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LCA Methodology

Bias in Normalization: Causes, Consequences, Detection and Remedies

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DOI: <http://dx.doi.org/10.1065/lca2006.07.260>**Abstract**

Introduction. Normalization is an optional step in LCIA that is used to better understand the relative importance and magnitude of the impact category indicator results. It is used for error checking, as a first step in weighting, and for standalone presentation of results. A normalized score for a certain impact category is obtained by determining the ratio of the category indicator result of the product and that of a reference system, such as the world in a certain year or the population of a specific area in a certain year.

Biased Normalization. In determining these two quantities, the numerator, the denominator, or both can suffer from incompleteness due to a lack of emission data and/or characterisation factors. This leads to what we call a biased normalization. As a consequence, the normalized category indicator result can be too low or too high. Some examples from hypothetical and real case studies demonstrate this.

Consequences of Biased Normalization. Especially when for some impact categories the normalized category indicator result is right, for others too low, and for others too high, severe problems in using normalized scores can show up. It is shown how this may affect the three types of usage of normalized results: error checking, weighting and standalone presentation.

Detection and Remedies of Biased Normalization. Some easy checks are proposed that at least alert the LCA practitioner of the possibility of a biased result. These checks are illustrated for an example system on hydrogen production. A number of remedies of this problem is possible. These are discussed. In particular, case-dependent normalization is shown to solve some problems, but on the expense of creating other problems.

Discussion. It appears that there is only one good solution: databases and tables of characterisation factors must be made more completely, so that the risk of detrimental bias is reduced. On the other hand, the use of the previously introduced checks should become a standard element in LCA practice, and should be facilitated with LCA software.

Keywords: Data gaps; hydrogen production; LCIA; normalization

Introduction

The ISO standard 14042 (Anonymous 2000) for life cycle impact assessment distinguishes a number of steps. One of these is the normalization step, an optional step that is used 'to better understand the relative magnitude for each indicator result'. It is used for 'checking for inconsistencies', for 'providing and communicating information on the relative significance of the indicator results', and for 'preparing for additional procedures, such as grouping, weighting or life cycle interpretation'. Norris (2001) adds to this the resolving of incommensurate units.

A normalized score for a certain impact category is obtained by determining the ratio of the category indicator result of the product and that of a reference system, such as the world in a certain year (Guinée et al. 2002) or the population of a specific area in a certain year (Hauschild & Wenzel 1998).

In theory, the calculation of normalized category indicator results is easy. Below, we follow the mathematical symbols of Heijungs & Suh (2002). Given the quantified elementary flows for the product system, g_i with $i = 1, \dots, n$, and the characterisation factors q_{ji} that link elementary flow i to impact category j , one calculates the (non-normalized) category indicator result for impact category j , denoted as b_j , by

$$b_j = \sum_{i=1}^n q_{ji} g_i \quad (1)$$

The same procedure is used to calculate the normalization factors, i.e. the category indicator results for the reference system. This reference system will in general relate to a given time period, so the elementary flows will be expressed in per-time units, like kg/yr. This is in contrast with the usual set-up of the product system, where a functional unit is generally chosen without a per-time specification. A functional unit like 100 km of car transport leads to elementary flows expressed in units like kg. Hence, g_i is expressed in units like kg, whereas the corresponding quantity for the reference system is expressed in units like kg/yr. To emphasize this time aspect, and to be able to distinguish the two quantities, we will use the convention to add a dot to the per-time variable, hence we use $\dot{g}_i$ for the elementary flows of the reference system. Thus, we construct

$$\dot{b}_j = \sum_{i=1}^n q_{ji} \dot{g}_i \quad (2)$$

The normalized category indicator result for impact category j is now calculated as the ratio of the system's category indicator result b_j and the reference system's category indicator result $\dot{b}_j$. We use the symbol $\tilde{b}_j$ to represent the normalized category indicator result:

$$\tilde{b}_j = \frac{b_j}{\dot{b}_j} \quad (3)$$

In most cases, it will be the ratio of a quantity and a per-time quantity, hence its dimension will often be that of time.

