Int J Life Cycle Assess (2015) 20:1557–1567
DOI 10.1007/s11367-015-0957-9

LIFE CYCLE MANAGEMENT

Rationality of eco-efficiency methods: Is the BASF analysis dependent on irrelevant alternatives?

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Received: 3 February 2015 / Accepted: 18 August 2015 / Published online: 4 September 2015
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Abstract
Purpose The paper aims at testing and improving the BASF eco-efficiency analysis (EEA) with regard to the rationality axiom "independence from irrelevant alternatives" (IIA). If this axiom is violated, rankings could be biased or influenced subjectively by the choice of considered alternatives.
Methods We introduce an artificial yet realistic "irrelevant alternative" in an EEA case study and compare the ranking results. The different stages of the EEA are analysed to uncover the potential source(s) of the detected violation.
Results and discussion The example proves the violation of the IIA rationality axiom in the EEA. It is shown at which stages and how the weights in the aggregation process have to be adjusted to avoid this shortcoming.
Conclusions In specific constellations, the EEA may violate the IIA rationality axiom. By adequately adjusting the weights at certain stages of the aggregation process, this shortcoming can be avoided. Also, the results account for the relevance of basic principles derived from decision theory in the life cycle assessment/sustainability context.

Keywords Decision-making tool · Decision theory · Eco-efficiency analysis · Environmental management · Environmental performance measurement · Metrics and measurement · SEEBALANCE® · Sustainability management · Sustainability performance measurement

Responsible editor: Ivan Muñoz

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1 Introduction

The selection of one "best choice" product or process[1] and the assembly of portfolios are fundamental decisions in the context of corporate environmental management and environmental performance measurement (Ilinitch and Schaltegger 1995; Schaltegger and Sturm 1998). Respective methods integrate both the ecological and the economical dimension, often founded on the concept of "eco-efficiency" (DeSimone and Popoff 1997; Schaltegger and Sturm 1990; WBCSD 2000). Among other potential uses, eco-efficiency is used at the business strategic level to support portfolio decisions and is operationalised for this purpose as a ratio between environmental impact and economic cost or value (Schaltegger and Sturm 1998; Ehrenfeld 2005; Huppes and Ishikawa 2005b). Although the determination of cost or value raises some new problems in this context (Kicherer et al. 2007; Ehrenfeld 2005; Huppes and Ishikawa 2005b), it is usually regarded as a standard. However, the determination of the environmental impact faces the same problems which are prevailing for life cycle assessment (LCA) and any other environmental assessment methods, e.g., functional unit definition, boundary selection, choice of impacts, allocation, weighting and valuation, data availability and quality (Finnveden 2000; Reap et al. 2008a, b; Wenzel 1998). As a consequence, the methods must compromise between scientific correctness and practical applicability.

The *eco-efficiency analysis* (EEA) of the chemical firm BASF is a well-known method for the relative evaluation of the economic and ecological performance of products or processes. The method was introduced in 1996 by BASF together with management consultants Roland Berger and, since then,

[1] In the following, instead of "products and processes" we will just use "products."

has been conducted in more than 600 studies (BASF 2015a). With regard to the methodology, the eco-efficiency analysis founds on the concept of eco-efficiency (Kicherer et al. 2007). Six ecological impact categories based on LCA data are normalised and aggregated using an additive weighting scheme in three stages. The resulting total ecological impact is plotted against the total costs. An overview of the method is provided e.g., by Saling et al. (2002).

Typical applications of the EEA are e.g., the comparison of alternative processes for the production of indigo and the dyeing of blue denim or the comparison of alternative packaging systems for bottled mineral water (Shonnard et al. 2003). The EEA has also been applied to compare corn grown with or without the BASF fungicide Headline®. The analysis demonstrates that the Headline system provides significantly lower estimated production costs and environmental impacts. The major contributing factor to this result is the yield increase associated with the use of the foliar fungicide (BASF 2010, 2015b).

Regarding the determination of the alternatives in an EEA, it is recommended to include "[…] as many alternatives in the marketplace or in development as possible that can perform the same function." (Uhlman and Saling 2010, p. 18). The compilation of the set of alternatives is crucial for the analysis and its results—both from the general decision-theoretic perspective and the particular LCA scoping perspective (Dyckhoff and Ahn 1998; Wenzel 1998). Usually, the set should include "like with like" alternatives, but sometimes "apples and oranges" are compared. Such outliers may appear when similar alternatives are compared to the alternative of maintaining a status quo, e.g., when comparing the use of pesticides to conventional cultivation. If the status quo alternative is irrelevant in the sense that it is worse in all the criteria considered than any other alternative, consequently, it must be ensured that this outlier is not relevant in the sense that it influences the preference between the other alternatives. Such an influence might appear inadvertently. However, as a general problem in the LCA context, the model parameters provide scope for manipulation (Wenzel 1998, p. 286)—at least if the process is intransparent: by adding or leaving out an outlier alternative, the analysis may produce a result within the ranking of the other alternatives which is preferred for some "political" reasons of the decision maker or some interested party.

