Commentary Articles

How Does ISO/DIS 14042 on Life Cycle Impact Assessment Accommodate Current Best Available Practice?

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Abstract

This article discusses to which extent the forthcoming ISO standard on life cycle impact assessment (ISO/DIS 14042) will be able to accommodate current best available practice in this field. There is, particularly, the risk that the requirement of scientific validity for public comparative assertions cannot be met sufficiently so that the standard may become counterproductive. It is concluded that current best practice for most of the impact categories is compatible with the forthcoming standard. However, difficulties will arise with the toxicity categories, in particular with human toxicity. There is no encompassing indicator is available which does not involve weighting between subcategories. A major improvement would be if, for weighting within categories, internationally accepted value choices would be established as a sufficient condition for public comparative assertions.

Keywords: Category indicators; comparative assertions, public; current best available practice, LCIA; human toxicity; impact categories; ISO/DIS 14042; Life Cycle Impact Assessment (LCIA); scientific validity; toxicity categories; value choices

1 Introduction

In the standardisation of LCA in ISO, the standard on Life Cycle Impact Assessment (LCIA) has now reached the level of a so-called Draft International Standard (ISO/DIS 14042) (International Organization for Standardization, 1998). This means that for the first time the participating national bodies will vote on the given text. In the working group preparing this draft consensus had been reached. Thus, it is likely that the present text will become a standard in about the present form. The preparation of this draft has been a long and difficult process of more than four years of debate involving active participation from national bodies of about 20 countries.

One of the most critical issues concerned the controversy on the acceptability of value choices in characterisation modelling in, what is called, "comparative assertions disclosed to the public"; this means, for public comparisons between product systems. Two standpoints opposed each other:

1. On the one hand, there was the viewpoint, particularly forwarded by the US delegation, stressing that the characterisation in LCIA must be fully based on natural science; the results must be reproducible, independent from the agent who performs the study. This viewpoint was also supported by some European countries, by Japan and a number of developing countries.

2. On the other hand, there was the viewpoint of the majority of European countries, in particular including the Scandinavian countries, Germany, Austria and the Netherlands, that LCA should be regarded as a supporting tool for decision making. In this view normative elements are not a problem, as long as a good procedure is followed with a clearly defined input from stakeholders, and as long as the results are presented in a transparent way.

These two standpoints are linked to different cultural backgrounds. The North-American society is often pictured as a litigation society, with a high risk that companies will sue each other if they come up with unwarranted claims about product superiority. Normative elements in LCIA then have the risk to render the results arbitrary and are therefore open to litigation. Developing countries generally are afraid that industrialised countries will shift from virgin resources towards recycled materials without a clear objective basis. On the other hand, many European countries have a strong cultural background towards consensus building in decision making. They take the current application of Environmental Impact Assessment as example, which also functions quite well despite the existing normative elements.

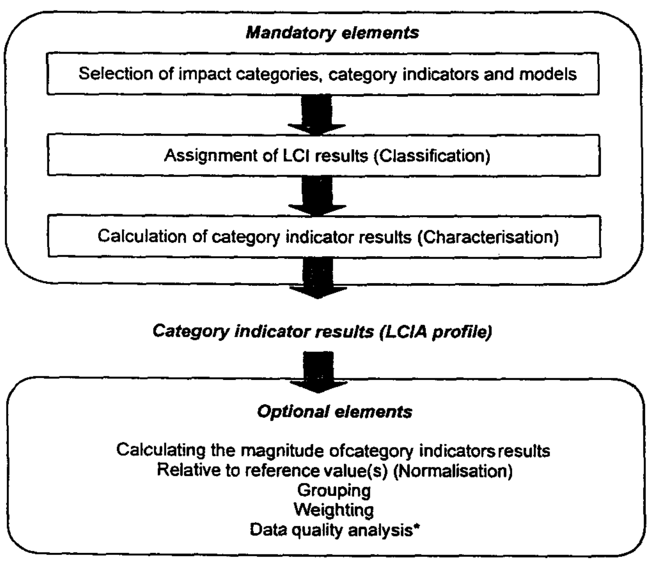
Both standpoints also have their limitations. The North-American viewpoint was rather far removed from current practice, as many impact categories involve points of choice which are beyond natural science (Owens, 1998). This would lead to an inconsistency, because the LCIA standard also requires that for public comparative assertions a sufficiently comprehensive set of indicators is employed. If strict requirements are set which cannot, or not yet, be met, such a comprehensive spectrum of impacts would at the moment be impossible to achieve. Thus, according to this viewpoint, the application on LCIA for public comparative assertions would be factually prohibited.

On the other hand, the European viewpoint would indeed result in the situation that LCAs will produce a different outcome if performed by different agents, at different moments in time, or at different locations. And indeed this would reduce the credibility of LCA and make it more open to litigation, as was feared by the North-American delegations.

