a solution based on *consumption responsibility*, allocating to the consumers of a country all the emissions caused by the production and consumption of that country's *final demand* for goods and services. This implies an incentive for customers to purchase from the value chain with the least impacts and overcomes the problems of the territorial allocation. A third alternative is *income responsibility* (term coined by Marques et al., 2012; first conceptual description by Gallego and Lenzen, 2005), allocating all the emissions generated *downstream* in the value chain, including final consumption and disposal, to the activities that receive income from the value chain, i.e. in proportion to their value added. This implies an incentive for suppliers to compete to supply the sales chain (chain of customers) with the least impacts. As an example, Lenzen and Murray (2010, p. 268) mention:

> "Important contributions to the downstream carbon responsibility of engineers are the emissions they enable through lending their technical skills to run power plants, beef farms, natural gas rigs, and coal mines."

implying – somewhat counterproductively – that, to reduce their personal responsibility, engineers should stop providing services (e.g. of advice on emission reduction) to the most polluting industries, and instead focus on the least polluting.

Although the discussion of these three responsibility principles (summarized in Table 1) has mainly taken place in the context of global regulation of greenhouse gas emissions, the principles are generally applicable to any environmental aspect of production and consumption.

The relevance and fairness of the three principles outlined above and in Table 1 are discussed in the above-mentioned sources and a number of others (Rodrigues et al., 2006, Lenzen et al., 2007; Rodrigues and Domingos, 2008; Marques et al., 2013). These authors suggest different criteria to determine the appropriate combinations of the consumption and income responsibility principles, based on the premise that both of these principles are legitimate, as formulated by Marques et al. (2013, p. 162):

> "After all, for final consumers (households and government) to purchase goods and services, they must first earn their income (wages, taxes or interests) as suppliers of primary factors of production. Thus, it is as legitimate to account for the emissions generated upstream that are embodied in final demand, as to account for emission, which are enabled downstream by primary inputs."

One of the proposed criteria is that the responsibility should be allocated to the actors receiving benefit from the transaction and thus indirectly benefit from the environmental damage. While most authors identify the consumers as the ones receiving the benefit, and therefore settles for consumption responsibility, Rodrigues et al. (2006) and Marques et al. (2012) follow Ferng (2003) and regard both consumption and income as benefits. None of the authors give any consideration to consumer and producer surplus as the normal measure of social benefit from a transaction (Mankiw and Taylor, 2006).

Some of the suggested criteria are mutually exclusive. In the case of Rodrigues et al. (2006) even within the same article, requiring both

> "that environmental responsibility should verify a normalization condition, such that the sum of the environmental responsibility of all agents should equal total environmental pressure"

and

> "the indicator should not display wrong signals, only allowing for a decrease in environmental responsibility of an agent if there was a decrease in overall direct environmental pressure" (both quotes on p. 259)

which are mutually exclusive conditions, since the first refers to - and can only be fulfilled in - a steady-state analysis of environmental pressure and the second condition refers to a change in environmental pressure and can only be fulfilled in an analysis of changes, which is not possible in the analysed steady-state system.

Lenzen et al. (2007) claim that their specific shared responsibility allocation approach will provide the same efficiency incentives as a property rights regulation or a tax/subsidy directly targeting the same impacts. However, since the shared responsibility is arbitrarily assigned from the principles, there is no reason to believe that this would incentivise behavioural change in the same way as economic incentives would. Aside from this unsubstantiated claim by Lenzen et al. (2007), none of the above-mentioned authors address the consequences, effectiveness or efficiency of the two modelling principles with respect to achieving a reduction in environmental damage, most explicitly formulated by Lenzen and Murray (2010, p. 264):

Table 1
Three principles for allocation of responsibility within a value chain.

Responsibility principle	Driver	Activity scope	Impacts
Territorial	Activity	Direct	Direct
Consumption	Demand	Direct and upstream	Direct and embodied
Income	Supply	Direct and downstream	Direct and enabled

> "For the purpose of this article, we take an ex-post perspective, in which actions have occurred, so that the problem of evaluating alternative scenarios does not come up."

Thereby they miss the important point that while a consumer shifting purchase from one unconstrained supplier to another will lead to a consequent shift in production of these suppliers, there is no similar direct consequence of a producer shifting between supplying different customers (if such a shift is at all practically possible), since there is nothing that stops other producers from filling the empty space and supply the abandoned customers, thus resulting in a net zero impact on production and emissions. This asymmetry of power in the value chain is a strong argument for consumption responsibility and against income responsibility, when the purpose is to efficiently stimulate real-life changes.

More importantly for our work here, all the sources mentioned above are based on the premise that the system for which responsibility is taken is the value chain, i.e. the economically allocated fraction of either purchases (consumption responsibility) or sales (income responsibility); see e.g. Rodrigues et al. (2006, p. 259) and Lenzen and Murray (2010, Table 1 on p. 263) for an explicit formulation of this. None of the authors appear to realise that by choosing the value chain as the system under study, they have already made an allocation of responsibility. None of the reviewed articles address the arguments for or the implications of this implicit modelling choice. In this article, we seek to fill this gap by pointing out that *prior* to the discussion on the relevance of the consumption and income responsibility principles and their implied modelling choices, there is a fundamental modelling choice in the delimitation of the system, and that the value chain is only one of these possible delimitations, and not necessarily the most relevant one.

2.2. Social responsibility

The concept of social responsibility has developed from its initial formulation as "corporate social responsibility" (Holme and Watts, 2000, p. 10):

> "the commitment of business to contribute to sustainable economic development, working with employees, their families, the local community and society at large to improve their quality of life"

to the current ISO 26000 definition in which the "corporate" has fallen away, following the realisation that the concept is equally applicable to all types of organisations:

> "responsibility of an organization for the impacts of its decisions and activities on society and the environment, through transparent and ethical behaviour that
>
> - contributes to sustainable development, including health and the welfare of society;
> - takes into account the expectations of stakeholders;
> - is in compliance with applicable law and consistent with international norms of behaviour; and
> - is integrated throughout the organisation and practised in its relationships" (ISO 26000:2010, Clause 2.18)

The term "relationships" in ISO 26000 refers to an organisation's "sphere of influence", defined as the:

> "range/extent of political, contractual, economic or other relationships through which an organization has the ability to affect the decisions or activities of individuals or organizations" (ISO 26000:2010, Clause 2.19)

which is further explained in Clause 5.2.3:

> "This sphere of influence includes relationships within and beyond an organization's value chain. However, not all of an organization's value chain necessarily falls within its sphere of influence. It can include the formal and informal associations in which it participates, as well as peer organizations or competitors. An organization does not always have a responsibility to exercise influence purely because it has the ability to do so. For instance, it cannot be held responsible for the impacts of other organizations over which it may have some influence if the impact is not a result of its decisions and activities. However, there will be situations where an organization will have a responsibility to exercise influence. These situations are determined by the extent to which an organization's relationship is contributing to negative impacts. There will also be situations where, though an organization does not have a responsibility to exercise influence, it may nevertheless wish, or be asked, to do so voluntarily."

In the report of the Special Representative of the United Nations Secretary-General on the Issue of Human Rights and Transnational Corporations and other Business Enterprises, Ruggie (2008, Clause 4):

> "concluded that "sphere of influence" is too broad and ambiguous a concept to define the scope of due diligence with any rigour"

notably because the concept:

> "conflates two very different meanings of "influence". One is "impact", where the company's activities or relationships are causing human rights harm. The other is whatever "leverage" a company may have over actors that are causing harm or could prevent harm. Impact falls squarely within the responsibility to respect; leverage may only do so in particular circumstances." (Ruggie, 2008; Clause 12)

In this article we apply insights from the debate on attributional and consequential modelling in LCA to contribute more formal clarity on the different ways to define the sphere of influence for social responsibility.

2.3. The unavoidability of a normative system delimitation in LCA

All economic activities are interlinked through a global network of product flows,[1] which means that there is no *objective* way to delimit an organisation's sphere of influence as any discrete part of the world's activities. From the perspective of a specific organisation, all other economic activities in the world can be found as contributing a small share to the value chain or supply chain of the organisation. Likewise, when looking at consequences forward in time, the activities of any specific organisation will to some extent contribute to or impact on a large part of the other economic activities in the world. In this network perspective, there is no end to the sphere of influence of an organisation. Any practically applicable delimitation will therefore require the application of a normative rule.

In the context of life cycle inventory analysis (LCI), Ekvall (2000) suggests that different normative rules for system delimitation can be linked to different ethical positions. Notably, teleological situation ethics (consequentialism) can be linked to the effect-oriented (consequential) LCI methodology and deontological rule ethics can be linked to accounting, thus suggesting that the latter may be relevant in a context where the actor wishes to support, be part of, or otherwise be associated with what is deemed to be a "good" system, or to be dissociated with what is deemed to be a "bad" system. However, Ekvall (2000) acknowledges that an application of deontological rule ethics would require an agreement on what is regarded as "being associated with" and therefore concludes that "Further research is required before an operational LCA methodology can be derived from rule ethics". In a comment to Ekvall, Weidema (2003) states:

> "Nevertheless, it is natural that a commissioner of a life cycle study may feel that it is more relevant to study the processes in the immediate supply chain than those actually affected by the product substitutions. It is important to clarify whether the interest of the commissioner is really in the environmental impacts of products (i.e. in LCA) or more in the environmental impacts of the supply chain as such, since the latter interest may be better handled through supply chain management." (Weidema, 2003, p. 18)

In this article, we build on and expand these ideas.

3. Value chains, supply chains and product life cycles

In this section, we describe three different system types (value chains, supply chains and product life cycles) that we have identified as representing three different social responsibility paradigms: Value chain responsibility, Supply chain responsibility and Consequential responsibility (see also Fig. 1). The two first system types (value chains and supply chains) have become known in LCA as attributional models, while the product life cycle is a consequential model. These terms were originally coined at an international workshop on electricity data for LCI in 2001, where it was stated that attributional and consequential modelling are intended to answer different questions (Curran et al., 2005):

- Attributional LCI aims to answer "how are environmentally things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window"?
- Consequential LCI aims to answer "how will flows change in response to decisions"?

The two modelling approaches have been defined in the glossary of the Shonan database guidelines (Sonnemann and Vigon, 2011) as:

- Attributional approach: System modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule.
- Consequential approach: System modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit.

Note that the three system types and their corresponding social responsibility paradigms have no immediate relation to the three principles for allocation of responsibility within the value chain, reviewed in section 2. In the discussion in section 4.1 we will furthermore argue that the discussion on allocation of responsibility is irrelevant for LCA because it is possible to take responsibility for systems from more than one responsibility perspective at a time and responsibility need not be divided. That is, contrary to Gallego and Lenzen (2005), we argue that several actors can assume full responsibility, so that responsibility is not a conserved quantity like mass.

3.1. Value chain responsibility

The term "value chain" was coined by Porter (1985) in the context of analysing competitive advantage. In his definition: "The value chain … views the firm as being a collection of discrete but related production functions, …defined as activities" (Porter, 1985, p. 39). When linked to the value chains of suppliers and buyers, "A firm's value chain is embedded in a larger stream of activities that I term the *value system*" (Porter, 1985, p. 34). While the term "value system" may be the most correct to signify such an interlinked system of activities, it may also easily be confused with the same term used in ethics as a system of consistent values used for the purpose of ethical integrity.[2] We therefore use the term *"value chain"* to signify ***a system of interlinked activities that contribute value added to a product***, and implicitly "value chain responsibility" to signify taking responsibility for this system.

In practice, the value chain of an activity is identified by tracing each cost item input to an activity backwards in the chain. The cost for one (purchasing) activity, is a revenue for the supplying activity. For each activity, a part of the revenue leaks out as payments to employees and entrepreneurs, taxes, and rents (together known as "value added"). In a closed steady-state system, all the original revenue must eventually leave the system as value added, thus providing a clear delimitation of the activities included in the system. When summing up all the value added over a value chain, you obtain the "life cycle cost", which is equal to the revenue of the analysed activity (Moreau and Weidema, 2015). When joint production activities are partitioned to obtain the value chain for a single product, the accounting balance (cost = revenue) for each single-product activity is maintained by partitioning each input proportionally to the share that each joint product has in the overall revenue. In LCA jargon this is known as "revenue allocation" or more loosely as "economic allocation" (Ardente and Cellura, 2012). Note that unless there is complete proportionality between the physical properties and the revenues for the joint products, the resulting systems will not be physically balanced (Weidema, 2017b). In general, a value chain will therefore not reflect the

[1] We use the term "product" to denote both goods and services.

[2] We generally use the term value in the meaning of "economic value", i.e. marginal values expressed in monetary units, as opposed to absolute ethical values.

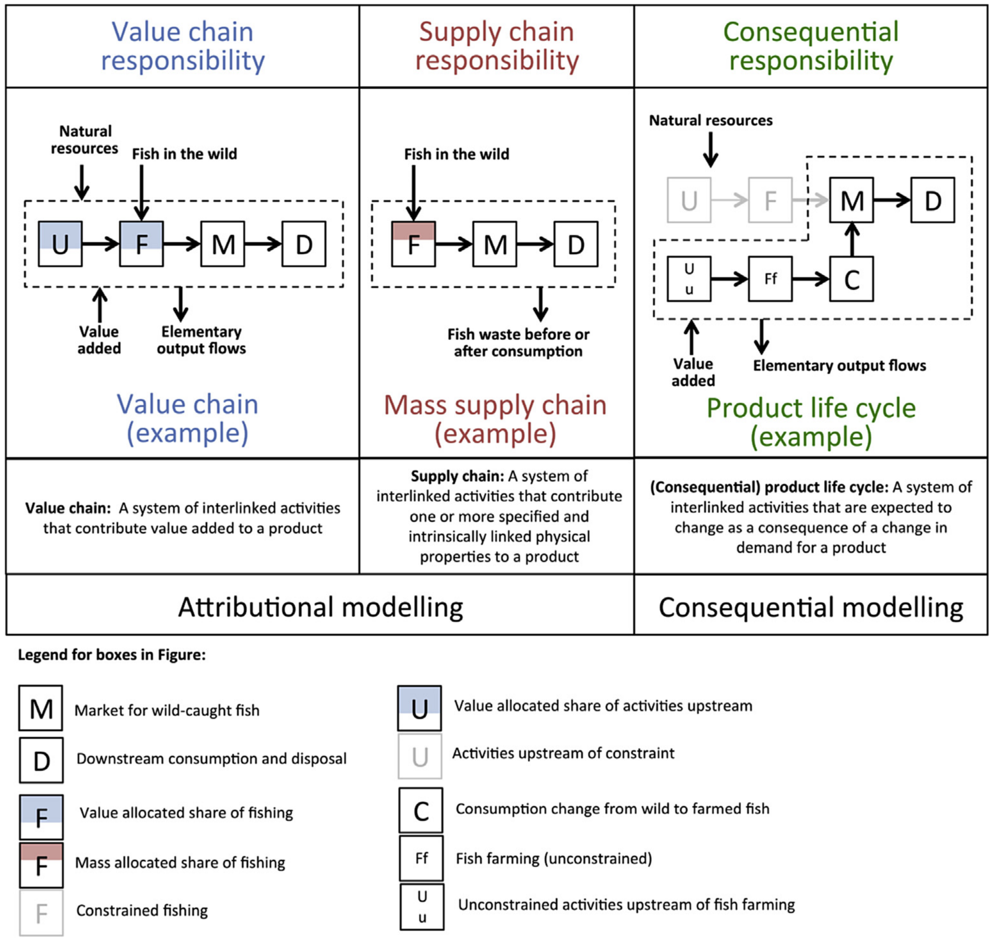

Fig. 1. The three identified responsibility paradigms and the three corresponding system types illustrated by the example of wild-caught fish.

physical causalities of purchasing a product.

As an example of a value chain, we can describe the value chain for wild caught fish: The revenue from the sale of the target fish goes to the fishing activity, which also has a valuable by-catch. The valuable inputs to the fishing, i.e. those that are accounted as cost items, are the different value added items (payments to employees and entrepreneurs, taxes, and rents for the fishing rights) and mainly energy inputs and the fishing gear and the fishing vessel. The non-value added items link to further activities, such as oil extraction, steel manufacture and so on, in the end including a little of practically all activities in the world. For each of the upstream activities, a share is allocated to the target fish output in proportion to the share of the revenue for the target fish in the total revenue (from target fish and by-catch). The value added items of each activity are system external inputs, which are not followed further upstream, and this thus provides closure to the system. Note that also the fish, before being caught, is not followed further upstream, because as a wild resource it has no value beyond the rent paid for the fishing rights, which is a transfer from the user to the (public or private) resource owner and part of the value added.

3.2. Supply chain responsibility

The term "supply chain" is sometimes used as a synonym for what we above called the value chain. However, in its more strict usage, the term refers to the logistic chain of suppliers to an activity, and can be used with reference to both physical supplies and supplies of services or information. It is increasingly being used in relation to traceability, where a customer wishes to know the exact origin of a product (Norton et al., 2014).

Obviously, the physical supply chain of materials and parts may be very different from the supply chain of information, energy or other services to the same activity. For an unambiguous identification, it is therefore necessary to specify a supply chain in terms of the physical (as opposed to economic) properties it supplies, e.g. mass, energy, or a specific elemental content. If two properties are not intrinsically linked, the supply chain of one property may at some point disengage from the supply chain of the other property, and it then becomes impossible to say which of the two chains should be followed. We thus define a supply chain as *a system of interlinked activities that contribute one or more specified and intrinsically linked physical properties to a product*. Obviously, it is possible to overlay several supply chains, depending on the purpose of the analysis, and how many properties the decision maker wishes to take responsibility for.