The question raised here is examined in decision theory under the heading "independence from irrelevant alternatives". Prescriptive decision analysis aims at supporting people in making decisions, in particular when these are complex and difficult. It is a general conception that decisions should be "rational". The evaluation of a decision as rational or irrational though depends on the individual value system of the decision maker and his subjectively influenced perception of information. According to this bounded notion of rationality, the support offered by prescriptive decision analysis has been specified as follows: "*Rationality* is not a clear-cut concept. It can, however, be put into more concrete terms by defining some procedural requirements […] and agreeing on some basic claims regarding consistency" (Eisenführ et al. 2010, p. 1–5). The *independence from irrelevant alternatives* (IIA) is one of these axioms. It requires that the choice between two alternatives should not be influenced by the existence or non-existence of a third alternative (Eisenführ et al. 2010, p. 7). The relevance of the axiom in the context of eco-efficiency methods has been pointed out (Huppes and Ishikawa 2005a).

For the EEA method, a systematic and objective test based on rationality axioms of decision theory has been requested (Schmidt 2007, p. 71ff., 210), but has not been systematically undertaken so far. It is the main purpose of the present paper to close this gap for the rationality axiom IIA and, in doing so, to stress the general relevance of decision-theoretic foundations for methods in the context of ecological and sustainable performance measurement analyses (Finnveden et al. 2002, p. 178; Hertwich and Hammitt 2001a, b; Seppälä et al. 2001).

The paper is organised as follows: Section 2 provides a brief overview of the EEA and of the respective literature for more detailed descriptions. Section 3 familiarises the reader with the IIA rationality axiom and its relevance. Section 4 introduces an EEA case study on alternatives for the disposal of contaminated soil. By including an "irrelevant alternative" and comparing the ranking results for the original and the modified set of alternatives; the violation of the IIA rationality axiom in the EEA is demonstrated. Section 5 analyses the so-called range effect as the source of the violation and shows how this shortcoming of the EEA can be avoided by adequately adjusting the weights when including or eliminating alternatives. Section 6 summarises the findings and embeds the relevance of a decision-theoretic foundation of the EEA in a discussion of more general limitations of eco-efficiency and life cycle analyses with regard to their political background in the global sustainability efforts.

2 Overview of the BASF eco-efficiency analysis

Established in 1996, the eco-efficiency analysis of the chemical firm BASF is a well-known method in the context of corporate environmental performance measurement. The aim of EEA is described by BASF as the comparison of similar products or processes with regard to the environmental impact in proportion to cost-effectiveness—usually under consideration of the entire supply chain and life cycle stages (BASF 2015a; Uhlman and Saling 2010, p. 17). The comparison may support the strategic, R&D, marketing or public relations departments in e.g., a best choice decision, the construction of a portfolio or the analysis of strengths and weaknesses (Saling et al. 2002, p. 217f.; Schmidt 2007, p. 60).

Eponymous for and underlying the EEA is the concept of eco-efficiency (DeSimone and Popoff 1997; Schaltegger and Sturm 1990; WBCSD 2000) and its operationalisation as a ratio between environmental impact and economic cost or value (Schaltegger and Sturm 1998; Ehrenfeld 2005; Huppes and Ishikawa 2005b). In terms of methodology, EEA can be characterised as a systematic aggregation of multiple input data based on weighting information. It is an iterative method and its steps can be repeated and/or modified according to the state of knowledge (Schmidt 2007, p. 62). Regarding the ecological component, the EEA directly ties into earlier work on life cycle assessment (e.g., SETAC 1993; ISO 14040 2006; ISO 14044 2006). Recent developments in the context of life cycle assessment are constantly integrated in the analysis (e.g., Landsiedel and Saling 2002).

The starting point for the EEA is to *define the goal and scope of the study* by determining the customer benefit (functional unit), the alternatives considered and the system boundaries. Usually, the entire life cycle "from cradle to grave" is considered, but sometimes the system boundary is restricted to those phases which differ for the products compared (Saling et al. 2002, p. 204; Uhlman and Saling 2010, p. 18). For example, in the context of low-energy modernisation/avoidance of GHG emissions of a house, the customer benefit "living in an existing, detached house in Germany at an average room temperature of 19 °C for 40 years (2011–2051)", is analysed for the three alternatives: (1) no insulation, (2) facade refurbishment with an external thermal insulation composite system based on Styropor® foam boards and (3) the same system, but based on Neopor® foam boards. For the first alternative, the relevant life cycle phase is the use (heating) of the house, whereas for the two latter alternatives also the phases production, installation, and recycling/disposal (incineration and landfill) of the insulation system are relevant. The phases construction and disposal of the house are omitted, since these processes are identical for the alternatives (Paczkowski and Russ 2013).