Given these two opposing views, how has this gap been bridged? This article discusses the relevant text elements in the present draft standard, which constitute the basis for the consensus in the working group.

2 A Consistent Terminology for LCIA

A first important basis for agreement consisted of the development of a common framework and a common terminology for LCIA. This was by no means an easy process, given the differences in background and in expectations on the outcome of the endeavour. The common framework consists first of all of the different elements which must or can be included (→ *Fig. 1*). As the figure shows, a distinction is made between mandatory and optional elements. The elements rather closely follow the SETAC framework (Consoli et al., 1993) although there are some differences, particularly in the optional elements.



*Mandatory in comparative assertions

Fig. 1: Elements of life cycle impact assessment (ISO/DIS 14042)

Another aspect of the common framework concerns the technical structure of impact assessment as defined for a single impact category (→ *Fig. 2*). Regarding the terminology, an impact category is defined as a class representing environmental issues of concern into which LCI results may be assigned. All environmental processes belonging to this category are called the **environmental mechanism** of that category. In the environmental mechanism, a category indicator is defined, being the basis for characterisation modelling. Thus, the characterisation modelling defines the relationship between the inventory results and the category indicator. In addition, the term environmental relevance is introduced, describing the links between the category indicator and the endpoints of the given category. Category endpoints are the variables which are of direct societal concern, such as human life span or incidence of illnesses, natural resources, valuable ecosystems or species, fossil fuels and mineral ores, monuments and landscapes, man-made materials, etc. (ISO term, but somewhat further explained here); the level of the endpoints is also called the "damage level" (SETAC-Europe term).

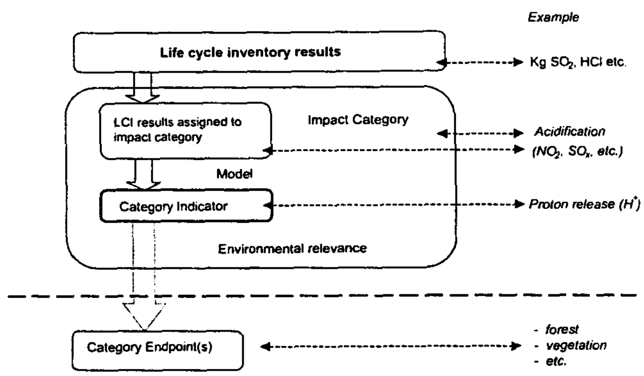


Fig. 2: The concept of "category indicator" as part of the structure of impact assessment for a single impact category (ISO/DIS 14042)

3 The Reach of "Comparative Assertions"

A next important point concerned the precise reach of the term "comparative assertions disclosed to the public". This is very important, as the requirements are much more strict for these applications than for others such as internal use of LCA. In ISO 14040, a comparative assertion is defined as an "environmental claim regarding the superiority or equivalence of one product versus a competing product which performs the same function". This means that single claims which are disclosed to the public will not fall under this heading. In a debate on this issue in the working group, this line was re-established. Single claims may pertain, for instance, to annual reports, but possibly also to scores characterising the environmental characteristics of separate building materials. However, the precise boundary between comparative assertions and single claims is as yet not precisely defined and will have to be further clarified in future practice.

Another point concerns the meaning of "disclosed to the public". An interesting question is whether exchange of comparative information within a branch of industry or within a chain of companies are to be regarded as internal or public; one may well argue that this can be regarded as internal as long as the public is not the addressed party.

4 Main Requirements for Category Indicators

The core requirements regarding public comparative assertions pertain to the quality of the category indicators. There are three requirements:

1. category indicators should be internationally accepted
2. category indicators shall be scientifically and technically valid
3. category indicators shall be environmentally relevant.

We will discuss these requirements in somewhat more detail.

The **first** requirement on international acceptance has not further been specified. There is no strict definition of "international acceptance" in the draft standard, but the discussions pointed at agreements reached in authoritative international bodies like WHO, UN or OECD. International acceptance regards only a "should"; this means that it is desirable, but not strictly necessary. On the other hand, it is also not sufficient, as there are two additional "shalls" to be met. So, if there is international acceptance in an authoritative international body on certain value choices, this would effectively remove arbitrariness, but would still not be sufficient for a public comparative assertion.

The **second** requirement on scientific validity is the core requirement. It is further exemplified with the following text: " ... using a distinct identifiable environmental mechanism, and/or reproducible empirical observation". So there are two ways to meet this criterion. It is important to realise that the *first option* regards knowledge on the environmental mechanism of the impact category as a whole; it does not require strict additivety at the level of a single environmental process as was originally suggested by the US delegation. The *second option* has been added in order to include empirical data, for instance on fate and exposure of substances as a complement to or as an alternative for fate modelling (JOLLIET and CRETTAZ, 1997; 1999), or on NOECs based on animal bio-assays without any knowledge about the underlying processes involved. Furthermore it is important to realise that there is no requirement on the level of the certainty of the modelling.