In practice, the supply chain of an activity is identified by tracing the investigated property backwards in the chain. The input of one activity is either the output of a supplying activity or an external input to the system. In a closed steady-state system, all the input must eventually enter the system as what, in LCA jargon, are called "elementary" flows, i.e. flows that come from the environment without previous human transformation, thus providing a clear delimitation of the activities included in the system. Taking the property mass as example, the sum of all mass inputs over a mass supply chain will equal the mass of the analysed product and all wastes and by-products from the system. When joint production activities are partitioned to obtain the supply chain for a single product, the mass balance for each single-product activity is maintained by partitioning each input proportionally to the share that each joint product has in the overall mass output. In LCA jargon this is known as "mass allocation". To provide a physically consistent and balanced system, each specific supply chain analysis can only trace one specific property (or several intrinsically linked properties). Note that unless there is complete proportionality between the analysed physical property and the revenues for the joint products, the accounting balance (cost = revenue) will be lost. In general, a supply chain will therefore not reflect the economic causalities of purchasing a product.

We use the mass of wild caught fish to provide an example of a supply chain: The mass of the target fish can be traced back to the fishing activity, again having a valuable by-catch and non-valuable discards. The mass of the wild fish input is the system external input, which is not followed further upstream, and this thus provides closure to the system. The input mass is allocated proportionally between target fish, by-catch and discards on a mass basis. Other inputs to the fishing activity, such as fishing gear, fishing vessel, etc. are often treated as service inputs (the service of providing fishing capacity) and therefore do not provide any net inputs to the mass of the supply chain. While it is obvious that such capital goods are necessary for the production of the primary product, they are not necessary for the mass balance or traceability of the primary product.

3.3. Consequential responsibility

The consequentialist idea that actors should be responsible for the consequences (impacts) of their production or consumption actions is fundamental to environmental management systems, as defined by the ISO 14000 series. These systems organise the management of an organisation's significant environmental aspects, which ISO 14001:2015 (Clause 3.2.2) defines as those aspects that have or can have one or more significant environmental impacts. Environmental impacts are in turn defined as a change to the environment (ISO 14001:2015, Clause 3.2.4).

The ISO 14040-series on LCA reflects this in the description of how product life cycles (product systems) are constructed, with respect to intermediate inputs:

> "The supplementary processes to be added to the systems must be those that would actually be involved when switching between the analysed systems. To identify this, it is necessary to know: (…)
>
> - which of the unconstrained suppliers/technologies has the highest or lowest production costs and consequently is the marginal supplier/technology when the demand for the supplementary product is generally decreasing or increasing, respectively." (ISO 14049, Clause 6.4)

and with respect to co-production:

> "The study shall identify the processes shared with other product systems and deal with them according to the stepwise procedure:
>
> a) Step 1: Wherever possible, allocation should be avoided by
>   - dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or
>   - expanding the product system to include the additional functions related to the co-products (…)
>
> The inventory is based on material balances between input and output. Allocation procedures should therefore approximate as much as possible such fundamental input-output relationships and characteristics." (ISO 14044, Clause 4.3.4.2).

What is reflected in these quotes is that the study of (potential) impacts must necessarily imply modelling the *changes* resulting from a (potential) decision. This implies modelling *marginal* changes when the studied changes are small, and *incremental* changes when the changes are larger, as opposed to the average modelling implied in the delimitation of value chains and supply chains.[3] For both marginal and incremental modelling "the supplementary processes to be added to the systems must be those that would actually be involved when switching between the analysed systems" (ISO 14049, Clause 6.4).

In practice, a product life cycle is identified by tracing each required product input, physical or monetary, backwards through the long-term marginal suppliers of each product, i.e., the suppliers that will change their production capacity in response to an accumulated change in demand for the product. The resulting product life cycle is thus demand-driven and based on consumption responsibility, cf. the literature reviewed in section 2 of this article.

When co-productions are modelled as described in the quote from ISO 14044 above, i.e. by subdivision of combined production and by placing dependent joint[4] by-products as negative inputs instead of as positive outputs, thus expanding the product system with the displaced substitutes of the joint by-products, all physical and economic balances remain intact. In parallel to what we found for the value chain, the value added summed over all activities in a product life cycle equals the "life cycle cost", and the sum of inputs of each balanceable property (such as dry wet and elemental mass as well as energy) will equal the outputs from the system. In general, a product life cycle will therefore reflect both the physical and economic causalities of purchasing a product.

In parallel to what we found for the value chain, the revenue generated by the original demand must eventually leave the product life cycle as value added, thus providing a clear delimitation of the activities included in the product life cycle. The activities included are limited to those that react to the change in revenue, corresponding to the first-order effects of the original spending. Implicitly, when comparing products with different prices, a consequential model will include first-order price rebound effect, while excluding second order effects, such as changes in consumption patterns that may result from the redistribution of the initial spending on the population groups that receive the primary factor income, or second order effects of stimulating specific activities, such as education, research and technological development (Weidema et al., 2015).[5]

We use the demand for wild caught fish as an example of a product life cycle: A marginal increase in demand for wild caught fish leads to a temporary increase in the price of wild fish and thus

[3] It is remarkable that in the extensive value chain and supply chain literature the concept of the *marginal* value or supply chain does not appear to have played any significant role, although this perspective could have important implications also for these fields.

[4] Subdivision by physical causality is only possible for combined production, where the amounts of the co-products can be varied individually, while this is not the case for joint production.

[5] Such multiplier effects can be – and are – included in distributional equity assessments, social life cycle impact assessment, and dynamic economic models for the study of societal developments, i.e. beyond the normal consequential, first order, steady-state modelling of the consequences of small-scale decisions.

to an increase in revenue of the fishing activity. However, under the current conditions, the increase in revenue does not lead to increased output of wild fish, since the total output of wild fish is currently constrained, partly by quotas, partly by the limited physical amounts that are available for harvesting within the current economic limits. Instead, the marginal increase in the price of wild fish leads the marginal consumer of wild fish to leave the market and substitute with a more affordable next-best unconstrained alternative, typically farmed fish. This increases the revenue to the aquaculture production and increases the output of farmed fish correspondingly.[6] The necessary inputs to aquaculture production are then stimulated and can be linked backwards to further activities in parallel to what was done for the value chain. The difference to the value chain analysis is that the inputs are only linked to marginal, unconstrained suppliers, and when co-products are encountered they are dealt with as described in the quote from ISO 14044 above. As for the value chain analysis, the value added items of each activity are not followed further upstream, thus providing closure to the system. Note that the primary fishing activity is *not* a part of the system, in spite of the system having a net functional output of wild caught fish (the demanded wild fish is actually supplied, but this is due to the marginal consumers renouncing their demand for wild fish and not due to changes in outputs from primary fishing).

What this example illustrates is that, in contrast to the average modelling applied for value and supply chains, consequential product life cycles do not include constrained activities, i.e. activities that do not change their output in response to a change in demand for this output. However, an actor may still want to influence such constrained activities, for example when constrained activities provide the most cost-efficient environmental improvements, or when the constrained activity is part of a value chain or supply chain that the actor wishes to take responsibility for. Since constrained activities cannot be influenced by general changes in demand for their products, it is thus necessary to use other instruments that link the improvements to products of unconstrained activities.

An example could be a food company wishing to change their wild fish supply (a constrained product due to quotas and physical/economic limits) from using bottom trawling to using less environmentally damaging alternatives (longlines, gillnets, pots and traps). One option for the food company would be to introduce a label for fish caught by low-impact gear. Since the *shift* in fishing gear is not constrained, the environmental consequence of consumers buying the labelled product will be 1) the difference in impact between the fish caught with low-impact gear and the "normal" trawl-caught fish, and 2) the additional production of the alternative farmed fish (because the change of fishing gear does not in itself mean that more wild fish is caught). Another option that would *not* require a separate label would be for the food company to publicly promise an increase its sourcing from this low-impact source in some proportion to the purchases of the consumer of the general (unconstrained) goods, thus linking the specific desired changes in the constrained activity to the demand for an unconstrained product.

The literal meaning of responsibility implies that it is delimited to what an organisation is *able to respond to* (Power et al., 2008), i.e. meaningfully act upon and change, cf. Section 2.2 on the concept of "sphere of influence". From this perspective, it can be argued that a socially responsible organisation should not be exclusively concerned with the activities within the specific value chains, supply chains, or product life cycles of the organisation, but rather be responsible for influencing what it *can* influence (cf. the concept of "noblesse oblige", i.e. that with wealth, power and prestige come responsibilities). The socially optimal action for any organisation at any specific point in time is to change the specific activities that provide the currently most cost-efficient environmental improvement, no matter whether these activities show up as important within the specific value chains, supply chains, or product life cycles of the organisation. However, this is still a consequential perspective in the sense that it is concerned with change, i.e. with the impacts of the actions of the organisation – it just does not limit these actions to those narrowly related to the product output. If an organisation decides to actively influence an activity that is not part of its product life cycles, this implies spending economic resources that indirectly are dependent on the revenue obtained from the product life cycles of the organisation, and this spending thus indirectly becomes a part of these product life cycles. Here it should be noted that a unit of analysis could be the product portfolio of an organisation, resulting in an aggregate of several product life cycles. Such a portfolio LCA has also been called "an Organisational LCA" (Blanco et al., 2015).

In line with this idea of responsibility for what can be influenced, it can also be argued that inaction, i.e. the omission of action in a situation where action could have been taken, has consequences. If comparing a change to "business-as-usual", the difference between the two can be expressed either as a consequence of a change (action) or as a consequence of inaction ("business-as-usual").

3.4. System cut-offs

As mentioned above, all three system types (value chains, supply chains and product life cycles) have a natural closure, when all of the analysed property has been accounted for as input to the system. The system boundary is thus given by the cut-off rule that the inputs of "elementary flows" are not followed further upstream. No further cut-off rules are required.

However, for some specific delimitations of responsibility, namely in those situations suggested by Ekvall (2000), where an actor wishes to support, be part of, or otherwise be associated with what is deemed to be a "good" system, or to be dissociated with what is deemed to be a "bad" system, Weidema (2003) suggests that an additional cut-off rule could be applied, namely that of tier distance or transactional distance:

> "the concept of "being associated with" is hardly meaningful beyond a few steps backwards or forwards in the supply chain, thus rendering LCA too sophisticated a technique for identifying the relevant associations." (Weidema, 2003, p.18)

4. Discussion

4.1. Mixed or combined systems: shifting or expanding responsibility?

Many published LCAs have applied a mix of allocation rules, combining revenue allocation, physical allocation, and system expansion within the same system, which of course obscures both how the system boundary is identified and what question the study will be able to answer (Weidema, 2017b). As illustrated by our example of wild caught fish, different system models leave out different activities and place different emphasis on those that are

[6] The constraint on wild fish implies that an increase in output of farmed fish can only be obtained by increasing the share of vegetable protein input in aquaculture. It is also possible that part of the consumers will substitute with other food products, such as chicken, which are also based on vegetable protein.

included, so that a skilful mix of modelling rules can make nearly any undesired activity disappear or lose importance.

Selective application or mixing of different system modelling rules within the same system can therefore only be seen as a way of shifting (away) responsibility. But if the different system models are instead used in combination, the possibility arises of taking responsibility for more than one system perspective at the same time, i.e. expanding rather than shifting responsibility. In contrast to the statement of Gallego and Lenzen (2005) that

> "it is intuitively clear that the responsibility for impacts associated with transactions in a productive system has to be somehow divided between the supplier and the recipient of the respective delivered commodity" (Gallego and Lenzen, 2005, p. 366)

we do not see any arguments suggesting that it is not possible to take responsibility for more than one system perspective at a time, nor any arguments suggesting that responsibility has to be divided when two or more actors take responsibility for the same activity. This view was already put forward by Eder and Narodoslawsky (1999). In this view, the allocation of responsibility becomes irrelevant, and what matters is instead what the co-responsible stakeholders can do about the impacts.

The problem with the allocation of responsibility within a value chain outlined in Section 2.1 and in Table 1 seems to be that responsibility is being allocated according to the actors' ability to influence/change, while the impact is being quantified by a steady-state model, rather than by the marginal product life cycle that is necessary for analysing changes/decisions between alternatives. In the latter, the life cycle impact is that of a product and not of a stakeholder: The life cycle impact is the same for all stakeholders, and need not to be divided.

4.2. Responsibility for improvements

Social responsibility and environmental management systems imply a responsibility to seek to improve. But if the focus is on responsibility for improvements, why choose a product perspective in the first place? It is important to distinguish between assessment of impacts of products (whether as value chains, supply chains or product life cycles), and assessments of impacts of industries/activities. Assessment of environmental problems of industries can be made without a product perspective. It is misleading to apply a product perspective when addressing industry/activity problems, because the product perspective leaves out relevant issues either by allocation (attributional) or by looking only at the unconstrained activities (consequential). And some of the important issues that should be tackled may lie in either the parts that have been allocated away or in the constrained parts of the industries that cannot be affected by indirect market-based demand changes, but only by incentivising direct changes. In our example of fisheries, an industry analysis would point to trawling as the most environmentally damaging activity. Studies using the attributional value or supply chain of wild caught fish would both allocate a part of the trawling away to the by-products, while the consequential product life cycle would not include trawling at all, except if included explicitly by the decision maker.

Changes to attributional systems have consequences beyond the system boundaries, i.e. in the parts that have been allocated away, or made less important through averaging. Choosing between attributional systems based on their relative impacts implies a disregard for these consequences, and can lead to sub-optimal decisions and even to increases in overall impacts. For example, an attributional study of cow's milk may conclude that impacts per litre of milk should be reduced by increasing the efficiency of the milk output, e.g. by changes in fodder composition. The result would be that the dairy cow system would produce less meat per litre of milk. Since the demand for meat is unchanged, the reduction in meat output from the dairy cow system would have to be compensated by an increase in dedicated meat production by beef cattle, which could lead to an overall increase in environmental impact per litre of milk compared to the original dairy system that substituted more of the dedicated meat production. This consequence of reduced substitution would not be captured by the attributional system because the meat by-product would have been cut-off by allocation. More examples can be found in Weidema (2017a).

If you want to make an improvement to a constrained part of a system, this implies actions that go beyond the consequential product life cycle, but these actions can still be assessed with a consequential model of the specific change. In our example of fisheries, even though the primary fishing is a constrained activity, and therefore not part of the consequential product life cycle, a consequential model can still be used for a separate assessment of the environmental implications of any specific decisions to *improve* the primary fishing activity, such as the shift from bottom trawling to longlines or gillnets. However, it is important to understand that such an improvement in fishing practices will not in itself change the (marginal) life cycle of wild fish products because primary fishery is not part of the life cycle due to the constraints. To link the changed fishing practices to the consumption of the marginal product, a label is required that implies a commitment of the producers to change fishing practices in proportion to the sale of the labelled product.

5. Conclusion

5.1. The consequences of responsibility

We have identified, described and exemplified three different system types: value chains, supply chains and consequential product life cycles, each representing a specific social responsibility paradigm: value chain responsibility, supply chain responsibility and consequential responsibility, respectively. Any choice between these systems will be *normative,* since the global interlinked nature of economic activities leaves no *objective* way to delimit an organisation's social responsibility to any discrete part of the world's activities. It is possible to take responsibility for more than one system perspective at a time, thus combining systems additively, while the selective use or mixing of different system modelling rules within the same system leads to obscuring and shifting of responsibility.

The literal meaning of responsibility implies a focus on consequences that can be meaningfully acted upon and changed, i.e. responding to the concerns of the stakeholders within the sphere of influence of the organisation. This consequentialist idea is also fundamental to social responsibility and environmental management systems, as defined by the ISO 26000 and ISO 14000 series. Even if a system is defined without concern for consequences, and thus includes activities that the decision maker cannot influence, it is the consequences of the chosen system for which responsibility is taken.[7]

Following from the first principles of formal ethics (Gensler,

[7] This can also be seen from the fact that Life Cycle Impact Assessment, i.e. the modelling of the further impact once the elementary flows have left the system, has always been based on marginal analysis, i.e. the additional impact of an additional unit of the elementary flow.

1996) it does not appear to be consistent for a decision maker to exclude consequences of own actions (i.e., the product life cycle) while including consequences from actions of others in the value chain or supply chain.[8] Thus, we conclude that a consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle and *may* additionally take responsibility for consequences of other activities in the value chain or supply chain. The consequential life cycle can be seen as the "impact" part of the sphere of influence discussed in Section 2.2, while the additional activities included from the value or supply chain can be seen as representing the "leverage" part of the sphere of influence.

Assessments of products (whether as value chains, supply chains or product life cycles) are not adequate for identifying important improvement options (hotspots) outside the narrow system boundaries of these assessments. Assessments of products are relevant once hotspots have been identified within the system you have chosen to be responsible for (value chain, supply chain, product life cycle, or the whole world).

In order to avoid burden-shifting and to fulfil the purpose of LCA as a tool for improvements, identified improvement options need to be compared, in order to identify the one that provides the largest improvement. These comparisons are always comparisons of consequences of implementing each of the options. In this context, it is also important to note that a decision to continue *business-as-usual* can also be modelled as a change relative to ceasing the activity (the zero baseline). This implies that a consequential product life cycle model can also be used to assess a single existing activity or product against the zero baseline. Two such separately developed life cycle models may then later be compared.

5.2. Communication

Both attributional and consequential system models may be difficult to communicate. Consequential product life cycles only include those activities that change as a result of a decision. This is not always the activities that one would intuitively think, which may make the consequential product life cycles appear counter-intuitive until context is communicated and the model is investigated more in detail. Attributional system models (value chains or supply chains) may at first sight appear simpler to communicate because they follow a more static logic. Communication difficulties appear only at closer examination in the form of the:

- Artificial nature of allocated activities (that have no real-life parallel) and systems (that violate the law of conservation of mass and energy).
- Subjective choices of allocation factors and system boundaries that leave out some of the consequences of the production and consumption of the product.

Ideally, an agreement should be made to use the same system model and database for all product assessments. The arguments provided in this article support an agreement on the consequential life cycle as the default system model that *must* always be used, which *may* then be supplemented by additional, specific, value chain or supply chain related activities.