In the next step, *the ecological and economic data are collected*. The data regarding the environmental impacts are acquired and calculated (predominantly) according to the life cycle assessment standards ISO 14040/14044 and presented in six categories: resource consumption; energy impacts; emissions (to air, water and soil); land use; toxicity potential and risk potential. In the economic category, the total cost calculation considers the real life cycle costs that occur as well as future subsequent costs. Apart from a standard procedure, different costing models have been applied in EEA case studies (Uhlman and Saling 2010, p. 18f.).

Following the data collection, *the data are normalised in each category* by attributing to the least favourable alternative the value of 1 and by setting the other alternatives in relation to that i.e., the value 0 reflects the most favourable value regarding a certain category.

The subsequent *aggregation of the environmental categories* can be considered as the essential step of the EEA. While the costs can be summed up without additional weighting, the multiple ecological impact categories do require a complex compressing procedure. The normalised values are aggregated using an additive weighting scheme organised in three stages, as depicted in Fig. 1.

The weighting scheme is based on *overall weighting factors* of the different ecological impact categories. They are derived stepwise in a bottom-up approach as the geometric mean of so-called relevance and societal factors (both normalised). *Societal factors* are constant (though periodically updated) for all EEA studies. They are determined through e.g., surveys, public opinion polling and expert interviews underlining their subjective character: They "account for society's opinion on the importance of each environmental impact" (Uhlman and Saling 2010, p. 22). In contrast, the *relevance factors* are considered as scientific and objective factors, derived uniquely for each EEA and indicating the significance of each ecological impact category for this particular EEA. They are calculated by dividing the alternative's impacts in each of the six environmental categories by the total impact in the EEA's geographic reference country or region, derived from e.g., statistical databases (Uhlman and Saling 2010, p. 22).[2]

To calculate each alternative's final *total ecological impact*, the categories' normalised impact values are multiplied by the respective overall weighting factor and summed up. Requiring a last weighting procedure (Kicherer et al. 2007), the total ecological impact is plotted against the total costs based on the average of all alternatives. Graphically, this results in the so-called eco-efficiency portfolio in which "[t]he distance of the individual alternatives to the portfolio diagonal is a measure of the respective eco-efficiency" (Saling et al. 2002, p. 216). Also, varying *representations of the disaggregated results* of the one economic and the six ecological categories in tables and graphs are of interest for the addressees. In particular, EEA provides an integrated illustration plotting the alternatives' performance in the various ecological impact categories as a spider web. This so-called environmental fingerprint "makes it easy to visualize the trade-offs between alternatives by clearly showing where certain alternatives perform well and where their performance is less desirable" (Uhlman and Saling 2010, p. 22).

For both the environmental fingerprint and the eco-efficiency portfolio, one should note that the EEA is a comparative tool: The results do not represent absolute values but

[2] Except for the categories *Emissions* and *Emissions to air*. Due to their aggregated form, they are calculated based on the weighting factors (relevance and societal) of their subcomponents.

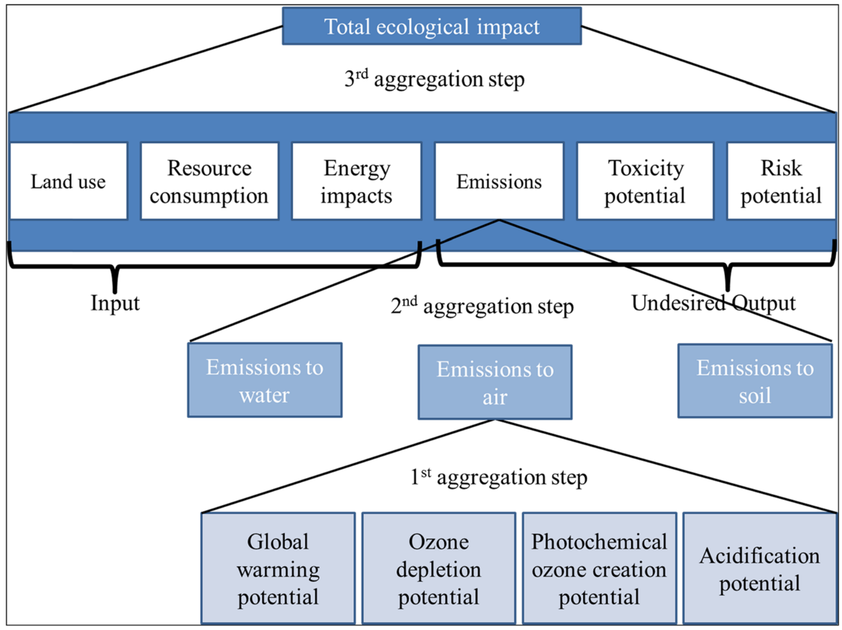

Fig. 1 EEA weighting scheme for the total ecological impact

are relative to the alternatives considered (Saling et al. 2002, p. 213). Especially, the balls' positions in the eco-efficiency portfolio change as soon as the position of one ball changes (Saling et al. 2002, p. 216).