1 Biased Normalization

So far the theoretical and general treatment of normalization. We can observe that the numerator and the denominator in (3) contain a summation over the elementary flows $i = 1, \dots, n$. In principle, these are the same sets of elementary flows. It may happen that these data sets are not the same, for instance when a product system does not emit HCFC-123, while the reference system does emit this. However, it may also happen that the product system in reality emits HCFC-123, but that there are data gaps with respect to this substance, so that the product system will appear to have no such emissions. Of course, the issue of data gaps in LCA has been addressed for a long time (see, e.g. Huijbregts et al. 2001, Guinée et al. 2002, Suh et al. 2004). In most cases, missing data will lead to an underestimation of the LCA result. For instance, inventory results will be too low when emission data are missing, or when no information is available for capital goods. Likewise, characterisation results will be too low when some characterisation factors are unknown. Surely, this may lead to biased comparisons, e.g. when some product alternatives show more data gaps than other product alternatives. For normalization, the situation is a bit more complicated. Incomplete information can lead to underestimations as well as to overestimations, and it can do so in a way that complicates comparisons even more. In this section, we will show how this phenomenon works.

To facilitate the discussion, we will restrict ourselves to only four elementary flows, labelled 1 to 4, and to only two impact categories, 1 and 2. We will assume that these four elementary flows in principle contribute to both impact categories. Now, we form the expression for category indicator result 1 as

$$b_1 = q_{11}g_1 + q_{12}g_2 + q_{13}g_3 + q_{14}g_4 \quad (4)$$

and so on for the second impact category and for the normalized scores. Now, we will assume that some of the data are missing. We will gradually do so, starting with the most simple case, and ending with the most complex case. A final case is taken from a real LCA study.

Hypothetical case 1: missing reference information

We will start with the assumption that there is no information for the reference system for elementary flow 3. This might concern a substance that is outside the national emission statistics, for instance, because it is a minor pollutant, or because monitoring systems do not yet take it into account. The expressions for the normalized category indicator results are as follows:

$$\tilde{b}_1 = \frac{q_{11}g_1 + q_{12}g_2 + q_{13}g_3 + q_{14}g_4}{q_{11}\dot{g}_1 + q_{12}\dot{g}_2 + q_{13}\dot{g}_3 + q_{14}\dot{g}_4} \quad (5)$$

for impact category 1, and

$$\tilde{b}_2 = \frac{q_{21}g_1 + q_{22}g_2 + q_{23}g_3 + q_{24}g_4}{q_{21}\dot{g}_1 + q_{22}\dot{g}_2 + q_{23}\dot{g}_3 + q_{24}\dot{g}_4} \quad (6)$$

for impact category 2. Here terms in parentheses indicate terms that could not be taken into account and are effectively zero. For both impact categories, the normalized result will be too high, because the denominator is too small. The size of this error depends on the combination of the size of the emission of the reference system and the characterisation factors for both impact categories. As the characterisation factor may differ substantially across these impact categories, it may well be that one of the normalized category indicator results is almost correct, while the other one is too high by a factor of two. As a result, the use of normalized scores 'to better understand the relative magnitude for each indicator result' is restricted.

Hypothetical case 2: missing LCI information

Next, we will assume that elementary flow 4 falls outside the LCI. This could happen, for instance, when an LCI database is used in which certain substances are missing. The resulting expressions are

$$\tilde{b}_1 = \frac{q_{11}g_1 + q_{12}g_2 + q_{13}g_3 (+q_{14}g_4)}{q_{11}\dot{g}_1 + q_{12}\dot{g}_2 + q_{13}\dot{g}_3 + q_{14}\dot{g}_4} \quad (7)$$

for impact category 1, and

$$\tilde{b}_2 = \frac{q_{21}g_1 + q_{22}g_2 + q_{23}g_3 (+q_{24}g_4)}{q_{21}\dot{g}_1 + q_{22}\dot{g}_2 + q_{23}\dot{g}_3 + q_{24}\dot{g}_4} \quad (8)$$

for impact category 2. For both impact categories the numerator is too low, and so is the normalized result. The extent to which this shows up depends on the size of the missing intervention and the characterization factors for impact categories 1 and 2. As these latter may differ, the resulting underestimation may be different for the two impact categories.