Published LCA studies rarely specify and justify their modelling choices and system boundaries. To make it possible to interpret LCA results meaningfully, it is necessary that the social responsibility context is clearly communicated, i.e.:

- who is responsible for the analysis and its use (decision maker, stakeholder, analyst, shared responsibility)?
- whether the full consequential product life cycle has been included, specifying any product-related consequences that may have been cut off, and
- whether any additional value chain or supply chain related activities have been included, specifying which or with what allocation rules (allocation keys, point of allocation) and cut-off criteria (tier distance, simple contribution, or specific activities for cumulated contribution).

These reporting requirements are more specific than those of the current LCA standards.

[8] The relevant ethical principle can in everyday language be expressed as "Practice what you preach". This is also known from the parable of the mote and the beam (Matthew 7:1–5).

Acknowledgements

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LCI METHODOLOGY AND DATABASES

Estimation of the size of error introduced into consequential models by using attributional background datasets

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Abstract
Purpose A systematic comparison is made of attributional and consequential results for the same products using the same unit process database, thus isolating the effect of the two system models. An analysis of this nature has only recently been made possible due to the ecoinvent database version 3 providing an access to both unallocated and unlinked unit process datasets as well as both attributional and consequential models based on these datasets. The analysis is therefore limited to the system models provided by ecoinvent.
Methods For both system models, the analysis was made on the life cycle inventory analysis (LCIA) results as published by ecoinvent (692 impact categories from different methods, for 11,650 product/activity combinations). The comparison was made on the absolute difference relative to the smallest absolute value.
Results and discussion The comparison provides quantified results showing that the consequential modelling provides large differences in results when the unconstrained (marginal) suppliers have much more/less impact than the average, when analysing the by-products, and when analysing determining products from activities with important amounts of other coproducts.
Conclusions The analysis confirms that for consequential studies, attributional background datasets are not appropriate as a substitute for consequential background. The overall error will of course depend on the extent to which attributional modelling is used as part of the overall system model. While the identified causes of differences between the attributional and consequential models are of general nature, the identified sizes of the errors are specific to the way the two models are implemented in ecoinvent.

Keywords Attributional modelling · Comparison · Consequential modelling · Coproducts · Decision support · Marginal suppliers

1 Introduction and objective of analysis

In life cycle inventory (LCI) analysis, it is common to distinguish between the two fundamentally different modelling principles that have become known as attributional and consequential modelling. These terms were originally coined at an international workshop on electricity data for LCI in 2001 (Curran et al. 2005), where it was stated that attributional and consequential modelling responds to different types of questions:

- Attributional LCI aims to answer "how are environmentally things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window?" (Curran et al. 2005)
- Consequential LCI aims to answer "how will flows change in response to decisions?" (Curran et al. 2005)

The two modelling approaches have been defined in the glossary of the Shonan database guidelines (Sonnemann and Vigon 2011) as follows:

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- Attributional approach: system modelling approach in which the inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule.
- Consequential approach: system modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit.

While it is thus generally acknowledged that the two approaches have different purposes and give answers to different questions, it is not unusual to see attributional modelling or a mix of the two modelling approaches applied to studies with a consequential purpose, with results presented as a comparative or as an estimate of the effect of increasing or decreasing system output, which points to a lack of understanding among practitioners of the relation between application area and modelling in general (Andrae 2014; Plevin et al. 2014).

Before the advent of the ecoinvent database version 3 (Ecoinvent 2013) that provided a generally available consequential background database, the only practically available option for consequential LCIs was to use an attributional database in the background and changing the foreground processes—or those processes with the largest influence on the result—to consequential modelling. It is therefore of interest to investigate the size and extent of the errors introduced by this practice.

In this paper, the size and extent of the errors introduced by using attributional data in consequential studies are investigated by comparing the life cycle impact analysis (LCIA) results obtained with the two modelling approaches for the same products using the same unit processes from the same unit process database, thus isolating the effect of the two modelling approaches. An analysis of this nature has only recently been made possible due to the ecoinvent database version 3 providing an access to both unallocated and unlinked unit process datasets as well as both attributional and consequential models based on these datasets. The analysis is therefore limited to the system models provided by ecoinvent, and while the identified causes of differences between the attributional and consequential models are of general nature, the identified *sizes* of the errors are specific to the way the two models are implemented in ecoinvent, see Box 1 (Electronic Supplementary Material): "Summary of linking rules distinguishing attributional and consequential system models in ecoinvent", which summarizes the automatic database linking rules as provided in more detail in Chapter 14 of Weidema et al. (2013).

This paper is not intended to be a validation of or a statement on the correctness of the consequential modelling in the ecoinvent database, but exclusively to investigate the size and extent of the errors introduced by using attributional background datasets for consequential studies.

2 Methods

The analysis has been performed on two system models from the ecoinvent database version 3.1, published in 2014, namely the attributional system model "Allocation, ecoinvent default" and the consequential system model "Substitution, consequential, long-term". For both system models, the analysis was made on the LCIA results as published by ecoinvent for 692 impact categories from different LCIA methods and for the 11,650 available product/activity combinations.

Of the 11,650 products, 1345 product/activity combinations are found only in the attributional model, not in the consequential. These are by-products that in an attributional model appear with their allocated result in a separate dataset, while this is not the case in a consequential model where by-products are eliminated by substitution.

Of the 11,650 products, 318 product/activity combinations are found only in the consequential model, not in the attributional. These are the first of all conditional exchanges that represent induced changes in consumption related to constrained markets, which are not included in an attributional model. Furthermore, due to the incompleteness of the database, some wastes are not produced by any activity in the database while an activity providing treatment of this waste can still exist as a dataset in the consequential model. In the attributional model, such treatment activities are not included.

The remaining 9987 product/activity combinations were then compared, on the absolute difference relative to the smallest absolute value:

$$\mathrm{ABS}(\mathrm{A}-\mathrm{C})/\mathrm{MIN}(\mathrm{ABS}(\mathrm{A}),\mathrm{ABS}(\mathrm{C}))$$

3 Results

Of the 9987 comparable product/activity combinations, 127 were excluded due to one or both of the values being zero, which occur when a dataset is empty or has no relevant exchanges for a specific LCIA method.

Out of the 127 excluded product/activity combinations, 29 only give zero result in the attributional model, which occurs when a multi-output dataset is empty except for the product outputs. The attributional LCIA result will be zero for all coproducts after allocation, while for the determining coproduct the consequential model will provide a result that includes the upstream displacements caused by the by-product output, and the dataset will therefore no longer be empty.

Four of the 127 excluded product/activity combinations only give zero results in the consequential model. This occurs for the not fully utilised by-products, Argon and Nitrogen, where the consequential market links to a burden-free reduction in release, while the attributional system performs allocation with the other inert gases.

For the 9860 remaining product/activity combinations, the most extreme differences of several orders of magnitude between the two system models are found for very specific impact categories, e.g. "EDIP: non-renewable resources, lanthanum", "ecological scarcity: pesticides to soil", "ReCiPe: ionising radiation".

To make the results more generally relevant, the following analysis exclude the specific LCI results, resource impact categories, and older LCIA methods (CML2001, EPS2000), bringing the total number of impact categories down from 692 to 401.

On an average in the 401 impact categories:

- 67% of the results have >10% difference
- 22% of the results have >100% difference
- 16% of the results have >200% difference
- 5% of the results have > order of magnitude difference

The differences are slightly smaller when limiting the comparison to the 11 total single-score categories, where:

- 56% of the results have >10% difference
- 13% of the results have >100% difference
- 9% of the results have >200% difference
- 3% of the results have > order of magnitude difference

This reduction in differences in the single-score categories is expected, since they aggregate a large number of small differences with opposite signs that counterbalance each other. Still, the amount and size of differences is noteworthy.

4 Discussion

When analysing the causes for the above differences between the attributional and consequential LCIA results for the same products, it was found that the causes can be grouped under five headings:

- Marginal suppliers very different from average
- Speciality products
- Multiple determining products
- By-products from treatment activities
- Determining products heavily influenced by by-products

In the following, each of these situations is described with an illustrative example taken directly from the ecoinvent database. The examples are not meant as examples of the best possible or most exhaustive modelling of each of the illustrated situations.

4.1 Marginal suppliers very different from average

Since the attributional model includes an average of all suppliers, while the consequential model only includes the marginal (unconstrained) suppliers, it is expected that large differences will result when the marginal suppliers have very different impacts than the average suppliers.

One of the more extreme examples of this is the data on land tenure, as illustrated in Fig. 1, where the consequential result is 100 times larger than the attributional. The output of the market for "land tenure, arable land" is on average supplied partly (1%) by intensification and clear-cutting with impacts of 0.069 Pt/unit of output and partly (99%) by land already in use with zero impacts. In the consequential model, "land already in use" is a constrained activity, not able to react to a change in demand, thus all the required output has to be supplied by the unconstrained "intensification and clear-cutting".

4.2 Speciality productions

A specialty production is an activity that is dependent on the input of a material for treatment, yet is not a treatment activity, i.e. the determining product of a speciality production is not a service of treatment for a material.

Products that are labelled based on a specified percentage of recycled material will always be speciality products, as in the example of "Graphic paper, 100% recycled" in Fig. 2. Upstream, the attributional modelling only includes the treatment activity, which is then—together with the speciality production itself—allocated by revenue together with the products of the waste paper providers, resulting in an impact of only 4 Pt/Mg recycled graphic paper. In the consequential model, an additional demand for the speciality product reduces the amount of the waste paper that goes to the marginal treatment activity for the material, here the production of recycled tissue paper. To keep the output of tissue paper constant, the reduced amount recycled tissue paper must be compensated by an increase in the virgin tissue paper production. The net impact of the reduced supply of waste paper to the tissue industry is 111 Pt/Mg, which together with the 51 Pt/Mg from the speciality production itself leads to an overall impact of 162 Pt/Mg recycled graphic paper (compared to the 4 Pt/Mg in the attributional model). Note that in the consequential model, an additional demand for a fully utilised recycled material does not change the overall amount of material recycling but only the relative proportions of recycled material in different products (here, more in graphic paper, less in tissue).

4.3 Multiple determining products

As a general rule for joint production, there will be a maximum of one determining coproduct from each coproducing

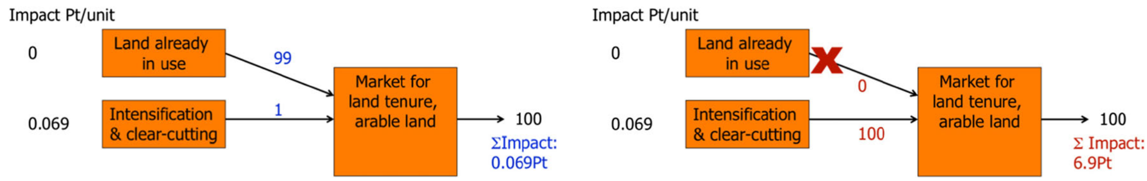


Fig. 1 LCIA results for the attributional (*left*) and consequential (*right*) models for the output of the market for "land tenure, arable land". Data from the "ReCiPe Endpoint (E, A):total:total" method

unit process. But a special situation arises when more than one joint product from the unit process have no alternative production routes, in which case, all of these joint products will be separately determining. In this situation, the prices of the joint products will adjust until all the joint products have the same normalised market trend, since only then their markets will be cleared (Weidema et al. 2009). In this situation, a change in demand for one of the joint products will influence the production volume of the joint production in proportion to its share in the gross revenue of the joint production. This is equivalent to the result of an economic partitioning (allocation) of the coproducing process. Furthermore, a consumption adjustment will take place via constrained markets for the joint products, in order to maintain a balance in supply and demand (and thus in mass balances, which is otherwise lost in economic partitioning).

In the ecoinvent database version 3.1, there is only one situation of multiple determining products, namely rare earth oxides production. Here, we describe the modelling for one of these products, namely lanthanum oxide. A more detailed description of the modelling of the entire dataset in ecoinvent can be found on the website consequential-lca.org (Consequential-LCA 2015).

A demand for 1 kg lanthanum oxide on the market is met by a 1 kg increase in supply in both the attributional and the consequential model. However, as shown in Fig. 3, the attributional model provides this supply exclusively from an allocated part of the coproducing activity, while the consequential model takes into account that the revenue provided by the demand for lanthanum oxide alone is only able to stimulate a change in output from the rare earth oxides production of 0.352 kg, namely the amount that corresponds to the revenue from 1 kg lanthanum oxide relative to the total revenue from all the determining products. The remaining supply (0.648 kg) must therefore come from a reduction in consumption by the marginal consumers of lanthanum oxide. At the same time, the stimulated increase in output of 0.352 kg lanthanum oxide implies a corresponding increase in the other rare earth oxides (Neodymium, Cerium, etc.). Clearing the markets for these products require a corresponding increase in consumption of these rare earth oxides of 0.897 kg. The inclusion of these consumption adjustments obviously leads to the impacts from the consequential model to be very different from the attributional.

It may be argued that the modelling of rare earth production as having multiple determining products is only necessary because of the rather crude modelling of this activity in ecoinvent, and that a more detailed analysis may show that only one (or fewer) of the products is really determining. However, even if this is the only example of this situation in ecoinvent version 3.1, the principle of the modelling is also relevant in other situations, e.g. the meat coproducts of an abattoir (Consequential-LCA 2015), for which data are not available in ecoinvent version 3.1.

4.4 By-products from treatment activities

When comparing outputs of treatment markets, where the output is the treatment of a by-product, the attributional model only includes the emissions from the treatment activity while the consequential model includes the upstream activities displaced by the by-products.

In Fig. 4, the attributional treatment market for a rape meal (a by-product from operation of a rape oil mill) only has inputs of transport with impacts amounting to 0.35 Pt/Mg of rape meal. In the consequential model, the displaced protein from

Fig. 2 LCIA results for the attributional (*left*) and consequential (*right*) models supplying an output of the speciality product "Graphic paper, 100% recycled". Values are per megagram paper from the ReCiPe Endpoint (H, A) method

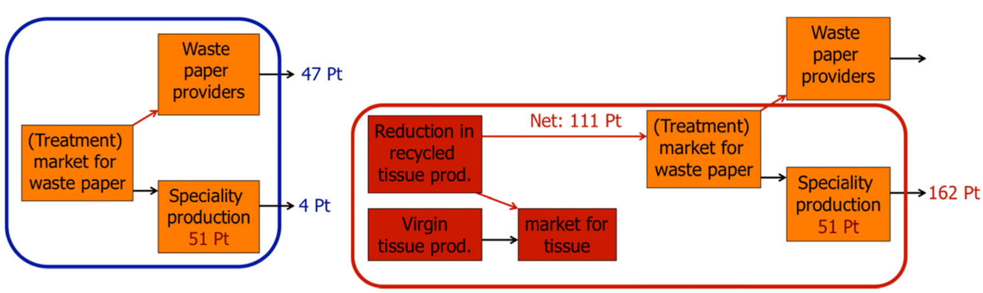
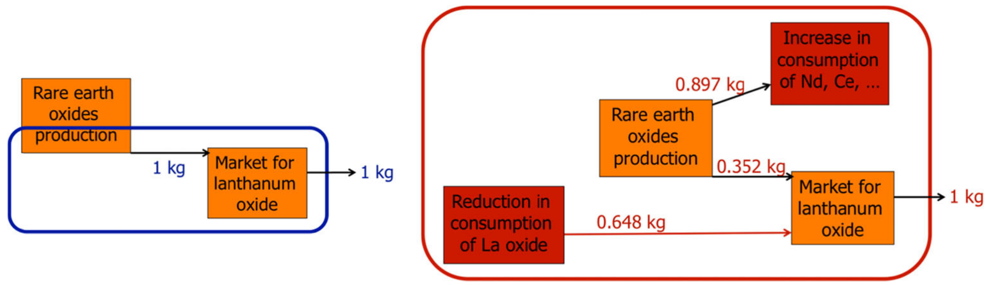


Fig. 3 The attributional (*left*) and consequential (*right*) models for the output "Lanthanum oxide" from rare earth oxides production



soy meal is included, giving a negative (avoided) impact of 140 Pt/Mg rape meal.

However, this comparison is only included because both product systems supply "treatment of 1 kg rape meal" and thus appear as the same product/activity combination in the two ecoinvent system models. But in reality, the comparison is not really meaningful because the two product systems do not play the same role: the consequential result shows the consequences of changing the demand for 1 Mg of rape meal, while the attributional result shows the average treatment required to bring 1 Mg of rape meal to the market.

In these cases, rather than comparing the treatment activities (here, "treatment of 1 kg rape meal"), a more reasonable comparison would be between the allocated outputs of the attributional system (here, rape oil and protein) and their consequential counterparts. Nevertheless, these comparisons still show a noteworthy difference between the two system models: 109 vs. 78 Pt/Mg rape oil and 107 vs. 140 Pt/Mg

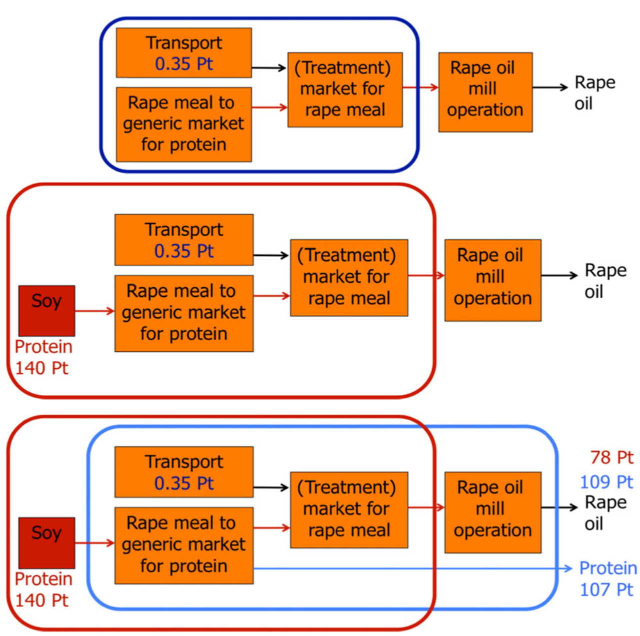


Fig. 4 The attributional (*upper drawing*) and consequential (*middle drawing*) models for the output from the (*treatment*) market for a rape meal, and a comparison (*lower drawing*) of the LCIA results of a rape oil and protein in the two models. Values are per megagram rape meal using the ReCiPe Endpoint (H, A) method

protein for the attributional and consequential impacts, respectively.