Verification of the results is mandatory for each EEA. The stability is tested by variation of the model's assumptions, boundaries and societal factors (Saling et al. 2002, p. 217). Regarding the variation of weights, Saling et al. (2002, p. 217) find in many EEAs that even substantial changes have only a very small impact on the conclusions, although they admit: "True, the positions in the portfolio change, in some instances even to the extent of the order on the ecological axis changing, but the conclusions with regard to eco-efficiency (environment and costs) change very rarely." The question at hand in this paper has the reverse perspective: (How) do the weights have to be adjusted in order not to change the relative position of the products if the portfolio is modified?

The EEA has been applied by BASF in more than 600 studies—for in-house purposes as well as for customers and customers' customers. The applications show a broad variety, e.g., the comparison of the relative eco-efficiency of pavement-preservation technologies for urban roads or of chelating agents for household automatic dishwashing applications. Moreover, the method has been constantly refined: Meanwhile, BASF has introduced the SEEBALANCE® tool, which not only allows for the assessment of environmental impact and costs but also for the societal impacts of products and processes. However, the aggregation mechanism is based on EEA, and therefore the following observations on the EEA generally also hold for the SEEBALANCE® tool.

3 Independence from irrelevant alternatives

It is a general conception that decisions should be rational. However, rationality is not an objective and provable property. In reality, decision processes are not characterised by a perfect "homo oeconomicus" but by "bounded" rationality (Vriend 1996). Capacity limits occur when decision makers collect and process information. But even if a decision process was "perfect", rationality would stay an "instrumental" concept as it is based on a choice among given alternatives. These alternatives are evaluated in terms of a means-ends relationship, without scrutinising the "ends" in some larger ethical, social, human or political sense. Rationality is also often linked to the knowledge available. Hence, the bounded and instrumental rationality is also relative to "the dominant view of experts" (Weber and Schäffer 2008, p. 50–59).

With regard to this conception of bounded rationality, the prescriptive decision theory aims at supporting the decision process both by developing methods and by laying down some basic requirements that seem plausible to most decision makers. The *procedural requirements of rationality* specify some of the basic guidelines that a decision procedure should comply with. For example, the decision maker should ensure that he is tackling the right problem (by extending or splitting the decision problem at hand) and that he invests an appropriate effort in solving it (Eisenführ et al. 2010, p. 5f.). *Consistency* of a decision means that it is based on assumptions which are not mutually contradictory and which represent certain standards or *axioms of rationality* that the decision maker(s) accepts, e.g., "prospective orientation", "invariance (to certain scale variations)", "transitivity" and independence from irrelevant alternatives. It should be noted that decision theorists proposed distinct catalogues of principles meaningful for rational decision making (Eisenführ et al. 2010, p. 6–9; Milnor 1954, p. 57).

In this paper, we focus on the rationality axiom independence from irrelevant alternatives (IIA) illustrated by Eisenführ et al. (2010, p. 9) as follows: "The menu in a restaurant offers two dishes: salmon and schnitzel. When ready to order the salmon, you are informed by the waiter that they are also offering pork chops with sauerkraut today. After some reflection on the three options, you decide to order the schnitzel. Such a decision pattern is irrational—the 'irrelevant' alternative of pork chops should not influence the preference you have with respect to salmon and schnitzel."

The relevance of the IIA rationality axiom has been shown, e.g., in the context of the bargaining problem (Nash 1950) or of voting procedures (Arrow 1963) and in particular, in the context of eco-efficiency analyses (Huppes/Ishikawa (2005a, p. 37). IIA is a requirement which should hold generally for all analytic methods based on the relative comparison of alternatives. When introducing or eliminating alternatives, the absolute results of the alternatives may differ, but the relative order of the other alternatives should be stable. This is an important requirement because otherwise, a manipulation of the results for subjective and political reasons is possible through an according choice of the set of considered alternatives. Therefore, the relevance of the IIA rationality axiom for the BASF eco-efficiency analysis—whose purpose has been described above as the relative evaluation of the economic and ecological performance of products or processes—should be obvious.

4 Testing the EEA for violation of the IIA rationality axiom

All publications on the BASF eco-efficiency analysis (EEA) we know of are written by authors who are more or less closely related to BASF. Therefore, our paper aims at a neutral, scientific evaluation of this method that has proved successful in practice. In particular, it has not been systematically tested regarding its decision-theoretic foundation. However, Schmidt (2007, p. 71ff., 210) initiated a basic foundation for the EEA in a decision-theoretic perspective and points out the necessity of further investigations. This article focuses on this gap by testing the EEA regarding its decision-theoretic foundations, in particular, the fulfilment of consistency axioms for rational decision-making. We can thereby stress the general relevance of those axioms for sustainability performance measurement based on multi-attributive decision methods. However, as testing all axioms proposed by prescriptive decision theory would go beyond the scope of this article, we select one exemplary rationality axiom, the IIA. Its selection is based on the fact that the requirement of IIA should hold generally for all analytic methods based on the relative comparison of alternatives. Furthermore, the axiom's importance for the EEA is stressed by the fact that a potential violation enables to manipulate the results for subjective and political reasons by an according choice of alternatives. We focus on the ecological aggregation process, due to the fact that it is less the economic evaluation and more the ecological facet and its link to the economic evaluation that represents the innovative part of the EEA.