Hypothetical case 3: missing characterisation factors

As a next step, we will consider what happens when the characterisation factor that measures the contribution of elementary flow 2 to impact category 2 is missing. We find

$$\tilde{b}_1 = \frac{q_{11}g_1 + q_{12}g_2 + q_{13}g_3 + q_{14}g_4}{q_{11}\dot{g}_1 + q_{12}\dot{g}_2 + q_{13}\dot{g}_3 + q_{14}\dot{g}_4} \quad (9)$$

for impact category 1, and

$$\tilde{b}_2 = \frac{q_{21}g_1 + (q_{22}g_2) + q_{23}g_3 + q_{24}g_4}{q_{21}\dot{g}_1 + (q_{22}\dot{g}_2) + q_{23}\dot{g}_3 + q_{24}\dot{g}_4} \quad (10)$$

for impact category 2. The normalized category result for impact category 1 is accurately determined. This is not so for impact category 2, but it is unclear if it is too low or too high (or accidentally correct).

Hypothetical case 4: a mixture of the three cases

All these above-mentioned problems can occur simultaneously. Table 1 specifies which data are available (+) and which are missing (-). For instance, the product system's emission for elementary flows 1, 2 and 3 is known, but is not known for elementary flows 4.

Table 1: Overview of the presence of knowledge in a hypothetical case study

Elementary flow (<i>i</i>)	Product system (<i>g</i>)	Total (<i>g_i</i>)	Characterisation factor for impact category 1 (<i>q_{1i}</i>)	Characterisation factor for impact category 2 (<i>q_{2i}</i>)
1	+	+	+	+
2	+	+	+	–
3	+	–	+	+
4	–	+	+	+

Now, we can form the expressions for the normalized category indicator results:

$$\tilde{h}_1 = \frac{q_{11}g_1 + q_{12}g_2 + q_{13}g_3 (+q_{14}g_4)}{q_{11}\dot{g}_1 + q_{12}\dot{g}_2 (+q_{13}\dot{g}_3) + q_{14}\dot{g}_4} \quad (11)$$

for impact category 1, and

$$\tilde{h}_2 = \frac{q_{21}g_1 (+q_{22}g_2) + q_{23}g_3 (+q_{24}g_4)}{q_{21}\dot{g}_1 (+q_{22}\dot{g}_2) (+q_{23}\dot{g}_3) + q_{24}\dot{g}_4} \quad (12)$$

for impact category 2. Both normalized category results will be wrong, and any direction of miscalculation is possible: both too high, both too low, or one too low and the other one too high.

A real case

From an LCA study on a comparison of production routes for hydrogen (Rovers 2005), the normalization results for one of the alternative systems are shown in Fig. 1.

It is remarkable that the normalized score for marine aquatic ecotoxicity is much larger than that for the other impact categories, although one would not expect that hydrogen production is a relatively marine toxic activity. This leads one to suspect that the normalization results might be biased, either marine aquatic ecotoxicity too high, or the other impact categories too low.

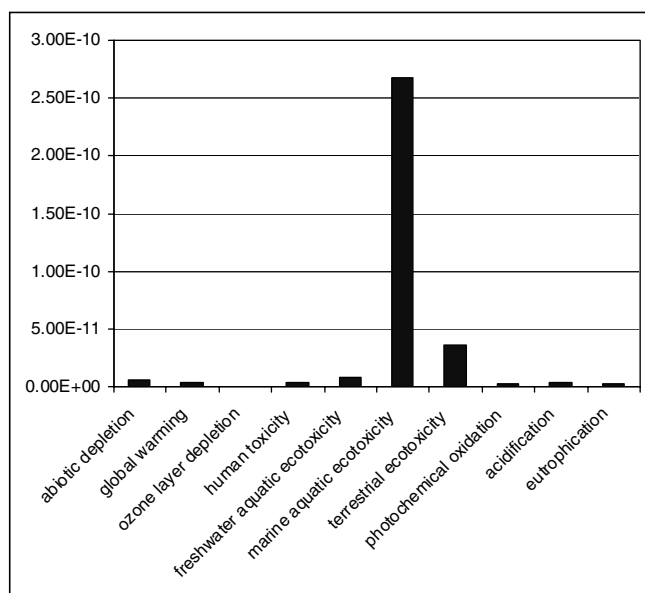


Fig. 1: Results (in yr) of the normalisation step for the production of hydrogen gas, using solar energy

2 Consequences of biased normalization

In the previous section, it has been shown that normalized indicator results may be biased in any direction and with any magnitude. In this section we will explore some consequences of this.