4.5 Determining products heavily influenced by by-products

Large differences between consequential and attributional LCIA results may also occur for determining products that are produced together with by-products that displace very polluting or very clean activities compared to the burden that is allocated away from the determining product in the attributional model.

One of the more extreme examples of this is silicon tetrahydride; here, analysed with the ReCiPe Endpoint (E, A) method: the upper part of Fig. 5 shows the attributional result where the majority of the impact is allocated to the relatively valuable by-product silicon tetrachloride, leaving only 11 Pt/kg to the silicon tetrahydride. This allocation also results in only 0.5 kg of silicon input to produce the output of 1 kg silicon tetrahydride. The stoichiometric requirement for the silicon hydrochloration activity is 17.2 kg silicon, as shown in the consequential model (lower part of Fig. 5), which arrives at an impact of 378 Pt/kg silicon tetrahydride, composed of 390 Pt/kg for the sum of all inputs and −12 Pt/kg from the avoided production of silicon tetrachloride. The extreme difference in the results is here caused by the low impact

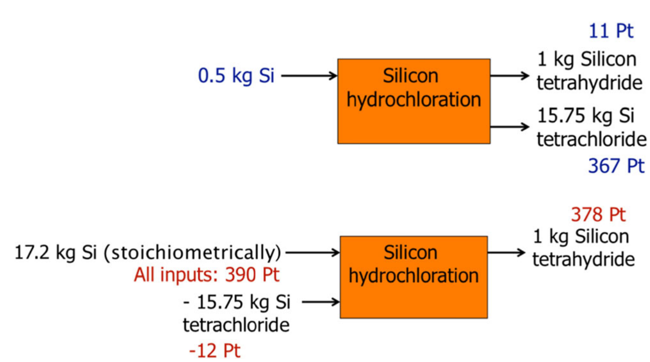


Fig. 5 LCIA results for the attributional (*upper drawing*) and consequential (*lower drawing*) models for the supply of silicon tetrahydride, using the ReCiPe Endpoint (E, A) method

of the marginal production route for the silicon tetrachloride combined with its relatively high value.

5 Conclusions

The comparison between the attributional and consequential LCIA results, for the same products using the same unit process database, isolates the effect of the two modelling approaches. The differences in results are large: for the 401 impact categories and 9860 product/activity combinations analysed, 67% of the results show more than 10% difference, and for 5% of the results the differences are above an order of magnitude.

Although some of the more extreme differences may be due to errors in the database and unreasonable comparisons of product/activity combinations for by-products for treatment with the same names with the same but with very different functions (as demonstrated with the rape meal example), the high frequency of differences above 10% confirms that the use of attributional models for consequential purposes, or vice versa, will imply important errors. The overall error will of course depend on the extent to which attributional modelling is used as part of the overall system model.

The analysis confirms that attributional background datasets are not appropriate as a substitute for consequential background data, especially in situations when:

- The unconstrained (marginal) suppliers have much more/less impact than the average,
- Analysing the use of by-products, especially from treatment activities, for speciality products and from activities with more than one determining product,
- Analysing determining products from activities with important amounts of other coproducts.

The full consequential model supplied with the ecoinvent background database thus removes what has hitherto been an important source of error for consequential results, besides saving LCA practitioners a lot of time-consuming modelling.

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Attributional or consequential Life Cycle Assessment: A matter of social responsibility

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ABSTRACT

Results of Life Cycle Assessment (LCA) are critically dependent on the system boundaries, notably the choice of attributional or consequential modelling. Published LCA studies rarely specify and justify their modelling choices. Since LCA studies are typically performed within the context of social responsibility and product life cycle management, this article investigates the relationship between social responsibility paradigms and the system modelling choices in LCA. We identify three different social responsibility paradigms: Value chain responsibility, Supply chain responsibility and Consequential responsibility. We point out that while there is no generally right or wrong choice of system model, each responsibility paradigm implies a specific matching system model. We then argue that all responsibility paradigms ultimately imply a consequential perspective, namely that of responding to the concerns of the system stakeholders. Although value or supply chains are systems defined without concern for consequences, and thus may include activities that the decision maker cannot influence, the chosen system is still analysed and assessed by accounting for its social consequences, and it is for these consequences that social responsibility is then taken. We argue that it is inconsistent to exclude consequences of own actions (i.e. the consequential system) while including consequences from actions of others in value chain or supply chain. We thus conclude that a consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle and *may* additionally take responsibility for the consequences of other activities in the value chain or supply chain. We end the article with recommendations on reporting on LCA system models that are more specific than those of the current LCA standards.

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1. Introduction

This article presents the results of an investigation into the relationship between social responsibility paradigms and the system modelling choices in LCA. This investigation was motivated by the widespread lack of specification and justification of the modelling choices made in published LCA studies, which leads to confusion when LCAs on the same product arrive at different results and conclusions. Since LCA studies are typically performed within the context of social responsibility and product life cycle management, the results of our investigation may contribute to improved clarity on and justifications of the modelling choices in LCA. The research methods applied are critical review and deductive logic from the first principles of formal ethics.

The current literature, which we review in Section 2, discuss different principles for allocation of responsibility within a value chain, but do this with the implicit premise that the relevant system for which responsibility is taken is the value chain, i.e. the economically allocated fraction of either purchases or sales. However, delimiting the system to that of value chain already implies an allocation of responsibility. Based on the definition of social responsibility and its concept of “sphere of influence” as well as the literature on system delimitation in LCA, we point out that any choice of system boundary necessarily involves the application of a normative rule.

In Section 3, we identify three different system types: value chains, supply chains and consequential product life cycles, each representing a specific social responsibility paradigm: value chain responsibility, supply chain responsibility and consequential

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responsibility, respectively. We describe and exemplify each of these system types and their rules for system boundary closure.

In our discussion, Section 4, we argue that it is possible to take responsibility for more than one system perspective at a time, thus combining systems additively, while the selective use or mixing of different system modelling rules in the assessment of a single system leads to obscuring and shifting of responsibility. We then argue that all responsibility paradigms ultimately imply a consequential perspective, namely that of responding to the concerns of the system stakeholders. We argue that it is inconsistent for a socially responsible decision maker to exclude consequences of own actions (i.e. the consequential life cycle) while including consequences from actions of others in the value chain or supply chain.

We thus conclude, in Section 5, that assessment of social responsibility *must* always include the consequential product life cycle and *may* additionally include consequences of other activities in the value chain or supply chain, and call for increased transparency in reporting of modelling choices and system boundaries for LCA studies.

This paper does not investigate the initial motivations for taking social responsibility, but exclusively investigates the consequences of a consistent application of social responsibility to LCA modelling.

2. Review of the current literature

2.1. Principles for allocation of responsibility within a value chain

In the current literature, allocation of responsibility for environmental impacts has mainly been discussed in the context of the global regulation of greenhouse gas emissions. In this literature, it has been pointed out that the allocation of responsibility to nations exclusively for their direct territorial emissions ignores the important role of final consumption as the main driver for the emissions (Kondo et al., 1998; Eder and Narodoslawsky, 1999; Munksgaard and Pedersen, 2001; Peters and Hertwich, 2008). This *territorial responsibility* (often referred to using the more ambiguous term "producer responsibility") creates perverse incentives for "carbon leakage", including the outsourcing of emissions to countries with lower commitments (Pedersen and de Haan, 2006), and thus for an inefficient allocation of resources. As an alternative to territorial responsibility, Life Cycle Assessment (LCA) and the closely related Environmentally Extended Input-Output Analysis (EEIOA) provide a solution based on *consumption responsibility*, allocating to the consumers of a country all the emissions caused by the production and consumption of that country's *final demand* for goods and services. This implies an incentive for customers to purchase from the value chain with the least impacts and overcomes the problems of the territorial allocation. A third alternative is *income responsibility* (term coined by Marques et al., 2012; first conceptual description by Gallego and Lenzen, 2005), allocating all the emissions generated *downstream* in the value chain, including final consumption and disposal, to the activities that receive income from the value chain, i.e. in proportion to their value added. This implies an incentive for suppliers to compete to supply the sales chain (chain of customers) with the least impacts. As an example, Lenzen and Murray (2010, p. 268) mention:

> "Important contributions to the downstream carbon responsibility of engineers are the emissions they enable through lending their technical skills to run power plants, beef farms, natural gas rigs, and coal mines."

implying – somewhat counterproductively – that, to reduce their personal responsibility, engineers should stop providing services (e.g. of advice on emission reduction) to the most polluting industries, and instead focus on the least polluting.

Although the discussion of these three responsibility principles (summarized in Table 1) has mainly taken place in the context of global regulation of greenhouse gas emissions, the principles are generally applicable to any environmental aspect of production and consumption.

The relevance and fairness of the three principles outlined above and in Table 1 are discussed in the above-mentioned sources and a number of others (Rodrigues et al., 2006, Lenzen et al., 2007; Rodrigues and Domingos, 2008; Marques et al., 2013). These authors suggest different criteria to determine the appropriate combinations of the consumption and income responsibility principles, based on the premise that both of these principles are legitimate, as formulated by Marques et al. (2013, p. 162):

> "After all, for final consumers (households and government) to purchase goods and services, they must first earn their income (wages, taxes or interests) as suppliers of primary factors of production. Thus, it is as legitimate to account for the emissions generated upstream that are embodied in final demand, as to account for emission, which are enabled downstream by primary inputs."

One of the proposed criteria is that the responsibility should be allocated to the actors receiving benefit from the transaction and thus indirectly benefit from the environmental damage. While most authors identify the consumers as the ones receiving the benefit, and therefore settles for consumption responsibility, Rodrigues et al. (2006) and Marques et al. (2012) follow Ferng (2003) and regard both consumption and income as benefits. None of the authors give any consideration to consumer and producer surplus as the normal measure of social benefit from a transaction (Mankiw and Taylor, 2006).

Some of the suggested criteria are mutually exclusive. In the case of Rodrigues et al. (2006) even within the same article, requiring both

> "that environmental responsibility should verify a normalization condition, such that the sum of the environmental responsibility of all agents should equal total environmental pressure"

and

> "the indicator should not display wrong signals, only allowing for a decrease in environmental responsibility of an agent if there was a decrease in overall direct environmental pressure" (both quotes on p. 259)

which are mutually exclusive conditions, since the first refers to - and can only be fulfilled in - a steady-state analysis of environmental pressure and the second condition refers to a change in environmental pressure and can only be fulfilled in an analysis of changes, which is not possible in the analysed steady-state system.

Lenzen et al. (2007) claim that their specific shared responsibility allocation approach will provide the same efficiency incentives as a property rights regulation or a tax/subsidy directly targeting the same impacts. However, since the shared responsibility is arbitrarily assigned from the principles, there is no reason to believe that this would incentivise behavioural change in the same way as economic incentives would. Aside from this unsubstantiated claim by Lenzen et al. (2007), none of the above-mentioned authors address the consequences, effectiveness or efficiency of the two modelling principles with respect to achieving a reduction in environmental damage, most explicitly formulated by Lenzen and Murray (2010, p. 264):

Table 1
Three principles for allocation of responsibility within a value chain.

Responsibility principle	Driver	Activity scope	Impacts
Territorial	Activity	Direct	Direct
Consumption	Demand	Direct and upstream	Direct and embodied
Income	Supply	Direct and downstream	Direct and enabled

> "For the purpose of this article, we take an ex-post perspective, in which actions have occurred, so that the problem of evaluating alternative scenarios does not come up."

Thereby they miss the important point that while a consumer shifting purchase from one unconstrained supplier to another will lead to a consequent shift in production of these suppliers, there is no similar direct consequence of a producer shifting between supplying different customers (if such a shift is at all practically possible), since there is nothing that stops other producers from filling the empty space and supply the abandoned customers, thus resulting in a net zero impact on production and emissions. This asymmetry of power in the value chain is a strong argument for consumption responsibility and against income responsibility, when the purpose is to efficiently stimulate real-life changes.

More importantly for our work here, all the sources mentioned above are based on the premise that the system for which responsibility is taken is the value chain, i.e. the economically allocated fraction of either purchases (consumption responsibility) or sales (income responsibility); see e.g. Rodrigues et al. (2006, p. 259) and Lenzen and Murray (2010, Table 1 on p. 263) for an explicit formulation of this. None of the authors appear to realise that by choosing the value chain as the system under study, they have already made an allocation of responsibility. None of the reviewed articles address the arguments for or the implications of this implicit modelling choice. In this article, we seek to fill this gap by pointing out that *prior* to the discussion on the relevance of the consumption and income responsibility principles and their implied modelling choices, there is a fundamental modelling choice in the delimitation of the system, and that the value chain is only one of these possible delimitations, and not necessarily the most relevant one.

2.2. Social responsibility

The concept of social responsibility has developed from its initial formulation as "corporate social responsibility" (Holme and Watts, 2000, p. 10):

> "the commitment of business to contribute to sustainable economic development, working with employees, their families, the local community and society at large to improve their quality of life"

to the current ISO 26000 definition in which the "corporate" has fallen away, following the realisation that the concept is equally applicable to all types of organisations:

> "responsibility of an organization for the impacts of its decisions and activities on society and the environment, through transparent and ethical behaviour that
>
> - contributes to sustainable development, including health and the welfare of society;
> - takes into account the expectations of stakeholders;
> - is in compliance with applicable law and consistent with international norms of behaviour; and
> - is integrated throughout the organisation and practised in its relationships" (ISO 26000:2010, Clause 2.18)

The term "relationships" in ISO 26000 refers to an organisation's "sphere of influence", defined as the:

> "range/extent of political, contractual, economic or other relationships through which an organization has the ability to affect the decisions or activities of individuals or organizations" (ISO 26000:2010, Clause 2.19)

which is further explained in Clause 5.2.3:

> "This sphere of influence includes relationships within and beyond an organization's value chain. However, not all of an organization's value chain necessarily falls within its sphere of influence. It can include the formal and informal associations in which it participates, as well as peer organizations or competitors. An organization does not always have a responsibility to exercise influence purely because it has the ability to do so. For instance, it cannot be held responsible for the impacts of other organizations over which it may have some influence if the impact is not a result of its decisions and activities. However, there will be situations where an organization will have a responsibility to exercise influence. These situations are determined by the extent to which an organization's relationship is contributing to negative impacts. There will also be situations where, though an organization does not have a responsibility to exercise influence, it may nevertheless wish, or be asked, to do so voluntarily."

In the report of the Special Representative of the United Nations Secretary-General on the Issue of Human Rights and Transnational Corporations and other Business Enterprises, Ruggie (2008, Clause 4):

> "concluded that "sphere of influence" is too broad and ambiguous a concept to define the scope of due diligence with any rigour"

notably because the concept:

> "conflates two very different meanings of "influence". One is "impact", where the company's activities or relationships are causing human rights harm. The other is whatever "leverage" a company may have over actors that are causing harm or could prevent harm. Impact falls squarely within the responsibility to respect; leverage may only do so in particular circumstances." (Ruggie, 2008; Clause 12)

In this article we apply insights from the debate on attributional and consequential modelling in LCA to contribute more formal clarity on the different ways to define the sphere of influence for social responsibility.

2.3. The unavoidability of a normative system delimitation in LCA

All economic activities are interlinked through a global network of product flows,[1] which means that there is no *objective* way to delimit an organisation's sphere of influence as any discrete part of the world's activities. From the perspective of a specific organisation, all other economic activities in the world can be found as contributing a small share to the value chain or supply chain of the organisation. Likewise, when looking at consequences forward in time, the activities of any specific organisation will to some extent contribute to or impact on a large part of the other economic activities in the world. In this network perspective, there is no end to the sphere of influence of an organisation. Any practically applicable delimitation will therefore require the application of a normative rule.

In the context of life cycle inventory analysis (LCI), Ekvall (2000) suggests that different normative rules for system delimitation can be linked to different ethical positions. Notably, teleological situation ethics (consequentialism) can be linked to the effect-oriented (consequential) LCI methodology and deontological rule ethics can be linked to accounting, thus suggesting that the latter may be relevant in a context where the actor wishes to support, be part of, or otherwise be associated with what is deemed to be a "good" system, or to be dissociated with what is deemed to be a "bad" system. However, Ekvall (2000) acknowledges that an application of deontological rule ethics would require an agreement on what is regarded as "being associated with" and therefore concludes that "Further research is required before an operational LCA methodology can be derived from rule ethics". In a comment to Ekvall, Weidema (2003) states:

> "Nevertheless, it is natural that a commissioner of a life cycle study may feel that it is more relevant to study the processes in the immediate supply chain than those actually affected by the product substitutions. It is important to clarify whether the interest of the commissioner is really in the environmental impacts of products (i.e. in LCA) or more in the environmental impacts of the supply chain as such, since the latter interest may be better handled through supply chain management." (Weidema, 2003, p. 18)

In this article, we build on and expand these ideas.

3. Value chains, supply chains and product life cycles

In this section, we describe three different system types (value chains, supply chains and product life cycles) that we have identified as representing three different social responsibility paradigms: Value chain responsibility, Supply chain responsibility and Consequential responsibility (see also Fig. 1). The two first system types (value chains and supply chains) have become known in LCA as attributional models, while the product life cycle is a consequential model. These terms were originally coined at an international workshop on electricity data for LCI in 2001, where it was stated that attributional and consequential modelling are intended to answer different questions (Curran et al., 2005):

- Attributional LCI aims to answer "how are environmentally things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window"?
- Consequential LCI aims to answer "how will flows change in response to decisions"?