The test of the EEA method regarding the violation of the IIA rationality axiom is based on a BASF study on alternatives for the disposal of soil contaminated by petroleum-derived hydrocarbon. The study aimed at the decision, whether ground preparation or dumping of the contaminated soil was preferable under consideration of ecological and economic criteria (Kleine et al. 2004). The study has been selected because it provides sufficient data for a reconstruction of the analysis. To facilitate the understanding of the analysis performed here, Table 1 shows the data for just three of the six alternatives in the aforementioned study, all of them representing ground preparation methods: alternatives A and B correspond to the microbiological processes of type 1 (MB1) and type 3 (MB3), and alternative C corresponds to the washing process (W). To be able to illustrate the IIA violation, compared with the original data for alternative A, the energy value was increased by 3000 MJ and the cost value decreased by 205 €. Both modifications result in values below or between those of the alternatives B and C i.e., the data for alternative A is still realistic.

We now apply EEA first on two alternatives A and B in sample 1 and afterwards on three alternatives A, B and C in sample 2. In any case, the IIA axiom demands that the inclusion of alternative C in sample 2 should not influence the preference order of the alternatives A and B in comparison to sample 1 when applying the EEA (or any other decision support tool). Alternative C is not dominated by either of the other two alternatives. Nevertheless, it seems to be the worst of all three alternatives and therefore appears to be totally irrelevant. When compared with alternative B, it shows a slightly better value just for energy (28,576 MJ versus 30, 029 MJ) and apart from that, worse values e.g., significantly for emissions to water, despite higher cost; compared with alternative A, it has slightly better values only for photochemical ozone creation potential (POCP) and resource consumption.

Table 2 shows the violation of the IIA rationality axiom occurring when applying EEA on samples 1 and 2. The last row displays the total ecological impact (a smaller value is preferred to a higher value). While alternative B is ecologically preferable over A in sample 1 (0.953>0.950), the preference order changes in favour of alternative A (0.689<0.693) when including alternative C (0.993). In a reverse perspective, the preference order A>B changes into A<B when alternative C is eliminated from sample 2.

Table 1 Basic data (impact values and costs) of three alternatives for the disposal of contaminated soil

Alternatives Criteria	A	B	C
GWP [CO_2 equivalents]	2,551,615	2,958,240	3,804,116
ODP [CFC equivalents]	0.0005	0.0001	0.0126
POCP [ethene equivalents]	1849	1505	1537
AP [SO_2 equivalents]	17,037	18,583	19,986
Emissions to water [crit. vol. cbm]	12,235	8039	227,417
Emissions to soil [weighted mass]	4223	4241	5280
Energy impacts [MJ]	25,752	30,029	28,576
Resource consumption [weighted kg]	11,247	9524	11,002
Land use [weighted sqm]	12	12	12
Toxicity potential - production [evaluation points]	550	350	1250
Toxicity potential - utilisation [evaluation points]	3500	2800	17,700
Toxicity potential - deposit/disposal [evaluation points]	50	50	50
Risk potential - pollution by traffic [evaluation points]	1.0	1.0	1.0
Risk potential - uncertainty of decline [evaluation points]	0.5	1.0	2.0
Risk potential - burden of change to the status quo [evaluation points]	0.0	0.0	2.5
Costs [€]	3119	3116	3331

Besides the total ecological impact, Table 2 displays the weighted normalised values of each of the six ecological impact categories. The analysis of the according preferences for the alternatives A and B demonstrates that in all ecological impact categories, the preferences between A and B are identical for samples 1 and 2. However, the preference order changes in the final aggregation stage. Thus, the violation of the IIA rationality axiom takes place immediately when calculating the total ecological impact. In the following section, we will analyse the source of the violation and the necessary adjustments to avoid that shortcoming.

5 Range effect and correct adjustment of weights

In prescriptive decision theory, it is the *range effect* being accounted for the cause of the IIA violation in multi-attributive evaluations (e.g., von Nitzsch and Weber 1993; Eisenführ et al. 2010, p. 154, 385f.). To be able to clarify its functionality and relevance regarding the IIA rationality axiom, we explain the basic structure of the additive value model of decision theory (e.g., Keeney and Raiffa 1993, p. 95ff., 117ff.; Eisenführ et al. 2010, p. 126–129) because it is the underlying aggregation form of all three stages of EEA. The additive model determines the value of an alternative a by summing up attribute-specific normalised and weighted values:

$$v(a) = \sum_{r=1}^{n} w_r v_r(a_r) \tag{1}$$

The symbol a_r indicates the level of attribute X_r for alternative a (also consequence or level of target achievement). The term $v_r(a_r)$ represents the respective value of the

Table 2 Weighted, normalised impact values of each category and total ecological impact