When normalization is used for checking for inconsistencies, new and perhaps larger inconsistencies may be introduced by the normalization itself. Whereas normalization can in principle point out errors, for instance in converting units, a biased normalization may hide these inconsistencies by somehow compensating them. As such, the value of normalization as a straightforward consistency check is limited. Still, it is a useful thing to do: errors introduced in misreading a mg for a kg may still show up, even in a biased normalization.

Another use of normalization is for providing and communicating information on the relative significance of the indicator results. This often presumes that the relative differences are not distorted by the process of normalization. We have shown above that a biased normalization is likely to introduce such distortions.

A final important reason for normalizing indicator results is to prepare for additional procedures, such as grouping, weighting or life cycle interpretation. Indeed, societally derived weighting factors, for instance by panel-stated weighting factors, in general apply to the results after normalization. When these results are incorrect, the weighting factors are applied to the wrong numbers and the procedure may result in an incorrect decision-support.

In conclusion, all uses of normalization will be problematic when the normalized indicator results are wrong.

3 Detection and remedies of biased normalization

We have seen that the results of normalization can be severely incorrect. But we also know the causes of this. This knowledge will be used to design bias detection methods.

Biased normalization is in most cases the result of a bias in the coverage of elementary flows for the product system and for the reference system. An analysis of this coverage is therefore the key to detecting a possible bias. Below, we will illustrate this for the hydrogen production example.

In Fig. 1, it was shown that the normalized result for marine aquatic ecotoxicity was surprisingly high for the production of hydrogen by solar energy. If we analyze the composition of the (unnormalized) indicator result for marine aquatic ecotoxicity (the numerator), we see that it is dominated by a one single substance: hydrogen fluoride to air. The indicator

Table 2: Contribution analysis (in %) of the category indicator result for marine aquatic ecotoxicity for the product system (production of hydrogen gas, using solar energy) (the numerator of the normalized category indicator result) and the reference system (the denominator of the normalized category indicator result)

Environmental flow	Product system	Reference system
Hydrogen fluoride[air]	85	0
Nickel[air]	0	5
Selenium[air]	0	4
Vanadium[air]	0	79
Beryllium[fresh water]	2	0
Cobalt[fresh water]	2	0
Nickel, ion[fresh water]	3	1
Vanadium, ion[fresh water]	5	0
Copper, ion[marine water]	0	3
Zinc, ion[marine water]	0	1
Copper[agric. soil]	0	2

result of the reference system (the denominator) is dominated by an entirely different substance: vanadium to air. A structured comparison is shown in Table 2.

A closer inspection reveals that hydrogen fluoride to air is released by the product, but is missing in the elementary flows of the reference system, for the simple reason that it has not been monitored by the several national emission accounting scheme that formed the basis of the normalization. The characterization factor for hydrogen fluoride is large for marine aquatic ecotoxicity. Thus, given a not too small reference emission of hydrogen fluoride, the denominator of the normalization formula is much too small, and a much too high normalization result is obtained for marine aquatic ecotoxicity.

Some remarks are in place. First it should be noted that the characterisation factor for marine aquatic ecotoxicity by hydrogen fluoride has only been added at a later stage to the other factors (Huijbregts 2000) and is under discussion

(Frischknecht et al. 2004 (p. 27)). True as this may be, the essential point remains that the coverage of substances by the product system and the reference system may be unequal, thereby inducing a biased normalization result, and in particular when the characterisation factor is high.