The two modelling approaches have been defined in the glossary of the Shonan database guidelines (Sonnemann and Vigon, 2011) as:

- Attributional approach: System modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule.
- Consequential approach: System modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit.

Note that the three system types and their corresponding social responsibility paradigms have no immediate relation to the three principles for allocation of responsibility within the value chain, reviewed in section 2. In the discussion in section 4.1 we will furthermore argue that the discussion on allocation of responsibility is irrelevant for LCA because it is possible to take responsibility for systems from more than one responsibility perspective at a time and responsibility need not be divided. That is, contrary to Gallego and Lenzen (2005), we argue that several actors can assume full responsibility, so that responsibility is not a conserved quantity like mass.

3.1. Value chain responsibility

The term "value chain" was coined by Porter (1985) in the context of analysing competitive advantage. In his definition: "The value chain … views the firm as being a collection of discrete but related production functions, …defined as activities" (Porter, 1985, p. 39). When linked to the value chains of suppliers and buyers, "A firm's value chain is embedded in a larger stream of activities that I term the *value system*" (Porter, 1985, p. 34). While the term "value system" may be the most correct to signify such an interlinked system of activities, it may also easily be confused with the same term used in ethics as a system of consistent values used for the purpose of ethical integrity.[2] We therefore use the term *"value chain"* to signify *a system of interlinked activities that contribute value added to a product*, and implicitly "value chain responsibility" to signify taking responsibility for this system.

In practice, the value chain of an activity is identified by tracing each cost item input to an activity backwards in the chain. The cost for one (purchasing) activity, is a revenue for the supplying activity. For each activity, a part of the revenue leaks out as payments to employees and entrepreneurs, taxes, and rents (together known as "value added"). In a closed steady-state system, all the original revenue must eventually leave the system as value added, thus providing a clear delimitation of the activities included in the system. When summing up all the value added over a value chain, you obtain the "life cycle cost", which is equal to the revenue of the analysed activity (Moreau and Weidema, 2015). When joint production activities are partitioned to obtain the value chain for a single product, the accounting balance (cost = revenue) for each single-product activity is maintained by partitioning each input proportionally to the share that each joint product has in the overall revenue. In LCA jargon this is known as "revenue allocation" or more loosely as "economic allocation" (Ardente and Cellura, 2012). Note that unless there is complete proportionality between the physical properties and the revenues for the joint products, the resulting systems will not be physically balanced (Weidema, 2017b). In general, a value chain will therefore not reflect the

[1] We use the term "product" to denote both goods and services.

[2] We generally use the term value in the meaning of "economic value", i.e. marginal values expressed in monetary units, as opposed to absolute ethical values.

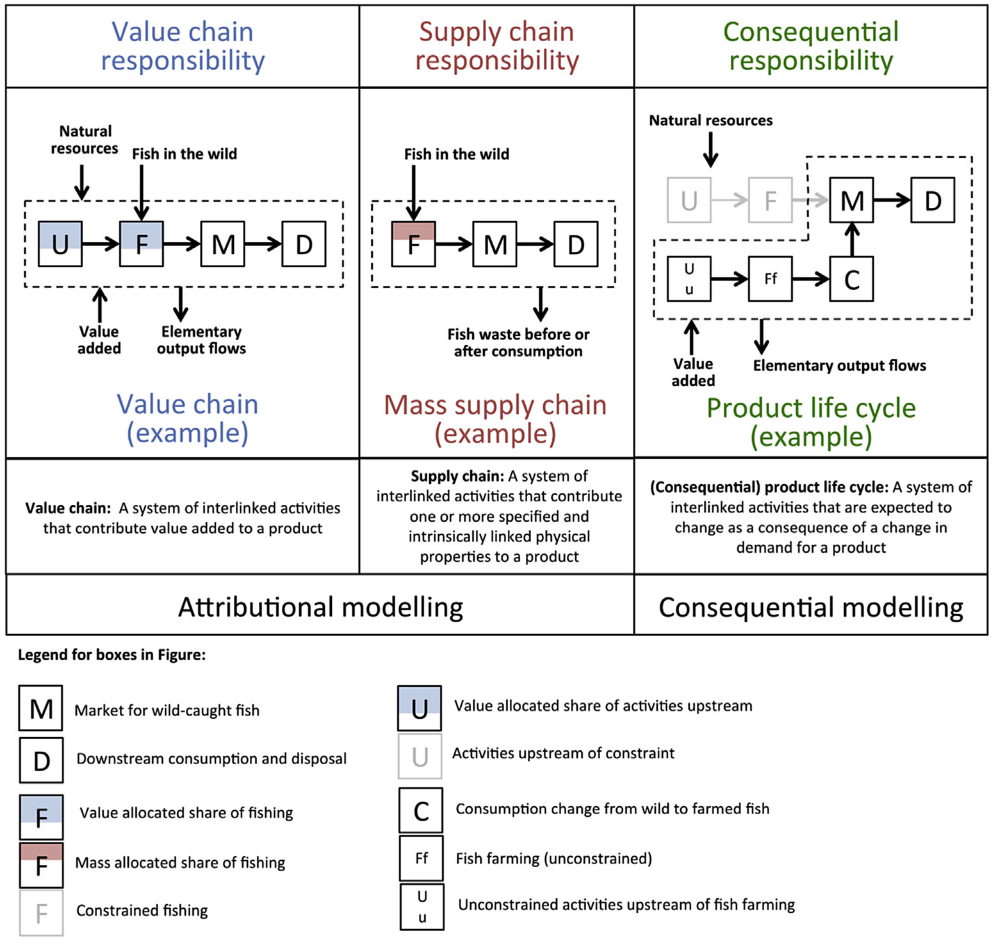

Fig. 1. The three identified responsibility paradigms and the three corresponding system types illustrated by the example of wild-caught fish.

physical causalities of purchasing a product.

As an example of a value chain, we can describe the value chain for wild caught fish: The revenue from the sale of the target fish goes to the fishing activity, which also has a valuable by-catch. The valuable inputs to the fishing, i.e. those that are accounted as cost items, are the different value added items (payments to employees and entrepreneurs, taxes, and rents for the fishing rights) and mainly energy inputs and the fishing gear and the fishing vessel. The non-value added items link to further activities, such as oil extraction, steel manufacture and so on, in the end including a little of practically all activities in the world. For each of the upstream activities, a share is allocated to the target fish output in proportion to the share of the revenue for the target fish in the total revenue (from target fish and by-catch). The value added items of each activity are system external inputs, which are not followed further upstream, and this thus provides closure to the system. Note that also the fish, before being caught, is not followed further upstream, because as a wild resource it has no value beyond the rent paid for the fishing rights, which is a transfer from the user to the (public or private) resource owner and part of the value added.

3.2. Supply chain responsibility

The term "supply chain" is sometimes used as a synonym for what we above called the value chain. However, in its more strict usage, the term refers to the logistic chain of suppliers to an activity, and can be used with reference to both physical supplies and supplies of services or information. It is increasingly being used in relation to traceability, where a customer wishes to know the exact origin of a product (Norton et al., 2014).

Obviously, the physical supply chain of materials and parts may be very different from the supply chain of information, energy or other services to the same activity. For an unambiguous identification, it is therefore necessary to specify a supply chain in terms of the physical (as opposed to economic) properties it supplies, e.g. mass, energy, or a specific elemental content. If two properties are not intrinsically linked, the supply chain of one property may at some point disengage from the supply chain of the other property, and it then becomes impossible to say which of the two chains should be followed. We thus define a supply chain as *a system of interlinked activities that contribute one or more specified and intrinsically linked physical properties to a product*. Obviously, it is possible to overlay several supply chains, depending on the purpose of the analysis, and how many properties the decision maker wishes to take responsibility for.

In practice, the supply chain of an activity is identified by tracing the investigated property backwards in the chain. The input of one activity is either the output of a supplying activity or an external input to the system. In a closed steady-state system, all the input must eventually enter the system as what, in LCA jargon, are called "elementary" flows, i.e. flows that come from the environment without previous human transformation, thus providing a clear delimitation of the activities included in the system. Taking the property mass as example, the sum of all mass inputs over a mass supply chain will equal the mass of the analysed product and all wastes and by-products from the system. When joint production activities are partitioned to obtain the supply chain for a single product, the mass balance for each single-product activity is maintained by partitioning each input proportionally to the share that each joint product has in the overall mass output. In LCA jargon this is known as “mass allocation”. To provide a physically consistent and balanced system, each specific supply chain analysis can only trace one specific property (or several intrinsically linked properties). Note that unless there is complete proportionality between the analysed physical property and the revenues for the joint products, the accounting balance (cost = revenue) will be lost. In general, a supply chain will therefore not reflect the economic causalities of purchasing a product.

We use the mass of wild caught fish to provide an example of a supply chain: The mass of the target fish can be traced back to the fishing activity, again having a valuable by-catch and non-valuable discards. The mass of the wild fish input is the system external input, which is not followed further upstream, and this thus provides closure to the system. The input mass is allocated proportionally between target fish, by-catch and discards on a mass basis. Other inputs to the fishing activity, such as fishing gear, fishing vessel, etc. are often treated as service inputs (the service of providing fishing capacity) and therefore do not provide any net inputs to the mass of the supply chain. While it is obvious that such capital goods are necessary for the production of the primary product, they are not necessary for the mass balance or traceability of the primary product.

3.3. Consequential responsibility

The consequentialist idea that actors should be responsible for the consequences (impacts) of their production or consumption actions is fundamental to environmental management systems, as defined by the ISO 14000 series. These systems organise the management of an organisation's significant environmental aspects, which ISO 14001:2015 (Clause 3.2.2) defines as those aspects that have or can have one or more significant environmental impacts. Environmental impacts are in turn defined as a change to the environment (ISO 14001:2015, Clause 3.2.4).

The ISO 14040-series on LCA reflects this in the description of how product life cycles (product systems) are constructed, with respect to intermediate inputs:

> “The supplementary processes to be added to the systems must be those that would actually be involved when switching between the analysed systems. To identify this, it is necessary to know: (…)
>
> - which of the unconstrained suppliers/technologies has the highest or lowest production costs and consequently is the marginal supplier/technology when the demand for the supplementary product is generally decreasing or increasing, respectively.” (ISO 14049, Clause 6.4)

and with respect to co-production:

> “The study shall identify the processes shared with other product systems and deal with them according to the stepwise procedure:
>
> a) Step 1: Wherever possible, allocation should be avoided by
>   - dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or
>   - expanding the product system to include the additional functions related to the co-products (…)
>
> The inventory is based on material balances between input and output. Allocation procedures should therefore approximate as much as possible such fundamental input-output relationships and characteristics.” (ISO 14044, Clause 4.3.4.2).

What is reflected in these quotes is that the study of (potential) impacts must necessarily imply modelling the *changes* resulting from a (potential) decision. This implies modelling *marginal* changes when the studied changes are small, and *incremental* changes when the changes are larger, as opposed to the average modelling implied in the delimitation of value chains and supply chains.[3] For both marginal and incremental modelling “the supplementary processes to be added to the systems must be those that would actually be involved when switching between the analysed systems” (ISO 14049, Clause 6.4).

In practice, a product life cycle is identified by tracing each required product input, physical or monetary, backwards through the long-term marginal suppliers of each product, i.e., the suppliers that will change their production capacity in response to an accumulated change in demand for the product. The resulting product life cycle is thus demand-driven and based on consumption responsibility, cf. the literature reviewed in section 2 of this article.

When co-productions are modelled as described in the quote from ISO 14044 above, i.e. by subdivision of combined production and by placing dependent joint[4] by-products as negative inputs instead of as positive outputs, thus expanding the product system with the displaced substitutes of the joint by-products, all physical and economic balances remain intact. In parallel to what we found for the value chain, the value added summed over all activities in a product life cycle equals the “life cycle cost”, and the sum of inputs of each balanceable property (such as dry wet and elemental mass as well as energy) will equal the outputs from the system. In general, a product life cycle will therefore reflect both the physical and economic causalities of purchasing a product.

In parallel to what we found for the value chain, the revenue generated by the original demand must eventually leave the product life cycle as value added, thus providing a clear delimitation of the activities included in the product life cycle. The activities included are limited to those that react to the change in revenue, corresponding to the first-order effects of the original spending. Implicitly, when comparing products with different prices, a consequential model will include first-order price rebound effect, while excluding second order effects, such as changes in consumption patterns that may result from the redistribution of the initial spending on the population groups that receive the primary factor income, or second order effects of stimulating specific activities, such as education, research and technological development (Weidema et al., 2015).[5]

We use the demand for wild caught fish as an example of a product life cycle: A marginal increase in demand for wild caught fish leads to a temporary increase in the price of wild fish and thus

[3] It is remarkable that in the extensive value chain and supply chain literature the concept of the *marginal* value or supply chain does not appear to have played any significant role, although this perspective could have important implications also for these fields.

[4] Subdivision by physical causality is only possible for combined production, where the amounts of the co-products can be varied individually, while this is not the case for joint production.

[5] Such multiplier effects can be – and are – included in distributional equity assessments, social life cycle impact assessment, and dynamic economic models for the study of societal developments, i.e. beyond the normal consequential, first order, steady-state modelling of the consequences of small-scale decisions.

to an increase in revenue of the fishing activity. However, under the current conditions, the increase in revenue does not lead to increased output of wild fish, since the total output of wild fish is currently constrained, partly by quotas, partly by the limited physical amounts that are available for harvesting within the current economic limits. Instead, the marginal increase in the price of wild fish leads the marginal consumer of wild fish to leave the market and substitute with a more affordable next-best unconstrained alternative, typically farmed fish. This increases the revenue to the aquaculture production and increases the output of farmed fish correspondingly.[6] The necessary inputs to aquaculture production are then stimulated and can be linked backwards to further activities in parallel to what was done for the value chain. The difference to the value chain analysis is that the inputs are only linked to marginal, unconstrained suppliers, and when co-products are encountered they are dealt with as described in the quote from ISO 14044 above. As for the value chain analysis, the value added items of each activity are not followed further upstream, thus providing closure to the system. Note that the primary fishing activity is *not* a part of the system, in spite of the system having a net functional output of wild caught fish (the demanded wild fish is actually supplied, but this is due to the marginal consumers renouncing their demand for wild fish and not due to changes in outputs from primary fishing).

What this example illustrates is that, in contrast to the average modelling applied for value and supply chains, consequential product life cycles do not include constrained activities, i.e. activities that do not change their output in response to a change in demand for this output. However, an actor may still want to influence such constrained activities, for example when constrained activities provide the most cost-efficient environmental improvements, or when the constrained activity is part of a value chain or supply chain that the actor wishes to take responsibility for. Since constrained activities cannot be influenced by general changes in demand for their products, it is thus necessary to use other instruments that link the improvements to products of unconstrained activities.

An example could be a food company wishing to change their wild fish supply (a constrained product due to quotas and physical/economic limits) from using bottom trawling to using less environmentally damaging alternatives (longlines, gillnets, pots and traps). One option for the food company would be to introduce a label for fish caught by low-impact gear. Since the *shift* in fishing gear is not constrained, the environmental consequence of consumers buying the labelled product will be 1) the difference in impact between the fish caught with low-impact gear and the "normal" trawl-caught fish, and 2) the additional production of the alternative farmed fish (because the change of fishing gear does not in itself mean that more wild fish is caught). Another option that would *not* require a separate label would be for the food company to publicly promise an increase its sourcing from this low-impact source in some proportion to the purchases of the consumer of the general (unconstrained) goods, thus linking the specific desired changes in the constrained activity to the demand for an unconstrained product.

The literal meaning of responsibility implies that it is delimited to what an organisation is *able to respond to* (Power et al., 2008), i.e. meaningfully act upon and change, cf. Section 2.2 on the concept of "sphere of influence". From this perspective, it can be argued that a socially responsible organisation should not be exclusively concerned with the activities within the specific value chains, supply chains, or product life cycles of the organisation, but rather be responsible for influencing what it *can* influence (cf. the concept of "noblesse oblige", i.e. that with wealth, power and prestige come responsibilities). The socially optimal action for any organisation at any specific point in time is to change the specific activities that provide the currently most cost-efficient environmental improvement, no matter whether these activities show up as important within the specific value chains, supply chains, or product life cycles of the organisation. However, this is still a consequential perspective in the sense that it is concerned with change, i.e. with the impacts of the actions of the organisation – it just does not limit these actions to those narrowly related to the product output. If an organisation decides to actively influence an activity that is not part of its product life cycles, this implies spending economic resources that indirectly are dependent on the revenue obtained from the product life cycles of the organisation, and this spending thus indirectly becomes a part of these product life cycles. Here it should be noted that a unit of analysis could be the product portfolio of an organisation, resulting in an aggregate of several product life cycles. Such a portfolio LCA has also been called "an Organisational LCA" (Blanco et al., 2015).

In line with this idea of responsibility for what can be influenced, it can also be argued that inaction, i.e. the omission of action in a situation where action could have been taken, has consequences. If comparing a change to "business-as-usual", the difference between the two can be expressed either as a consequence of a change (action) or as a consequence of inaction ("business-as-usual").

3.4. System cut-offs

As mentioned above, all three system types (value chains, supply chains and product life cycles) have a natural closure, when all of the analysed property has been accounted for as input to the system. The system boundary is thus given by the cut-off rule that the inputs of "elementary flows" are not followed further upstream. No further cut-off rules are required.