Alternatives	Sample 1 (S1)			Sample 2 (S2)				S2/S1	
Criteria	A	B	Most polluting alternative*	A	B	C	Most polluting alternative*	A (%)	B (%)
Weighted, normalised impact value for emissions	0.480	0.482	B	0.364	0.368	0.498	B	75.9	76.3
…for energy impacts	0.097	0.113	B	0.090	0.105	0.100	B	92.6	92.6
…for resource consumption	0.100	0.085	A	0.093	0.078	0.091	A	92.6	92.6
…for land use	0.005	0.005	identical characteristic	0.004	0.004	0.004	identical characteristic	92.6	92.6
…for toxicity potential	0.200	0.165	A	0.097	0.087	0.200	A	48.7	52.6
…for risk potential	0.071	0.100	B	0.040	0.050	0.100	B	56.0	50.0
Total ecological impact	0.953	0.950	A	0.689	0.693	0.993	B	72.2	72.9

*Only alternatives A and B considered

individual value function v_r i.e., it provides an evaluation of the alternative's consequences in the attribute X_r and reflects the preference of the decision maker (Eisenführ et al. 2010, p. 107–113). Each individual value function is normalised onto the interval [0, 1] by:

$$v_r\left(x_r^+\right) = 1 \text{ and } v_r\left(x_r^-\right) = 0 \tag{2}$$

$$\text{such that } \Delta v_r(B_r) = v_r\left(x_r^+\right) - v_r\left(x_r^-\right) = 1 \tag{3}$$

B_r is the interval or *range* of possible values for attribute X_r, i.e., all relevant values lie between x_r^+ (best level) and x_r^- (worst level). The weights $w_r > 0$ assigned to the attributes by the decision maker should hold for

$$\sum_{r=1}^{n} w_r = 1 \tag{4}$$

To neutralise the range effect, changes in the range of the consequences (e.g., when alternatives are modified, included or eliminated) should result in clearly prescribed adjustments of the attributes' weights (Eisenführ et al. 2010, p. 154, 385f.). Otherwise, the preference order of the original situation might not be reproducible, and a violation of the IIA axiom is possible. Eisenführ et al. (2010, p. 154) point out that: "Any method for the determination of weight factors that does not refer to particular attribute intervals is doomed to fail." von Nitzsch and Weber (1993) showed in experimental studies little sensitivity of decision makers to the size of the intervals. When faced with different intervals, the decision makers do not adjust their importance statements at all or on average, only insufficiently.

The general logic behind the correct adjustment of weights is that the weight attached needs to be decreased (increased) for a narrower (broader) range of the respective criterion. "This is due to the fact that for the small interval, the value difference between the worst and best attribute levels is larger than the value difference that we obtain for the same attribute levels in the extended interval (with a re-normalised value function)" (Eisenführ et al. 2010, p. 151). More precisely, the necessary adjustment of weights can be derived from the transformation factor of the normalised attribute values. Multiplying the weight by the reciprocal of this factor, i.e., by $M = \Delta v_r(B_r')/\Delta v_r(B_r) = \Delta v_r(B_r')$, leaves the alternatives' ranking consistent with the original situation. After that, the resulting weights have to be normalised to the value of 1 again (Eisenführ et al. 2010, p. 151–154; von Nitzsch and Weber 1993, p. 938f.).

We now transfer these reflections on the range effect and the correct adjustment of weights to the EEA methodology. Looking at the data in Table 1, the inclusion of alternative C results in a broader range for nine ecological criteria: e.g., for global warming potential (GWP), alternative B has the worst value (2,958,240 units) and in sample 2 it is alternative C (3, 804,116 units).

The last two columns in Table 2 represent the relative shift in the impact categories after inserting alternative C. On the one hand, the impact categories without variation of the range (energy impacts, resource consumption and land use) are identically reduced for the alternatives by 92.6 % of the original value. On the other hand, the weights of the impact categories with variation of the range (all others) do not neutralise the range effect thus causing a modified relative valuation of the alternatives in the sub-categories and the resulting total ecological impact.[3] A change of the preference order for the sub-criteria is possible but does not occur in this example. However, the relative changes have an impact when aggregated: as noted before, the preference order changes at the third and last aggregation stage. What should be noted at this point is that, obviously, the range effect is the cause of the reversal of the preference order.

The last column of Table 3 displays the weights which the EEA should use for a correct compensation of the range effect and the weights used in the current procedure. From the inconsistencies, it becomes clear that there is no aggregation stage with a correct compensation. To avoid the influence of the range effect and a potential violation of the IIA axiom, the weights have to be adjusted as explained earlier. For example, for GWP, the EEA uses the weight 44.4 %. After compensation of the range effect according to

$$44.4\,\%*\frac{1-0.863}{0.778-0.671} = 44.4\,\%*\frac{0.137}{0.107} = 44.4\,\%*1.286$$
$$= 57.14\,\%$$

and a normalisation step, the weight is calculated to 48.3 %.