A second remark concerns the question that the phenomenon outlined above applies to all impact categories that are taken into account. Why would one see a result as in Fig. 1, when the coverage of substances by the product system and the reference system is unequal? The answer to this is that a data gap with respect to one or two important substances will in general affect only a small number of substances. Almost all the substances that occur in the CML baseline method (Guinée et al. 2002) have a characterisation factor for four or less impact categories. Moreover, substances that dominate a certain impact category will not often dominate a second or third impact category as well. It will seldom occur that a data gap for the product or reference system will show up in most or all impact categories, and it is unlikely that such data gaps will have a large effect on two or more impact categories. More specifically, we can speculate that for the major transboundary environmental problems, such as global warming, ozone layer depletion, and acidification, the inventories will be more complete than for more local problems, such as toxicity and odour. As a consequence, we can expect outliers like for marine aquatic ecotoxicity in Fig. 1, while we would be surprised to see them for global warming.

A third remark is that a 85–0 for hydrogen fluoride is more evidence of bias than a 0–79 for vanadium. It is very natural that not all product systems emit all substances. It is less likely, however, that a product system is dominated by a certain emission and that this emission does not show up in the reference system. After all, the reference system used for normalization is normally the world in a certain year or the population of a certain area in that year. And this total is of course made up of the individual product systems. Below Table 3, we will return briefly to this issue.

Table 3: Contribution analysis (in %) of the category indicator result for eight different product systems (numbered 1 to 8) and the reference system. The production of hydrogen gas, using solar energy, which was used in Fig. 1 and Table 2, corresponds to product system 1

Environmental flow	1	2	3	4	5	6	7	8	Reference system
Copper[air]	0	2	3	1	0	0	0	0	0
Hydrogen fluoride[air]	85	27	32	26	74	73	36	53	0
Nickel[air]	0	4	6	5	0	0	3	2	5
Selenium[air]	0	1	2	0	0	0	0	0	4
Vanadium[air]	0	1	2	36	3	3	41	23	79
Barium[fresh water]	0	0	1	2	2	2	2	2	0
Beryllium[fresh water]	2	2	3	2	10	9	3	6	0
Cobalt[fresh water]	2	17	13	6	1	2	1	2	0
Copper, ion[fresh water]	0	1	2	0	1	1	0	0	0
Nickel, ion[fresh water]	3	34	24	12	2	3	4	5	1
Vanadium, ion[fresh water]	5	7	8	4	4	4	4	4	0
Copper, ion[marine water]	0	0	0	0	0	0	0	0	3
Zinc, ion[marine water]	0	0	0	0	0	0	1	0	1
Copper[agric. soil]	0	0	0	0	0	0	0	0	2
Barium[indus. soil]	0	0	0	0	0	0	1	0	0

The procedure around Table 2 may be extended to cover all product alternatives simultaneously. Table 3 shows such a table for eight alternative routes for hydrogen production. It is logical that there are huge differences in the quantitative contributions to marine aquatic ecotoxicity by the eight production routes and the reference system. All of these differences provide reasons for inconsistency checks: errors, data gaps, and other issues that may create unjustified differences across alternatives.

Below Table 2, we remarked that vanadium dominates the reference system, and is absent in the product system, but also speculated that this should not surprise us, as not all product systems emit all substances. This speculation is confirmed by Table 3: vanadium appears to be among the two most important substances for marine aquatic ecotoxicity for 3 of the 8 alternative production routes for hydrogen. Most unit processes are shared by all 8 production routes, so it appears that the 0 for vanadium in the second column of Table 2 and Table 3 is indeed a coincidence.

It is tempting to devise not only schemes for detecting bias, but also for removing bias, or for avoiding it altogether. We will discuss three possibilities: case-dependent normalization, internal normalization and regionalized normalization.

In case-dependent normalization (Guinée et al. 2002, p. 627), the denominator in the normalization formula contains only terms with elementary flows that are covered in the numerator as well. Thus, when copper to marine water is not a part of the product system, it should also be left out of the denominator of the normalization. Although this indeed makes the result less biased, it is questionable whether this is a good solution. In particular: what to do with substances that are emitted for some product alternatives but not for all? Should we have one normalization factor for all alternatives (but case-dependent), or should we even go for an alternative-dependent normalization? Next, when do we say that a substance is covered by the product system? When it occurs in only one out of its hundreds of unit processes? When it occurs only in the foreground system? And finally, how can we expect to use generic weighting factors in combination with non-generic (case-dependent) normalization factors?