However, for some specific delimitations of responsibility, namely in those situations suggested by Ekvall (2000), where an actor wishes to support, be part of, or otherwise be associated with what is deemed to be a "good" system, or to be dissociated with what is deemed to be a "bad" system, Weidema (2003) suggests that an additional cut-off rule could be applied, namely that of tier distance or transactional distance:

> "the concept of "being associated with" is hardly meaningful beyond a few steps backwards or forwards in the supply chain, thus rendering LCA too sophisticated a technique for identifying the relevant associations." (Weidema, 2003, p.18)

4. Discussion

4.1. Mixed or combined systems: shifting or expanding responsibility?

Many published LCAs have applied a mix of allocation rules, combining revenue allocation, physical allocation, and system expansion within the same system, which of course obscures both how the system boundary is identified and what question the study will be able to answer (Weidema, 2017b). As illustrated by our example of wild caught fish, different system models leave out different activities and place different emphasis on those that are

[6] The constraint on wild fish implies that an increase in output of farmed fish can only be obtained by increasing the share of vegetable protein input in aquaculture. It is also possible that part of the consumers will substitute with other food products, such as chicken, which are also based on vegetable protein.

included, so that a skilful mix of modelling rules can make nearly any undesired activity disappear or lose importance.

Selective application or mixing of different system modelling rules within the same system can therefore only be seen as a way of shifting (away) responsibility. But if the different system models are instead used in combination, the possibility arises of taking responsibility for more than one system perspective at the same time, i.e. expanding rather than shifting responsibility. In contrast to the statement of Gallego and Lenzen (2005) that

> "it is intuitively clear that the responsibility for impacts associated with transactions in a productive system has to be somehow divided between the supplier and the recipient of the respective delivered commodity" (Gallego and Lenzen, 2005, p. 366)

we do not see any arguments suggesting that it is not possible to take responsibility for more than one system perspective at a time, nor any arguments suggesting that responsibility has to be divided when two or more actors take responsibility for the same activity. This view was already put forward by Eder and Narodoslawsky (1999). In this view, the allocation of responsibility becomes irrelevant, and what matters is instead what the co-responsible stakeholders can do about the impacts.

The problem with the allocation of responsibility within a value chain outlined in Section 2.1 and in Table 1 seems to be that responsibility is being allocated according to the actors' ability to influence/change, while the impact is being quantified by a steady-state model, rather than by the marginal product life cycle that is necessary for analysing changes/decisions between alternatives. In the latter, the life cycle impact is that of a product and not of a stakeholder: The life cycle impact is the same for all stakeholders, and need not to be divided.

4.2. Responsibility for improvements

Social responsibility and environmental management systems imply a responsibility to seek to improve. But if the focus is on responsibility for improvements, why choose a product perspective in the first place? It is important to distinguish between assessment of impacts of products (whether as value chains, supply chains or product life cycles), and assessments of impacts of industries/activities. Assessment of environmental problems of industries can be made without a product perspective. It is misleading to apply a product perspective when addressing industry/activity problems, because the product perspective leaves out relevant issues either by allocation (attributional) or by looking only at the unconstrained activities (consequential). And some of the important issues that should be tackled may lie in either the parts that have been allocated away or in the constrained parts of the industries that cannot be affected by indirect market-based demand changes, but only by incentivising direct changes. In our example of fisheries, an industry analysis would point to trawling as the most environmentally damaging activity. Studies using the attributional value or supply chain of wild caught fish would both allocate a part of the trawling away to the by-products, while the consequential product life cycle would not include trawling at all, except if included explicitly by the decision maker.

Changes to attributional systems have consequences beyond the system boundaries, i.e. in the parts that have been allocated away, or made less important through averaging. Choosing between attributional systems based on their relative impacts implies a disregard for these consequences, and can lead to sub-optimal decisions and even to increases in overall impacts. For example, an attributional study of cow's milk may conclude that impacts per litre of milk should be reduced by increasing the efficiency of the milk output, e.g. by changes in fodder composition. The result would be that the dairy cow system would produce less meat per litre of milk. Since the demand for meat is unchanged, the reduction in meat output from the dairy cow system would have to be compensated by an increase in dedicated meat production by beef cattle, which could lead to an overall increase in environmental impact per litre of milk compared to the original dairy system that substituted more of the dedicated meat production. This consequence of reduced substitution would not be captured by the attributional system because the meat by-product would have been cut-off by allocation. More examples can be found in Weidema (2017a).

If you want to make an improvement to a constrained part of a system, this implies actions that go beyond the consequential product life cycle, but these actions can still be assessed with a consequential model of the specific change. In our example of fisheries, even though the primary fishing is a constrained activity, and therefore not part of the consequential product life cycle, a consequential model can still be used for a separate assessment of the environmental implications of any specific decisions to *improve* the primary fishing activity, such as the shift from bottom trawling to longlines or gillnets. However, it is important to understand that such an improvement in fishing practices will not in itself change the (marginal) life cycle of wild fish products because primary fishery is not part of the life cycle due to the constraints. To link the changed fishing practices to the consumption of the marginal product, a label is required that implies a commitment of the producers to change fishing practices in proportion to the sale of the labelled product.

5. Conclusion

5.1. The consequences of responsibility

We have identified, described and exemplified three different system types: value chains, supply chains and consequential product life cycles, each representing a specific social responsibility paradigm: value chain responsibility, supply chain responsibility and consequential responsibility, respectively. Any choice between these systems will be *normative,* since the global interlinked nature of economic activities leaves no *objective* way to delimit an organisation's social responsibility to any discrete part of the world's activities. It is possible to take responsibility for more than one system perspective at a time, thus combining systems additively, while the selective use or mixing of different system modelling rules within the same system leads to obscuring and shifting of responsibility.

The literal meaning of responsibility implies a focus on consequences that can be meaningfully acted upon and changed, i.e. responding to the concerns of the stakeholders within the sphere of influence of the organisation. This consequentialist idea is also fundamental to social responsibility and environmental management systems, as defined by the ISO 26000 and ISO 14000 series. Even if a system is defined without concern for consequences, and thus includes activities that the decision maker cannot influence, it is the consequences of the chosen system for which responsibility is taken.[7]

Following from the first principles of formal ethics (Gensler,

[7] This can also be seen from the fact that Life Cycle Impact Assessment, i.e. the modelling of the further impact once the elementary flows have left the system, has always been based on marginal analysis, i.e. the additional impact of an additional unit of the elementary flow.

1996) it does not appear to be consistent for a decision maker to exclude consequences of own actions (i.e., the product life cycle) while including consequences from actions of others in the value chain or supply chain.[8] Thus, we conclude that a consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle and *may* additionally take responsibility for consequences of other activities in the value chain or supply chain. The consequential life cycle can be seen as the "impact" part of the sphere of influence discussed in Section 2.2, while the additional activities included from the value or supply chain can be seen as representing the "leverage" part of the sphere of influence.

Assessments of products (whether as value chains, supply chains or product life cycles) are not adequate for identifying important improvement options (hotspots) outside the narrow system boundaries of these assessments. Assessments of products are relevant once hotspots have been identified within the system you have chosen to be responsible for (value chain, supply chain, product life cycle, or the whole world).

In order to avoid burden-shifting and to fulfil the purpose of LCA as a tool for improvements, identified improvement options need to be compared, in order to identify the one that provides the largest improvement. These comparisons are always comparisons of consequences of implementing each of the options. In this context, it is also important to note that a decision to continue *business-as-usual* can also be modelled as a change relative to ceasing the activity (the zero baseline). This implies that a consequential product life cycle model can also be used to assess a single existing activity or product against the zero baseline. Two such separately developed life cycle models may then later be compared.

5.2. Communication

Both attributional and consequential system models may be difficult to communicate. Consequential product life cycles only include those activities that change as a result of a decision. This is not always the activities that one would intuitively think, which may make the consequential product life cycles appear counter-intuitive until context is communicated and the model is investigated more in detail. Attributional system models (value chains or supply chains) may at first sight appear simpler to communicate because they follow a more static logic. Communication difficulties appear only at closer examination in the form of the:

- Artificial nature of allocated activities (that have no real-life parallel) and systems (that violate the law of conservation of mass and energy).
- Subjective choices of allocation factors and system boundaries that leave out some of the consequences of the production and consumption of the product.

Ideally, an agreement should be made to use the same system model and database for all product assessments. The arguments provided in this article support an agreement on the consequential life cycle as the default system model that *must* always be used, which *may* then be supplemented by additional, specific, value chain or supply chain related activities.

Published LCA studies rarely specify and justify their modelling choices and system boundaries. To make it possible to interpret LCA results meaningfully, it is necessary that the social responsibility context is clearly communicated, i.e.:

- who is responsible for the analysis and its use (decision maker, stakeholder, analyst, shared responsibility)?
- whether the full consequential product life cycle has been included, specifying any product-related consequences that may have been cut off, and
- whether any additional value chain or supply chain related activities have been included, specifying which or with what allocation rules (allocation keys, point of allocation) and cut-off criteria (tier distance, simple contribution, or specific activities for cumulated contribution).

These reporting requirements are more specific than those of the current LCA standards.

[8] The relevant ethical principle can in everyday language be expressed as "Practice what you preach". This is also known from the parable of the mote and the beam (Matthew 7:1–5).

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Avoiding Allocation in Life Cycle Assessment Revisited

Bo P. Weidema and Jannick H. Schmidt

The problem of coproduct allocation has remained one of the most controversial issues in life cycle assessment (LCA), as can be seen from the discussions during the development of the United Kingdom's Publicly Available Specification on carbon footprinting (PAS 2050);[1] the Greenhouse Gas (GHG) Protocol for Product Life Cycle Accounting & Reporting, developed by the World Resources Institute and the World Business Council on Sustainable Development;[2] and the draft guidance document from the EU Platform on LCA (ILCD 2009).

With system expansion, the affected unit processes are scaled up or down, but there is no artificial partitioning, and because the resulting systems are simple sums of the affected unit processes, each of which maintains its physical balances intact, the resulting systems also have all physical balances intact.

We therefore revisit the issue, stressing a key argument for system expansion that has not yet been adequately described in the scientific literature—namely, that allocated systems nearly always fail to maintain mass and energy (and carbon) balances, whereas system expansion by its nature always ensures that mass and energy balances are maintained intact. We also refer to the article by Suh and colleagues (2010) in this issue of the *Journal of Industrial Ecology* (*JIE*), which paves the way for LCA databases to provide equal support for system expansion and any desired allocation key, thus eliminating the excuse that data for system expansion are not available.

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Carbon Balance

First, let us expand on an underilluminated issue—namely, that of the (lack of) ability of allocation procedures to preserve carbon balances (as well as mass and energy balances in general). The chosen system boundaries are likely correct if, first, the mass and energy balances are correct and, second, the balances for elements such as carbon are correct. In this context, *correct* means zero—that is, what comes in must go out, when one takes into account use and build-up of stocks.

Here, it is interesting to note that system expansion always ensures mass and energy balances, whereas allocation nearly always fails this test. In brief, allocation breaks up the original system into two or more artificial systems according to an allocation key, and the only balance that remains intact in the resulting systems is that given by the allocation key. With mass allocation, the mass balance remains intact, but energy and elemental balances are skewed; with economic allocation, none of the physical balances remains intact except if by chance a physical parameter follows the price of the products. With system expansion, the affected unit processes are scaled up or down, but there is no artificial partitioning, and because the resulting systems are simple sums of the affected unit processes, each of which maintains its physical balances intact, the resulting systems also have all physical balances intact.

Table 1 Obtaining two single-output systems, meat and milk, from the multi-output system "dairy cow" by allocation or system expansion

	Original systems		*System expansion*		*Dry matter allocation*		*Economic allocation*	
	Dairy cow (1)	*Meat cattle (2)*	*Milk = dairy cow – meat cattle (3)*	*Meat = Meat cattle (4)*	*Milk 81% (5)*	*Meat 19% (6)*	*Milk 77% (7)*	*Meat 23% (8)*
Feed DM	100.0	26.0	74.0	26.0	81.0	19.0	77.0	23.0
Milk DM	−9.3		−9.3		−9.3		−9.3	
Meat DM	−2.2	−2.2		−2.2		−2.2		−2.2
CH_4	−2.0	−0.5	−1.5	−0.5	−1.6	−0.4	−1.5	−0.5
Manure DM	−23.2	−4.7	−18.5	−4.7	−18.8	−4.4	−17.9	−5.3
C in CO_2	−28.3	−8.1	−20.2	−8.1	−22.9	−5.4	−21.8	−6.5
Respiratory water	−35.0	−10.5	−24.5	−10.5	−28.4	−6.6	−27.0	−8.1
Mass balance	0.0	0.0	0.0	0.0	0.0	0.0	−0.4	0.4
Feed C	46.5	12.0	34.5	12.0	37.66	8.84	35.8	10.7
Milk C	−5.3		−5.3		−5.3		−5.3	
Meat C	−1.0	−1.0		−1.0		−1.0		−1.0
CH_4-C	−1.5	−0.4	−1.1	−0.4	−1.22	−0.29	−1.2	−0.3
Manure C	−10.4	−2.2	−8.2	−2.2	−8.42	−1.98	−8.0	−2.4
C in CO_2	−28.3	−8.4	−19.9	−8.4	−22.92	−5.38	−21.8	−6.5
C balance	0.0	0.0	0.00	0.0	−0.2	0.2	−0.5	0.5

Note: DM = dry matter; CH_4 = methane; C = carbon; CO_2 = carbon dioxide.

Let us take a well-known example: A system with one feedstock input, feed, and two product outputs, milk and meat. For convenience, we may call the system "dairy cow." A few facts about the system are given in the first column of table 1. If we use the dry matter input of feed as a reference, this dry matter leaves the system as follows: 9.3% is milk and 2.2% is meat; 2.0% is emitted as methane; 23.2% leaves the system as manure, which, for simplicity, we treat as a waste in this example; 28.3% leaves the system as carbon in respiratory carbon dioxide (CO_2); and 35% is emitted as respiratory water. It may seem illogical that a dry matter balance contains water, but this is due to the nature of the feed input (roughly $C_6H_{12}O_6$), which releases water as part of the system metabolism. The carbon contents of feed, milk, meat, and manure are 46.5%, 57.0%, 44.0%, and 45.0%, respectively, on a dry matter basis. We thus have 46.5% carbon input in feed, which leaves the system with 5.3% in milk and 1.0% in meat, 1.5% in methane, 10.4% in manure, and 28.3% in respiratory CO_2.

For system expansion, we need the additional system "meat cattle," which is displaced by the additional output of meat from the dairy cow. The mass and carbon balances for meat cattle are given in column 2 of table 1, scaled to the output of meat from the dairy cow in column 1. System expansion implies that the original dairy cow is assigned entirely to the determining product—that is, the milk system—and that one must subtract the production of the corresponding amount of dependent by-product—that is, the meat from the alternative production route, the system "meat cattle." The meat system is exclusively assigned the meat cattle route; that is, in the expanded system, no meat comes from the dairy cow. Because both dairy cow and meat cattle are systems with complete mass, energy, and elemental balances, these balances are also maintained in the resulting single-product systems, as can be seen in columns 3 and 4 in table 1.

If we instead use an allocation key to partition the dairy cow, columns 5 to 8 in table 1 shows what happens:

- If we use the dry mass of the outputs as allocation key, we will obtain a partitioning of the inputs and outputs of the dairy cow with a ratio of 81.0% to the milk and 19.0% to the meat. The system "milk" will thus have an input of 81.0% of the feed. Exactly because *mass* is the allocation key, the *mass* balances of the allocated systems are correct. Carbon in and out do not balance, however: Of the 46.5% carbon into the dairy cow, 37.66% enters the milk system, but 37.86% leaves the system, for an imbalance of 0.2%. Similarly, the meat system has 8.84% in but 8.64% out.
- If we use the economic value of the output as the allocation key, we will obtain a partitioning of the inputs and outputs of the cow with a ratio around 77.0% to the milk and 23.0% to the meat, depending on local market conditions. Table 1 clearly shows that neither mass nor carbon balances are maintained. The only balance that fits is the economic one: The milk system has 77.0% of the income from product sales and 77.0% of the expenditures for feed inputs.

We have on purpose chosen an example in which the imbalances are small. We could easily find more extreme examples—for example, we might use wet mass as an allocation key or use economic allocation on a product system for which the relative prices of the coproducts deviate more from the relative masses (e.g., high-quality timber and wood residuals for pulp or biofuel). The examples above, however, should be adequate to illustrate the point: The only balance that remains intact in allocation is the balance of the allocation key.

Likewise, we could easily find more complicated examples of system expansion, in which more processes come into play, such as transport, intermediate treatments, downstream differences between the displacing and displaced products, and displaced products with further coproducts, as described by Weidema (2001a, 2001b, 2003). In all such cases, however, we still operate with fully intact processes that are simply scaled up or down. Because there is no partitioning, mass, energy, or elementary balances will always remain intact.

System Expansion in LCA Databases

It is interesting to note that the two technology models most widely used by input–output economists to convert supply–use matrices into direct requirement matrices are identical to the two methods for coproduct allocation most favored by LCA practitioners—namely, economic allocation and system expansion (Suh et al. 2010). The industry technology model (so called because it assumes that all products produced by an industry have identical inputs and outputs per monetary unit) uses the proportional value of the coproducts as the allocation key—that is, identical to the economic allocation of LCA. Applying the commodity technology model (so called because it assumes that each commodity is produced by a unique technology, with the same inputs and outputs, irrespective of which industry it is produced in) gives the same results as applying system expansion in LCA, where the inputs and outputs for the main product result from subtracting the inputs and outputs that relate to the production of the by-product from its alternative (main or marginal) production route.

Thus, if an LCA database stores process data unallocated in a format corresponding to supply–use tables, any desired technology model can be applied to these unallocated data, and the database is just as applicable for producing analytical models with system expansion as analytical models with any desired allocation key.