In detail, we identified the following revision requirements regarding the current EEA weighting procedure:

- The explanations on the range effect showed that weights should always be assessed subject to the ranges of the alternatives. However, Table 3 displays that in the EEA, the weights for the attributes toxicity potential and risk potential are assessed globally instead of being assessed subject to the ranges. The last eight rows in Table 3 show that this statement holds for both the weights of the preliminary aggregation of life cycle phases and the weights used for the aggregation at the third stage. The weights stay fixed in both scenarios.
- In addition, to avoid the range effect, for the remaining attributes (GWP, ozone depletion potential (ODP), POCP,

[3] Please note that the ranges of the sub-criteria *POCP*, *Toxicity potential - deposit/disposal* and *Risk potential - pollution by traffic* also do not change, but due to varying ranges of other sub-criteria, the final aggregated criteria ranges for *Emissions*, *Toxicity potential* and *Risk potential* do change.

Table 3 EEA weights versus weights with correct adjustment to range variation

Aggregation step	Sample Criteria	Normalised weight in S1 - acc. to EEA (%)	Normalised weight in S2 - acc. to EEA (%)	Normalised weight in S2 - adjusted (%)
1	GWP	44.4	46.8	48.3
	ODP	0.1	0.7	3.0
	POCP	26.9	25.0	22.8
	AP	28.5	27.5	25.9
2	Emissions to air	31.1	27.6	21.6
	Emissions to water	2.9	10.4	30.9
	Emissions to soil	66.0	62.0	47.5
3	Emissions	48.2	49.8	43.9
	Energy impacts	11.3	10.5	5.9
	Resource consumption	10.0	9.3	5.2
	Land use	0.5	0.4	0.2
	Toxicity potential	20.0	20.0	34.3
	Risk potential	10.0	10.0	10.4
Weights for aggregation of the life cycle phases: toxicity potential	Production	20.0	20.0	13.8
	Utilisation	50.0	50.0	77.0
	Deposit/disposal	30.0	30.0	9.1
Weights for aggregation of the life cycle phases: risk potential	Pollution by traffic	30.0	30.0	21.4
	Uncertainty of decline	40.0	40.0	57.1
	Burden of change to the status quo	30.0	30.0	21.4

acidification potential (AP), emissions to air, emissions to water, emissions to soil, emissions, energy impacts, resource consumption and land use) the assessment of weights has to be assessed subject to the ranges, too. Here, the pivotal problem is the combination of the relevance factors with the societal factors (in form of the geometric mean, both normalised) as the latter are constant for all EEA studies (though periodically updated) without considering individual ranges and therefore enabling the range effect. In contrast, the relevance factors in an isolated perspective do consider the range effects correctly as they are calculated by dividing the worst alternative's impact in each of the environmental categories by the total impact in the EEA's geographic reference country or region. This applies only to the non-aggregated criteria and therefore the relevance factors for air emissions and emissions are an exemption. These could not be used exclusively but needed to be revised furthermore as they are calculated differently, based on the weighting factors of their subcomponents and thereby including—beside the relevance factors of their subcomponents—also the global societal weighting factors of their subcomponents.

- Finally, the stepwise aggregation procedure of the ecological impact categories prescribes the re-normalisation of the results of the pre-aggregated attributes (emissions to air, emissions, toxicity potential, risk potential) according to the highest value of the alternatives. As their respective weights are calculated based only on the weighting factors (relevance and societal) of their subcomponents, the respective weights do not consider the according range effect and therefore lead to distortions in the evaluation.

Table 4 shows that after an according adjustment of all weights in the EEA, i.e., applying the weights from the last column in Table 3 for sample 2, there are consistent results regarding the preference order in samples 1 and 2. Here, the reconstruction of the preference order holds for both the total impact aggregation and its preparation stages. The influence of the range effect is neutralised by the consistent adjustment of the weights, and the potential violation of the IIA is prevented.

6 Conclusions

Eco-efficiency analysis (EEA) by BASF is a fruitful method as it addresses the rising relevance of ecological performance measurement by means of a precise aggregation procedure. At the same time, EEA stresses the fact that (for competitive reasons alone) ecological analysis cannot stand isolated but must be combined with the analysis of economic factors. Nevertheless, the method has some shortcomings. As an example, we illustrated the violation of the well-founded rationality axiom independence from irrelevant alternatives (IIA). The range effect was accounted for the cause of the violation. By adequately adjusting the weights when including or

Table 4 EEA results for consistent weights

Alternatives	Sample 1 (S1)			Sample 2 (S2) without range effect				S2/S1	
Criteria	A	B	Most polluting alternative*	A	B	C	Most polluting alternative*	A (%)	B (%)
Weighted, normalised impact value for emissions	0.480	0.482	B	0.251	0.252	0.439	B	52.2	52.2
… for energy impacts	0.097	0.113	B	0.051	0.059	0.056	B	52.2	52.2
… for resource consumption	0.100	0.085	A	0.052	0.044	0.051	A	52.2	52.2
… for land use	0.005	0.005	identical characteristic	0.002	0.002	0.002	identical characteristic	52.2	52.2
… for toxicity potential	0.200	0.165	A	0.104	0.086	0.343	A	52.2	52.2
… for risk potential	0.071	0.100	B	0.037	0.052	0.104	B	52.2	52.2
Total ecological impact	0.953	0.950	A	0.498	0.496	0.996	A	52.2	52.2

*Only alternatives A and B are considered

eliminating alternatives, changes in the preference order induced by irrelevant alternatives can be avoided.