The internal normalization (Norris 2001) that is often seen in multicriteria analysis is an approach to normalization without the use of external information. For instance, when there are 5 product alternatives, all impact scores are normalized in a way that they fall between 0 and 1. As observed by Norris (2001), internal normalization does not allow for an assessment of relative significance, whereas external normalization, of which the use of a region's total is a widely used example, has this as an explicit objective. Because the internal normalization does not use regional or global totals, the bias that was described above will not occur. In our view, internal normalization is an extremely useful step in the context of a comparative analysis (Heijungs & Kleijn 2001) or multicriteria analysis (Seppälä et al. 2001). But, given the fact that the ISO standards and the major

textbooks on LCA use the term normalization to refer to the act of putting the results in perspective using external reference information, we are reluctant to consider internal normalization as a form of normalization.

One should note that Udo de Haes et al. (2002) use case-specific normalization as a synonym of internal normalization. Their interpretation accords with what we have referred to above as internal normalization. The case-dependent (or case-specific) normalization is not so often mentioned in the theoretical literature.

Regional normalization, finally, has been proposed in slightly different forms by quite a few authors (e.g., Tolle 1997, Hauschild & Wenzel 1998, Breedveld et al. 1999). In such a normalization set-up one either selects a different reference region for different impact categories (e.g., a state or province for smog, a country for acidification, and the world for climate change), or one selects the region of interest for all impact categories (e.g., a country if the LCA is used for national decision making). It has been argued (Heijungs 1997) that regionalized normalization leads to many paradoxes. For instance, there are numerous products for which the majority of impacts have been caused outside the reference area. Without the need of entering the discussion of the pros and cons of the various forms of regional normalization vis-à-vis a generic global normalization, it should be noted that a regional normalization does not solve the bias issues that were identified in this paper. Even in a regional normalization set-up, it may happen that the substance data set differs between the product system and the reference system.

In conclusion, although internal normalization does not have the identified bias risk, it avoids the use of external reference information, and thereby reduces the usefulness of normalization. Regional normalization does not reduce the risk of biased normalization in the sense discussed, and case-dependent normalization has severe conceptual problems, although it might solve the bias issue.

4 Discussion

In our view, the best strategy to deal with biased normalization is – after filling data gaps – carefully detecting the bias and discussing its relevance. In Table 3, we find that hydrogen fluoride to air, while dominating the numerator, is missing entirely in the denominator. On the other hand, vanadium to air dominates the denominator, but is also very important for some of the product alternatives. It appears that it is not creating a bias, not for the product alternatives in which it is important, but by extension not for the product alternatives in which it is unimportant or absent.

In addition, it may be important to distinguish between an explicit zero and a missing value. There is no information on the emission of hydrogen fluoride in the emission registration databases that were used for the reference system: it is simply an unmonitored substance. In practical calculations, however, this missing value is treated as a zero, hence

the 0 for hydrogen fluoride in the third column of Table 2. Detecting problems in normalization becomes easier when we carefully distinct zeros from missing values in the presentation of tables.

Communication of such detailed information is essential for understanding the extent to which normalization results are biased, and to what extent this might distort a conclusion. LCA software should facilitate the detection of a possible bias in the normalisation results. Table 3 shows a basic scheme that can be used for this.

A future analysis of case studies, with different impact assessment methods and normalization data sets, may give insight to the question which impact categories are particularly vulnerable to the problems exposed in this paper. This is not an easy task, however, and it is not assured to yield clear results. Based on the theoretical analysis, we can already put forward a number of hypotheses:

1. The bias may be large for impact categories that are not so often included in LCA, or that are not well established and not widely recognized. This includes land use, noise, radiation, marine and sediment toxicity, etc.
2. The bias may be large for impact categories that are connected to many substances. This includes different forms of toxicity and radiation.
3. The bias may be small for impact categories that are dominated by just a few substances. This includes climate change and acidification.