Conclusion

We restate the main arguments for the ISO 14044 (ISO 2006) requirement for applying system expansion whenever possible: Only system expansion consistently fulfils the two further requirements of ISO 14044—that all significant processes that are affected should be included, in this case by a change in the amount of coproducts, and that all systems should yield comparable product outputs, which is ensured in system expansion by subtraction or balancing of the processes that provide product outputs that do not occur in all of the compared systems. As Weidema (2001a, 2001b) noted, these two important requirements are, in general, not fulfilled by allocation. First, allocation does not consider the extent to which a change in the amount of the coproducts affects the functional output and other exchanges of the coproducing process. Second, allocation ignores the effects that a coproduct may have on the further fate of the other coproducts—that is, displacement effects and additional treatment of the coproducts before displacement takes place. Thus, traditional coproduct allocation only fulfills the above two requirements in those particular instances in which the allocation factors are chosen to reflect the way the coproducts actually affect the coproducing process and in which there are no significant effects on the further fate of the other coproducts. In such instances, allocation may be regarded as a special instance of system expansion. The availability of hybrid LCA databases with consistent implementation of system expansion, according to the models provided by Suh and colleagues (2010), resolves the problem of data for system expansion.

Notes

1. www.bsigroup.com/Standards-and-Publications/How-we-can-help-you/Professional-Standards-Service/PAS-2050
2. www.ghgprotocol.org/

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Has ISO 14040/44 Failed Its Role as a Standard for Life Cycle Assessment?

Bo Weidema

The role of a standard is to provide uniform rules for a procedure, so as to minimize or eliminate unnecessary variation in its performance, typically with the aim of reducing costs. The basic standard for performing life cycle assessment (LCA) was published by the International Organization for Standardization (ISO) as ISO 14040/41/42 in 1996 and reorganized, largely unchanged, into ISO 14040/44 in 2006.

In recent years, we have seen a proliferation of guidelines that interpret the basic ISO 14040/44 standards for LCA, either for a specific geography as, for example, the BPX 30-323 for France and the Product Environmental Footprint Guideline for the European Union, a specific sector with the so-called product category rules (PCRs) that seek to regulate the production and communication of LCA information for products within the product category, or a specific topic as in carbon or water footprints.

Unfortunately, these guidelines, as currently published, sometimes cover the same product categories and markets without adequate or reasonable justification, and they reflect different interpretations of ISO 14044 with respect to system boundaries (cut-off rules, which unit processes to link to specific inputs, and rules for handling coproducts).

Interestingly, all these interpretations claim to be based on—if not directly to be in accord with—ISO 14040/44. If this is really true, it points to a serious failure of ISO 14040/44 to fulfill its role as a standard, that is, to minimize or eliminate unnecessary variation.

Different application areas, whether geographical, sectoral, or related to specific impact categories, may indeed require different kinds of data, specific definitions of functional units to make comparisons fair, and specific impact assessment methods. These are all issues that are *not* regulated in ISO 14040/44 and where specific guidelines are therefore relevant. Yet, all the current specific guidelines also specify further requirements for the *life cycle inventory modeling* that deviate more or less from the ISO 14044 requirements. These disparities in interpretation of the ISO 14044 standard are *not* caused by differences across geographies, product groups, or impact categories and cannot be justified by reference to scientific disagreements. At best, the disparities can be explained by the vagueness of ISO 14044 on key methodological points, which makes it possible for everyone to get away with their own interpretation. Thus, a more unambiguous wording of ISO 14044 could help to reduce the current disparity in LCI modeling requirements in the geographical, sectoral, and impact-specific guidelines.

The proliferation of PCR programme operators and guidelines with different definitions and rules for the same product groups, and different interpretations of the basic LCA standards, lead to increased costs for industry seeking to comply with these guidelines, and increased confusion among end-users of the environmental information, who in the end are paying for all the diversity, without any added value or benefit to the environment

The most critical vagueness in the current ISO 14044 relates to which unit processes to include in a product system and how to link these unit process data sets together. This has given rise to different interpretations, notably the attributional and consequential interpretation:

- An attributional product system is composed of the activities that *have contributed* to the production, consumption, and disposal of a product, that is, tracing the contributing activities backward in time (which is why data on specific or market average suppliers are relevant in such a system).
- A consequential product system is composed of the activities that are *expected to change* when producing, consuming, and disposing of a product, that is, tracing the consequences forward in time (which is why data on marginal suppliers are relevant in such a system).

Several different attributional product systems can be constructed for the same product, depending on whether the activities included are selected for their contribution to the cost of the product, the mass of the product, or some other selected

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contribution. The most common procedure is to trace the monetary flows between unit processes, using revenue allocation: an allocation where all economic expenditures in a unit process are allocated to all outputs in proportion to the revenue generation of the latter. This system model thus answers the following question: "What are (the environmental impacts related to) the activities that contribute to the cost of the product?" An issue that is often overlooked is that revenue allocation makes it impossible to maintain the mass and elemental balances of the resulting product systems (Weidema and Schmidt 2010). This, of course, complicates the interpretation of what is meant by "the environmental impacts related to ... the product."

When mixing allocation keys (mass, revenue, 100% to determining products, and so on) within the same product system, it becomes obscure which question the model is supposed to answer. Mixing consequential and attributional modeling (marginal vs. average/allocated) within the same product system, of course, has the same problem as when mixing allocation keys: It becomes unclear which question the analysis is supposed to answer.

Consequential modeling is based on the description in ISO 14049, clause 6.4 (original without italics): "The supplementary processes to be added to the systems must be those that would actually be involved when switching between the analysed systems. To identify this, it is necessary to know:

- whether the production volume of the studied product systems fluctuate in time (in which case different submarkets with their technologies may be relevant), or the production volume is constant (in which case *the base-load marginal is applicable*),
- (...) whether (...) the inputs are delivered through an open market, in which case it is also necessary to know:
- whether any of the processes or technologies supplying the market are *constrained* (in which case they are not applicable, since their output will not change in spite of changes in demand),
- which of the unconstrained suppliers/technologies has the highest or lowest production costs and consequently is *the marginal supplier/technology* when the demand for the supplementary product is generally decreasing or increasing, respectively."

Because technological, coproduct, or market constraints are often found in practice, the results of consequential systems often differ substantially from the corresponding attributional systems based on average, allocated inputs. It is therefore of utmost importance to add to ISO 14044 the more detailed description of the consequential modeling from ISO 14049 and other, more recent specific descriptions, such as the one describing the implementation in ecoinvent v3 (Weidema et al. 2013). It would also be helpful to add to the ISO 14044 a clear, complete, and unambiguous description of at least one attributional model, and to clarify the difference to the consequential model, especially specifying for which questions each model is relevant.

The issue of the differences in system models cannot be separated from the issue of the so-called allocation hierarchy in clause 4.3.4.2 of ISO 14044, which provides a step-wise procedure for handling coproduction, that is, for reducing multiproduct systems to single-product systems. This allocation hierarchy is being interpreted differently by different practitioners, which can be referred back to a lack of clarity in the initial sentence:

> "The study shall identify the processes shared with other product systems and deal with them according to the step-wise procedure presented below.
> a) Step 1: Wherever possible, allocation should be avoided by ... "

Some practitioners focus on the "should" in the last sentence, which allows them to interpret the allocation hierarchy as a recommendation, that is, not as a requirement, so that any allocation rule can be applied as long as it is made explicit. Other practitioners focus on the "shall" in the initial sentence and interpret the quote as a requirement always to avoid allocation (by subdivision and system expansion), because this is always possible by one of these approaches. The proponents of the "shall" interpretation point out that a more liberal interpretation would be at odds with the very purpose of providing a standard, because a liberal interpretation would allow practitioners to allocate as much or as little as they may desire of the total environmental impacts to the product system of interest.

The allocation hierarchy could be significantly simplified if a separate description was made for situations of combined production with variable relationships between the coproducts, such as found in petroleum refineries that can regulate the outputs to meet variation in demands, and situations of joint production with fixed relationships between the coproducts, such as found in chlor-alkali electrolysis. This would clarify that step 2 in the ISO hierarchy is exclusively intended for situations of combined production, but equally applicable in consequential and attributional models, and would correct the common misunderstanding that this step is identical to allocation according to mass or energy, a misunderstanding that has become more common after the explanatory sentence "The resulting allocation will not necessarily be in proportion to any simple measurement such as the mass or molar flows of coproducts" fell out in the transcription of the text from ISO 14041 to ISO 14044. For situations of joint production, substitution (system expansion) would be the only relevant option for consequential modeling, whereas a separate option should be provided for attributional modeling. Probably, at least revenue allocation should be described (maybe with additional rules for how to deal with prices that fluctuate over time and place, currency conversion, taxes and subsidies, nonmarket products, the point of allocation, value correction, and whether any of the resulting mass imbalances should be corrected for). Any other allocation key than revenue will have the problem that it cannot be applied consistently to all cases of coproduction, which would then require additional rules for when to apply which allocation key.

There is a different, but related, conflict in interpretation with respect to the allocation rules for recycling, both among practitioners and guidelines. The difference in interpretation is related to—or at least amplified by—an ambiguity in ISO 14044, clause 4.3.4.3, which, on the one hand begins, with the sentence "The allocation principles and procedures in 4.3.4.1 and 4.3.4.2 also apply to reuse and recycling situations" but, on the other hand, provides an additional allocation hierarchy for reuse and recycling with a *different* order than the one in clause 4.3.4.2. The simplification of the allocation hierarchy suggested in the previous paragraph would eliminate the need for any separate rules for recycling.

The proliferation of PCR program operators and guidelines with different definitions and rules for the same product groups, and different interpretations of the basic LCA standards, lead to increased costs for industry seeking to comply with these guidelines, and increased confusion among end users of the environmental information, who, in the end, are paying for all the diversity, without any added value or benefit to the environment, thus further reducing the likelihood that we will identify truly sustainable solutions and implement them efficiently. This cannot be in the long-term interest of industry, consumers, and society at large.

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Ecoinvent database version 3 – the practical implications of the choice of system model

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Abstract In the new version 3 of the ecoinvent LCI database, the user can choose between different system models that reflect different LCI modelling algorithms applied to the same underlying unit process data. The system models reflect different consequential and attributional models, i.e. linking of inputs to either average or unconstrained suppliers, and arriving at single-product systems either by partitioning (allocation) according to different allocation properties or by substitution (system expansion). New approaches, such as the ILCD handbook's recommendation of system expansion with average flows, can also be accommodated by the new, flexible database structure. The presentation explains the new linking structure of the database that allows this flexibility and the structure of the workflow in database management. The practical implications for the end users of the different system models are highlighted.

1 System models in ecoinvent database version 3

Since its first publication in 2003, the ecoinvent database has become the most widely used background database for life cycle inventory (LCI) data. The harmonised guidelines for data collection and modelling developed by the project "ecoinvent 2000" have been an inspiration for many later national database guidance documents. After the successful update of the ecoinvent database to version 2 in 2007 it was decided to plan for a major re-structuring of the database for its version 3, due for publication late 2011, in order to ensure further expansion of its geographical and technological reach and detail, as well as enhancing its flexibility for use in different modelling contexts.

LCI modelling is composed of two elements: *Unit process modelling* and the linking of these unit processes in *system modelling*. The focus of the following is on the latter aspect, and especially the different LCI system models that will be supported by the ecoinvent database version 3 [1].

From the same unit process data, different LCI system models can be constructed that differ in two aspects:

- The linking of inputs to either average or unconstrained suppliers.
- The procedures to arrive at single-product systems in situations of joint production of products, which apply either partitioning (allocation) of the multi-product system into two or more single-product systems, or substitution (system expansion), which eliminates the by-products by including the counterbalancing changes in supply and demand on the affected markets.

A large number of different system models can be generated by combining these two modelling choices, which can be further modulated by varying the extent of constraints on suppliers and varying the allocation keys chosen for partitioning, respectively.

However, only a few combinations and variations are demanded in practice, where it is common to distinguish between consequential and attributional modelling:

- The *consequential* system model focuses on the long-term consequences of decisions, i.e. consequences that include changes in production capacity (capital equipment). This model therefore applies substitution (system expansion) consistently, and excludes suppliers that are constrained in their ability for capacity changes.
- The attributional system models all link inputs to the average suppliers, i.e. in proportion to their current production volumes, and therefore mainly differ in the way they arrive at single-product systems. In accordance with common practice, the ecoinvent database version 3 has one system model that consistently applies partitioning with the *revenue* obtained from the co-products as allocation key.
- A slight variation of this is a system model that applies instead *"true value"*, which is a modification of the revenue to correct for situations where the revenue does not reflect the "true value" of the co-products. This system model is also planned to have an allocation correction for carbon, see below.
- Finally, the system model that the ILCD Handbook [2] recommends for decisions in the so-called *situation A*, where changes are not expected to affect production capacity, links the inputs to the average suppliers, except for suppliers of by-products, which are constrained. This system model also applies substitution, and is therefore only a slight variation of the consequential system model, but is nevertheless classified by its authors as an attributional model, probably due to its linking to average suppliers. However, unlike other attributional models the sum of the

product systems after substitution does not equal the system before substitution.

System models that apply revenue or true value partitioning have the drawback that they do not provide correct mass and elemental balances for the product systems. Systems that are partitioned will only be balanced for the property that is applied as allocation key. For this reason, it is planned to add an allocation correction for carbon to the system model with "true value" partitioning. An allocation correction is two datasets that counterbalance each other, re-allocating one or more environmental exchanges, so that the resulting allocated product systems have correct mass balances for the re-allocated exchanges. The rationale for applying the corrections only to carbon is that for carbon, in contrast to most other elements, the same substance as both input (capture of carbon dioxide from air) and output (carbon dioxide to air) has the same significant environmental impact pathway (change in the atmospheric concentration).

2 Database structure and modelling algorithms

A central prerequisite for the new flexible database structure is the systematic introduction of separate names for activities and their products, and the introduction of market datasets for all products. It is through these markets that the inputs and outputs of all other datasets are automatically linked by the database modelling algorithms, unless the markets are deliberately circumvented by manually linking to specific suppliers. In its simplest form, a market dataset consists of a reference product, representing a consumption mix, and one or more inputs of the same product from the different transforming activities that are located within the geographical delimitation of the market. For each product at least one global market dataset is available. Depending on the product, this may then be sub-divided into geographically specific market datasets.

The algorithm for linking the intermediate inputs of an activity is to link to the local market activity dataset that supplies this input as its reference product. The local market activity dataset is identified by matching the geographical location of the activity with the available market for this location. Since markets do not overlap, there will generally be one and only one such market activity for each intermediate input. If the activity is defined for a geography that spans over more than one market, each of the market activities contribute in proportion to their production volume, implying that the intermediate input will be duplicated to match the number of supplying markets and the amount of the intermediate input will be divided over these in proportion to the production volume of each market.

The algorithm for linking the inputs of each market activity is to link to all those transforming activities within the geographical area of the market activity, which have the market reference product as an output, in proportion to their available production volume. It is here that different constraints on the availability of a supplying dataset can be taken into account.
The linking algorithms can specify one or more of the following constraints to be taken into account, leading to the exclusion of the constrained supplies from the market:

- *By-product constraints*, implying that only reference products (determining products) will be included in the supply.
- *Technological constraints*, implying that only suppliers with a specified technology level will be included as suppliers.
- *Market constraints*, implying that the corresponding input to the market is resulting from a reduction in consumption in another activity.

The consequential system model takes all three types of constraints into account while the ILCD situation A model only takes the first one into account. The other attributional models do not take any constraints into account.
The introduction of market activity datasets for all products implies that the naming of products become more important, because they define the markets and what products are included in the same consumption mix and thereby also what products can substitute each other. This implies that the same procedures that are relevant for defining the functional unit in a life cycle assessment is now relevant already when embedding an activity dataset into the database. Therefore, the description on how to correctly identify market boundaries and functional units (here: reference products) now has a much more prominent position in the ecoinvent Data Quality Guidelines [1] than hitherto.
For all system models, the linking algorithm identifies materials for treatment (by-product/wastes which are not provided as positive reference products of any other activity in the same geographical area), and move these to be negative inputs of the same activities, in order to include the treatment activities for the materials into the product systems. Since a negative input is the same as a positive output, this operation does not affect the mass, energy and monetary balances of the activities.
Finally, the linked, multi-product datasets are converted to single-product datasets, either by partitioning or through substitution.
Partitioning involves the generation of as many single-product datasets from each multi-product dataset as the dataset have products with the specified allocation property. For each of the single-product datasets, the original inputs and elementary outputs without the allocation property are multiplied by the ratio of the specified allocation property for the product (when multiplied by the amount

of the product) relative to the sum of this (multiplied) property for all outputs. This procedure is also known as co-product “allocation”.

Substitution is implemented by moving the by-products from being outputs of the multi-product activity to be negative inputs of this activity and linking this negative input to its local market, in the same way as described above for all other intermediate inputs. Note that by-products and wastes for which substitutes are not available have already been placed as materials for treatment by the procedure described above. This implies that for the remaining by-products there will always be an activity that supplies the by-product as its reference product, and which will therefore be displaced when an additional amount of the by-product from the multi-product activity is supplied to the market.

3 Workflow in the database management

The new database structure implies a number of changes in the procedures for data providers and the database management.

Data providers no longer have to link datasets to suppliers, but only have to indicate the inputs by their product name.

In general, data providers no longer have to supply allocation factors, as these are now autogenerated from the relevant product properties, which in turn will be suggested by the relevant editors if omitted in a submitted dataset. The exception is if the price of a product is not reflecting its "true value", in which case the "true value relation" has to be specified as a property by the data supplier.

Data providers *do* have to specify one of the product outputs as the reference product (determining product), as opposed to a by-product/waste, which is normally straightforward. The ecoinvent Data Quality Guidelines [1] provide further guidance for situations where more than one product appears to be the reference product. As mentioned above, correct naming of intermediate outputs is now more important, since the name will determine which market the output will contribute to.

Data providers no longer have to specify whether an output is a by-product or a waste, since materials that require treatment are automatically identified by the database, as mentioned above.