As we know no better methods, it cannot be the motivation of this article to scrutinise EEA in general. Quite the opposite: we aim to improve EEA by obeying rationality requirements well established in the field of decision theory and analysis. Ultimately, rationality represents an underlying requirement of any controlling conception and tools of decision support in practice. “The fact that intuitive behaviour so often violates the most basic rationality principles increases the relevance of decision analysis for all people eager to obey the rules of rationality” (Eisenführ et al. 2010, p. 9). In particular, an augmented decision-analytic theoretical foundation will be fruitful not only for the EEA method of BASF but for all methods of ecological performance measurement and life cycle analyses (Finnveden et al. 2002, p. 178; Hertwich and Hammitt 2001a, b; Seppälä et al. 2001). Respective methods must compromise between comprehensible and non-ambiguous results on the one hand and the rising influence of subjective value judgements that comes with a rising aggregation level on the other hand. Some of the conceptual limitations which have been criticised for eco-efficiency and life cycle analyses, and which might be supported by findings of decision analysis are discussed in the following.

Some authors have pointed at the importance of transparency regarding the choices made within modelling, e.g., functional unit definition, boundary selection, choice of impacts, allocation, weighting and valuation, data availability and quality (Finnveden 2000; Reap et al. 2008a, b; Wenzel 1998). Those choices are based on subjective value judgements and therefore are exclusively valid for the decision maker and the situation at hand. However, in many applications, mandatorily neither a decision maker nor a value system is determined. “A framework developed from a societal perspective rather than a company perspective might thus look different” (Dreyer and Hauschild 2006). It is therefore recommended to thoroughly investigate the interested parties or sponsors (Wenzel 1998, p. 286f.) and for the decision maker to disclose the choices as transparently as possible (Finnveden et al. 2002, p. 180f.).

As one element of scoping, the compilation of the set of alternatives is crucial for the analysis and its results. In this article, we pointed at the case of irrelevant alternatives which may violate the IIA axiom due to the range effect. However, the compilation of the set of alternatives is crucial also in a broader perspective. Efficiency analyses focus on a relative evaluation of given alternatives and cannot guarantee a sound decision if the set of alternatives is incomplete or insignificant (Dyckhoff and Ahn 1998). “Better choices among a set none of which are good enough are fool’s choices” (Ehrenfeld 2005).

Closely related to this aspect is the distinction between the concepts of efficiency and effectivity (Ahn and Dyckhoff 2004). Eco-efficiency methods do not provide assessments of absolute contributions to ecological and economic goals (Kicherer et al. 2007, p. 542). Instead, they analyse a means-ends relationship and do not scrutinise if the ends are satisfactorily met. This problem is tightened as rebound-effects (take-back effects) may occur, i.e., beneficial effects of eco-efficiency which are reduced or even overcompensated by behavioural responses (von Weizsäcker et al. 2009). Accordingly, Huppes and Ishikawa (2005a, p. 29) point out: “The usefulness of such win-win analysis hence is limited, because it cannot give guidance on the question whether the win-win realised is good enough for society to improve its overall environmental performance”. In the EEA, the use of surveys, public opinion polling and expert interviews for the determination of societal factors can be regarded as a respective means to include societies’ preferences on different ecological impact categories (Landsiedel and Saling 2002).

Furthermore, it has been scrutinised whether the objectives considered by eco-efficiency analyses are in line with “normative overtones” (Huppes and Ishikawa 2005b, p. 44), i.e., the political perspective of global sustainability efforts. Ehrenfeld (2005, p. 7f.) concludes: “This simple tool and similar variants are very useful for strategic decisions within a firm but only loosely connected, if at all, to global improvements consistent with the limit driving other related concepts such as Factor X". To become a useful indicator, he suggests to couple eco-efficiency with other indicators and tools. BASF provides an according development: the SEEBALANCE® tool does not only allow for the assessment of environmental impact and costs but also for the societal impacts of products and processes. Methodically, however, the introduction of new criteria complicates the challenges of transparency regarding modelling and subjective value judgments as raised above.

In this context, data envelopment (DEA) techniques show some potential (Dyckhoff and Allen 2001; Kuosmanen and Kortelainen 2005). DEA approaches have the advantage of avoiding explicit value judgements in the aggregation process of multiple inputs and outputs. Instead, the weights are determined method-immanently. However, for practical DEA applications, it is required to evaluate a larger set of alternatives with regard to a small selection of efficiency criteria.

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