The real case example that we investigated (Rovers 2005) showed problems with marine aquatic ecotoxicity. This impact category indeed is (1) not widely recognized, (2) connected to a lot of substances, and (3) not dominated by just a few substances. As such, this single piece of evidence seems to confirm the hypothesis. More evidence is obviously needed. If this hypothesis is right, one would have to be especially careful with normalization for the following impact categories:

- all forms of toxicity (human/aquatic/terrestrial, carcinogenic/respiratory/ecotoxic, etc.);
- photochemical oxidant formation;
- all forms of resource depletion (biotic/abiotic, minerals/fossil fuels);
- radiation;
- any of the more obscure impact categories, like land use noise, odour, and desiccation.

References

- Anonymous (2000): Environmental management – life cycle assessment – life cycle impact assessment (ISO 14042). ISO, Geneva, Switzerland
- Breedveld L, Lafleur M, Blonk H (1999): A Framework for Actualising Normalisation Data in LCA: Experiences in the Netherlands. *Int J LCA* 4 (4) 213–220
- Frischknecht R, Jungbluth N, Althaus H-J, Doka G, Dones R, Hirschier R, Hellweg S, Humbert S, Margni M, Nemecek T, Spielmann M (2004): Implementation of life cycle impact assessment methods. Ecoinvent report No. 3. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland
- Guinée JB, Gorée M, Heijungs R, Huppes G, Kleijn R, Koning A de, Oers L van, Wegener Sleswijk A, Suh S, Udo de Haes HA, Bruijn H de, Duin R van, Huijbregts MAJ (2002): Handbook on life cycle assessment. Operational guide to the ISO standards. Kluwer Academic Publishers, Dordrecht, The Netherlands
- Hauschild M, Wenzel H (1998): Environmental assessment of products. Volume 2: Scientific background. Chapman & Hall, London, United Kingdom
- Heijungs R (1997): Normalization of impact scores in LCA: what, why and how? In: 'Ökobilanzen – Trends und Perspektiven'. Workshop der GDCh-Fachgruppe Umweltchemie und Ökotoxikologie; Gesellschaft Deutscher Chemiker, Frankfurt am Main, Germany
- Heijungs R, Kleijn R (2001): Numerical approaches to life-cycle interpretation. Five examples. *Int J LCA* 6 (3) 141–148
- Heijungs R, Suh S (2002): The computational structure of life cycle assessment. Kluwer Academic Publishers, Dordrecht, The Netherlands
- Huijbregts MAJ (2000): Priority assessment of toxic substances in the frame of LCA. Calculation of toxicity potentials for ethylene oxide and hydrogen fluoride. University of Amsterdam, Amsterdam, The Netherlands
- Huijbregts MAJ, Norris GA, Bretz R, Ciroth A, Maurice B, Bahr B von, Weidema BP, Beaufort ASH de (2001): Framework for Modelling Data Uncertainty in Life Cycle Inventories. *Int J LCA* 6 (3) 127–132
- Norris GA (2001): The Requirement for Congruence in Normalization. *Int J LCA* 6 (2) 85–88
- Rovers V (2005): Life cycle assessments of different production routes for hydrogen. Student report, Leiden University, Leiden, The Netherlands
- Seppälä J, Basson L, Norris GA (2001): Decision Analysis Frameworks for Life-Cycle Impact Assessment. *J Industr Ecol* 5 (4) 45–68
- Suh S, Lenzen M, Treloar GJ, Hondo H, Horvath A, Huppes G, Jolliet O, Klann U, Krewitt W, Moriguchi Y, Munksgaard J, Norris G (2004): System boundary selection in life-cycle inventories using hybrid approaches. *Environ Sci Technol* 38 (3) 657–664
- Tolle DA (1997): Regional Scaling and Normalization in LCIA. Development and Application of Methods. *Int J LCA* 2 (4) 197–208
- Udo de Haes HA, Finnveden G, Goedkoop M, Hauschild M, Hertwich EG, Hofstetter P, Jolliet O, Klöpffer W, Krewitt W, Lindeijer E, Müller-Wenk R, Olsen SI, Pennington DW, Potting J, Steen B (2002): Life-cycle impact assessment: Striving towards best practice. SETAC Press, Pensacola, USA

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