Data providers are asked to consider specifying the technology level of transforming activity datasets as outdated, old, current, modern or new. If not specified by the data provider it will be assigned the default "current". Since the technology level is used by the linking algorithm to identify technology constraints, this may affect whether the activity is included or excluded from

contributing to the consumption mix of the consequential model. The specific algorithm for technology level depends on the market trend: When the production volume of the reference product is decreasing more than 3.33% annually, the activity is identified as unconstrained if its technology level is “old”, and when the production volume of the reference product is decreasing less than 3.33% annually, increasing, or stable, the activity is identified as unconstrained if its technology level is “modern”. If there are no supplying activities with the required technology setting, the requirement for “modern” is replaced by ”new”, and “old” is replaced by “outdated”, and if these do not exist, by the option “current”. The 3.33% is derived from a generalised lifetime of 30 years for production equipment.
Data providers normally do not have to supply market datasets. If a new product is added to the database, a global market will be autogenerated by default. Only markets with more specific geographical locations or constrained markets have to be submitted separately. Constrained markets are modelled by adding a conditional exchange, i.e. an exchange that is only activated for the consequential system model, representing the amount of product that is resulting from reduction in consumption. The conditional exchange is added as a negative by-product output with the same name and unit as the reference product, and with a direct link to the consumption activity that supplies the reduction in consumption.
After approval by the review panel of the Editorial Board, the unlinked, multi-product datasets become part of the working version of the database, from which they are publicly available. By application of the above described linking algorithms, the unit processes are automatically linked to the rest of the database for each of the system models. The linked united processes and the fully linked systems are then ready to be provided to LCA software users, without any requirements that the software have implemented the linking algorithms. Finally, matrix inversion is applied to the linked systems so that the accumulated LCI results for each system model can also be provided to end users without own LCA software.

4 Practical implications for end users

With the new flexible database structure the conscious choice of system model comes within reach of the ordinary database end user. It also becomes possible to compare results of modelling the same product in different system models, with the same underlying data. This means that discussions on choice of system model can be separated from discussions of differences in data between the models.

It should be noted that the choice of system model is not an arbitrary choice. Different system models may be relevant for different purposes, while for a specific purpose typically only one model is relevant. Thus, not all system models may be of interest to all users.

Different system models can vary significantly in their results for the same products. The data for average suppliers may be very different from the data for a modern or old supplier. For example, chlorine production by the mercury cell process (old), the diaphragm cell process (current), or the membrane cell process (modern) gives significant different results. As activity datasets describing average production conditions are increasingly being replaced by separate datasets for current and modern technologies, the difference between models using average suppliers and models using unconstrained suppliers can be expected to increase. Also excluding a by-product from an average, as in the ILCD situation A model, can be significant when the by-product makes up a large part of the market, as for example in the markets basic metals like aluminium and steel.

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THE FUTURE OF ECOLABELS

Relevance of attributional and consequential information for environmental product labelling

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Abstract

Purpose Considering the general agreement in the literature that environmental labelling should be based on consequential modelling, while all actually implemented environmental labelling schemes are based on attributional modelling, we investigate the arguments for this situation as provided in the literature, and whether a dual label, representing on the same label the attributional and consequential results for the same product, can be a relevant solution or at least contribute to a more informed discussion.

Methods We developed a dual label for three hypothetical, comparable products and presented this for a small test audience, asking three questions, namely "Which product would you choose?", "Was the attributional information useful?" and "Would you accept to have only the attributional information?"

Results and discussion From this small pilot exercise, it appears that informed consumers may have a strong preference for consequential information and that the main problem in communicating consequential results is that they are perceived as less trustworthy and more uncertain due to the fact that the consequences are located in the future. It thus appears important to build into a consequential label some increased level of guarantee of future good behaviour.

Conclusions We propose to apply the above questions to a more statistically representative audience to confirm or refute the findings of this little test exercise.

Keywords Additivity · Consumer acceptance · Environmental labelling · Past environmental impact · Product comparisons · Product improvements · Scale of decision · Uncertainty

1 Introduction

Attributional and consequential models in life cycle assessment (LCA) provide different perspectives on the same products, reflecting differences in the purpose of the assessment. The two modelling paradigms can lead to very different results when applied to the same product, see, e.g. Weidema (2017) and Schmidt and de Saxcé (2016).

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In most applications of LCA, the information is ultimately intended for making improvements to the studied systems and the applied model should therefore compare the consequences of choices between different improvement options. The ILCD Handbook also recommends that when LCA is used as decision support, the LCI model should reflect the consequences of the decision (JRC-IEA 2010, p. 37).

An Environmental Product Declaration (EPD) can, as expressed by Rydin (2014), "be seen as means for the customer to influence the environmental impact of the purchased products, which gives a requirement on the EPD that it reflects the expected environmental consequences of buying the declared product compared not to buying it", thus making the information from a consequential model the most relevant. For a customer that seeks information with this intention, an environmental label based on attributional modelling may be misleading, as was already pointed out by Weidema (2001). Also, Tillman (2010) recognises that "purchasing inherently involves decisions, and according to the logic described

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above, information intended to support it, such as eco-labels, carbon footprints and environmental product declarations, should be based on consequential LCA".

Yet, all actually implemented EPD schemes specify that an attributional modelling approach should be applied (see, e.g. EPD International 2015), i.e. a modelling where the aggregated impacts are those of the specific or average market suppliers of the inputs to the production, traced backwards in the value chain or supply chain, using one or more specific product properties, often economic value. We have not been able to find any clearly expressed reasons for this choice in the documents of the EPD schemes, but descriptions in other sources, cited below, allow us to speculate several explanations:

- Product declarations in general, especially those of content or origin, usually imply a tracing of the history of the specific product, rather than any consequences of purchasing and consuming it. When transferred to the area of environmental impacts, this could intuitively appear to imply that an EPD should be a declaration of the past environmental impact of the specific product and its expected use and disposal phase, but not specifically intended to indicate the environmental consequences of buying the declared product (Rydin 2014).
- Each environmental label is provided for one specific product only and therefore does not immediately seem to indicate any comparison to any other product, and even though the information is only actionable after the customer has done such a comparison, the sender of the information does not know what comparisons the customer will make (Tillman 2000, 2010). However, as pointed out by Weidema (2003, p. 17): "a consequential LCA may very well assess the consequences of production, use and disposal of a defined quantity more or less of the investigated product. This can be done independently for any product, without prior knowledge on the specific comparison that each assessment may later be used for. Later, when specific comparisons are required, these may be obtained simply by subtracting the individual product systems. These comparisons will be valid as long as the product quantities studied are small."
- The individual purchases of labelled products may appear so small that it can be counter-intuitive to think that the purchase will have any consequences at all beyond a simple change in capacity utilisation of the existing production infrastructure, cf. the idea that "if I stay at home instead of flying, the plane will fly anyway". The fault in this logic is of course that any additional purchase sends a signal to the producers and the sum of all the small signals is what eventually will lead to changes in production capacity (Weidema et al. 1999), e.g. the purchase of an extra aeroplane. The faulty logic is parallel to the idea of a voter that stays at home with the argument that "my vote will not change anything". This logical error also appears in the statement in the ILCD Handbook (JRC-IEA 2010, p. 39) that "small-scale marginal consequences alone are not strong enough to overcome thresholds and trigger large-scale consequences in the market" which leads to the recommendation to use average data rather than marginal data for "micro-level decisions", such as environmental labelling, in spite of the Handbook's general recommendation to use marginal (consequential) modelling when LCA is used for decision support. The statement begs the question why an environmental label is relevant at all if it is not expected to contribute to any large-scale (and long-term) consequences? The ILCD Handbook misrepresents the model for "micro-level decisions" as being "attributional", when in fact it is applying consequential modelling of co-production (popularly known as system expansion), although with average data instead of marginal. This misrepresentation may be the reason for more recent publications conveying the misunderstanding that attributional models already "describe the effect of changes" while consequential models should only be relevant "when changes are made to established processes" (Muthu 2014, p. 115).
- An important feature of EPD information is additivity, i.e. that an EPD of an assembled product can be calculated as the sum of the life cycle impacts of the assembled parts plus the impacts from the assembling activity itself. That only an attributional model should provide such additivity is postulated by Tillman (2000), a postulate that is repeated in Tillman (2010) although already Weidema (2003, p. 16–17) pointed out that this feature is also inherent to consequential models. The reason for the misunderstanding may be the (unrelated) fact that the sum of consequential LCA results for all products in the world does not sum to the total global impact of all human activities, because the impacts of a marginal product are different from the impact of the corresponding average product.
- Consequential EPDs include market-mediated impacts, and ideally also rebound effects (Christiansen et al. 2006), which may sometimes show results that can be seen as surprising or counter-intuitive. Schmidt and Poulsen (2007) suggest that counter-intuitive results may lead to non-acceptance by consumers, although this has not been tested in practice. In contrast to this, one could argue that if results were always intuitive, there would be no need for EPDs.
- Also applicable to environmental labelling, Vessia (2013) and Brandão et al. (2014) suggest that for some policy implementations, as opposed to policy development, precision may be more important than accuracy, and Wardenaar et al. (2012) suggest that consistency of the assessment with the policy objective may be less important than robustness, where robustness is understood as

invariance of the assessment outcomes to uncertainty and changes in real-life conditions, e.g. by choosing a physical allocation key that does not change over time. However, the robustness of a decision is not normally understood as a decision that ignores real-life uncertainty and requires precise and fixed outcomes, but rather as a decision that remains valid under uncertainty and changes in real-life conditions. This can be obtained by combining consequential modelling with robust decision methods (see, e.g. Hall et al. 2012). And if precision or invariance of assessment outcomes is required for a specific policy implementation, this can more simply be obtained by applying temporarily fixed assumptions and background data to the same consequential model that is used to provide assessments consistent with the policy objective.

- While popularity is not an argument for validity or relevance, it may be a strong argument for practical decision-making, as evidenced by "the much stronger history in use" (Henry et al. 2016, p.61), i.e. that it "is more commonly used at present" (ibid, p.6), being used as argument for choosing an attributional approach for the guidelines for LCA of wool textiles—including also other applications than environmental labelling—although "it is anticipated that Guidelines for consequential LCA for wool […] will be developed in future. This will help to overcome the inherent limitations associated with attributional LCA that constrain use of results" (Henry et al. 2016, p. 63).

Both the theoretical literature reviewed above and the practical application in EPD schemes show that the relevance of attributional and consequential information for environmental labelling is still a debatable issue. In this article, we therefore investigate whether a dual label, i.e. representing on the same label the attributional and consequential results for the same product, can be a relevant solution or at least contribute to a more informed discussion.

2 Methods

As shown in Fig. 1, we developed a dual label and applied this to three hypothetical products, A, B and C. The three products are comparable, i.e. they all deliver the same performance and have the same functional unit. The LCA results are hypothetical, to illustrate clearly different results, but they are still realistic, i.e. it would be possible in real life to encounter three comparable products with such LCA results. The top label for each product is the attributional label, while the bottom label is the consequential label.

We now used the representation in Fig. 1 to solicit answers to the question "Which product would you choose?" using a test audience composed of 19 Ph.D. students with a thorough LCA background. Given the very small and not randomly selected audience, the intention of this test was not to obtain unbiased or statistically representative answers, but simply to obtain a first indication of the potential relevance of such a duel label. Before explaining further, we would urge the reader to consider the question in Fig. 1 to obtain a first-hand experience of the thoughts that this question can provoke.

Our test audience was presented with a variant of these labels where the unit was "hours of forced labour" instead of "kg CO_2-equivalents", with the expectation that this would provoke more distinct answers and arguments.

The test audience was then asked in plenum to articulate their deliberations and arguments for their choice. After some plenum discussion on the arguments, two more questions were asked: "Was the attributional information useful?" and "Would you accept to have only the attributional information?"

After the initial test, the same questions, using the labels in Fig. 1, were applied with a larger audience at the SETAC-Europe LCA case studies symposium in Barcelona, November 2017, as well as with another group of 25 Ph.D. students, in both cases using anonymous answering via the Web Clicker online response system.

3 Results

After some time for deliberation on the first question ("Which product would you choose?"), 16 out of 19 students answered that they would choose product C, two would choose product A, and one would choose B. This distribution between response options was largely confirmed by the later application to the two other audiences.

Given the very small and not randomly selected audience, what is interesting here is not so much the distribution of the answers, but rather the arguments given afterwards, especially the arguments for the "deviating" preferences for products A and B. While the preference for product C was supported by the argument that this would lead to the largest improvement, the preference for product B was supported by the argument that both labels showed an improvement, thus giving less uncertainty. The exclusive argument given for the preference for product A was that a low impact reported by the attributional label indicated past good behaviour, which was perceived as a providing a good indication for the expected future impact, while it was perceived as more uncertain whether the improvement indicated by the consequential label could actually be trusted to happen. Interestingly, traceability was not used as an argument in favour of the attributional label.

In the following discussion, it was pointed out that the reason for the relatively low attributional impact of products A and B does not need to be the result of past good behaviour, but can simply be the result of products A and B by chance

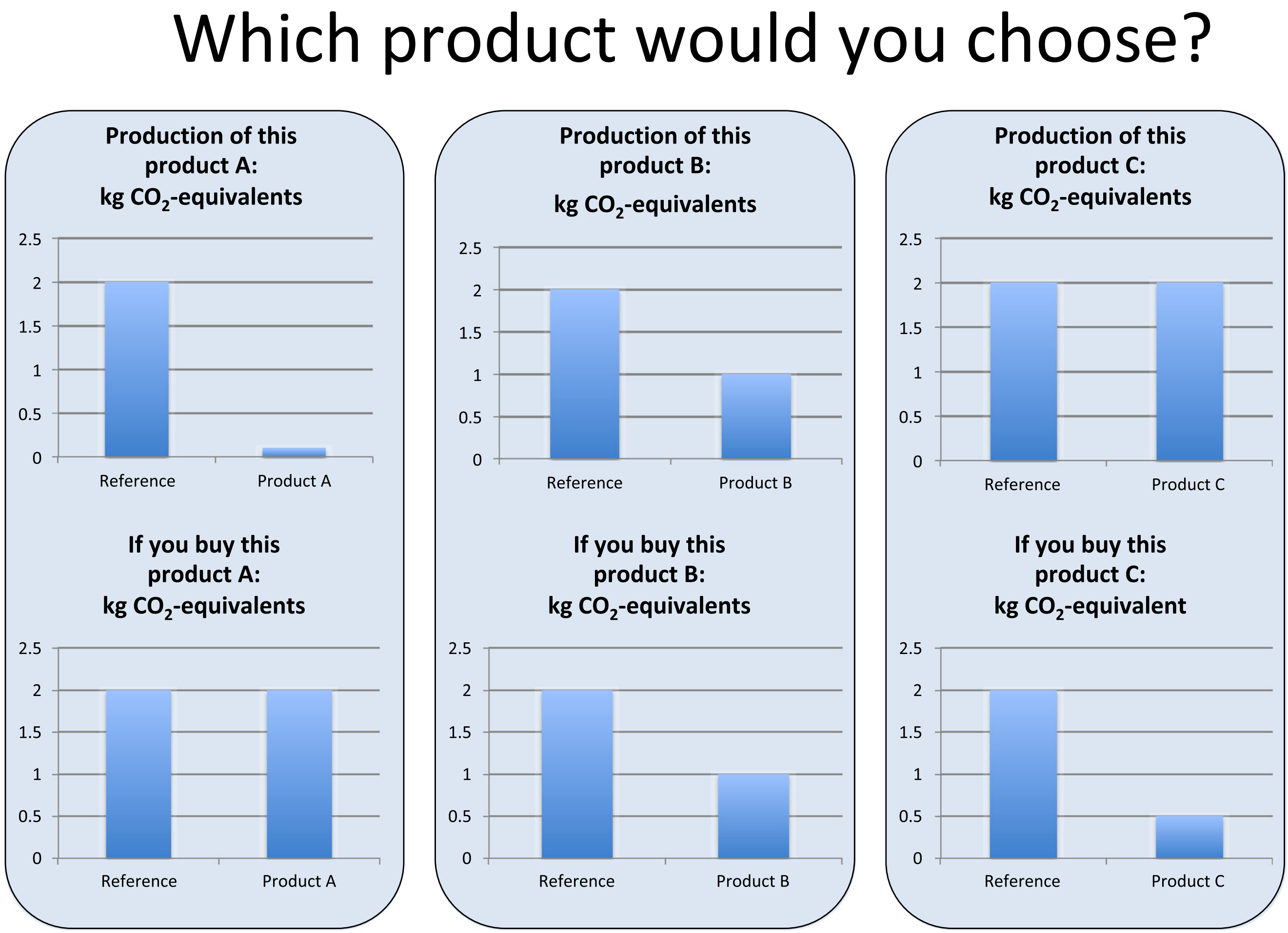

Fig. 1 Three dual labels: For each of the three products A, B and C, which provide the same functional unit, a dual label is provided. The top labels are attributional, while the bottom labels are consequential

being located in a supply chain with a lower occurrence of impacts. One example given was that of average Norwegian electricity being dominated by "clean" hydropower, while the marginal Norwegian electricity could be based on fossil fuels. Norwegian producers—without any active performance improvements—will thus have an attributional label with a low CO_2 emission relative to other countries, while the consequential label reflects that buying Norwegian products will affect the marginal electricity with its higher CO_2 emissions.

To the second question: "Was the attributional information useful?", the general finding was that this information was more confusing than helpful. The dual label should thus rather be seen as a vehicle for investigation into the relevance of attributional and consequential information than as an actual proposal for providing both types of information.

To the final question "Would you accept to have only the attributional information?", the answer was a unanimous "No". This is of course in stark contrast to what is provided by all currently implemented EPD schemes.

4 Conclusions

From this little pilot exercise, it appears that informed consumers *may* have a strong preference for consequential information and that the main problem in communicating consequential results is that they are perceived as less trustworthy and more uncertain due to the fact that the consequences are located in the future. This is in spite of the knowledge by the respondents that the attributional and consequential labels are based on the same data from the recent past (but respectively representing the recent past average and the recent past marginal) with the same degree of data quality and uncertainty. It thus appears important to build into a consequential label some increased level of guarantee of future good behaviour.

We propose to apply the above questions to a more statistically representative audience to confirm or refute the findings of this little test exercise.

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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