The second requirement particularly aims to exclude "value choices"; it does not aim to exclude "assumptions" which are of a more technical character. Although not strictly defined in the ISO document, assumptions will particularly include choices which can empirically be validated if sufficient knowledge would be available. They deal for instance with the choice of the background level of the substance in question, the use of animal bio-assays as a model for assessing human toxicity, the use of QSARs, etc. Value choices are of a political, cultural or ethical character. They, for instance, deal with the use of Acceptable Daily Intakes (ADIs) or Reference Doses (RfDs) as a basis for aggregation, because the safety factors contained in these criteria are value based, at least in part. They also deal with different weights which can be attached to human life vs. biodiversity or the availability of resources; or more in general with the relative importance of different impact categories in relation to each other. Assumptions are accepted as part of characterisation modelling for public comparative assertions, value choices are not. It is of interest to observe, however, that climate change modelling as basis for the global warming potentials (GWPs) and ozone depletion modelling as basis for the ozone depletion potentials (ODPs), are regarded as being acceptable for public comparative assertions. This is the case, notwithstanding the fact that the GWP definition makes a choice for specific time periods, whereas the ODP definition chooses for full time integration. These choices are surely not of technical character and do imply values on how to weight the future against the present. Apparently, the international acceptance on these indicators was in fact regarded as a sufficient condition.

The **third requirement** on environmental relevance has been set as a guard against irrelevant indicators. If the link with the category endpoints, i.e. with the things which we really want to protect, is not sufficiently clear, then such an indicator cannot be used. For instance, the weight of material put on a landfill may, without further specification, be not sufficiently relevant as category indicator. It should be noted, however, that the link between the indicator and the endpoints does not need to be of a full quantitative character; qualitative relationships together with an indication of their likelihood are sufficient.

5 The Position of the Category Indicator in the Environmental Mechanism

The draft standard clearly states that the category indicator can be chosen anywhere in the environmental mechanism of the impact category. So it can be chosen at the level of the environmental interventions (the "LCI results" in ISO terminology), at the level of the category endpoints, or somewhere in between (at "midpoint" level). This is significant for a number of reasons:

Firstly, of all it opens the possibility to optimise between uncertainty in characterisation modelling and the environmental relevance of the chosen indicator. The choice of an indicator close to the environmental interventions, as is the case with the GWPs and ODPs, will generally yield relatively certain characterisation results, at the cost of uncertainties in the links to the endpoints and therefore a relatively low environmental relevance. On the other hand, the choice of an indicator at endpoint level will yield rather uncertain characterisation results, but with a high environmental relevance.

Secondly, the choice of the indicator may be relevant in view of the further interpretation of the outcome. Thus, for a final evaluation of characterisation results based on individual preferences, a choice of the indicator at endpoint level seems a prerequisite (NOTARNICOLA et al., 1998), as endpoints are the variables which are of direct societal concern. More in general, social panels will preferably deal with indicators at endpoint level, or at least require clear information on the environmental relevance of category indicators.

Thirdly, and particularly relevant in the present context, the choice of the indicator at endpoint level may open new possibilities for a science based aggregation.

The second and the third point need some further elaboration.

Let us take as an example the **impacts on human health** caused by **toxic substances**. Current category indicators at midpoint level, like those based on Acceptable Daily Intakes (ADIs) or Reference Doses (RfDs), cannot be regarded scientifically valid, because of the safety factors involved which contain strong value based elements. Indicators based on NOEC, ED10 or LD50 data, which result from animal bio-assays, can in themselves be scientifically valid but will generally only enable aggregation within subcategories with common types of impact (cf. BURKE et al., 1995). Science-based aggregation across these subcategories seems difficult to achieve. One may leave it like that and use a great number of human health impact categories instead. However, new options for aggregation may arise if we are able to assess the impacts on human health in terms of "Years of Life Lost" (YLL) or "Years of Life Disabled" (YLD), i.e. the human health indicators developed under auspices of the WHO (MURRAY and LOPEZ, 1996). These indicators are based on a combination of epidemiological human health data or on models using bio-assay data, and may in principle bring science-based ways of aggregation within reach for human toxicity impacts. This also would open up the possibility to include other types of impact on human health, as, for instance, caused by fine dust or by physical casualties. For these reasons they are now being further elaborated for application in LCIA context (HOFSTETTER, 1998; GOEDKOOP et al., 1998). Data on YLL or YLD enable to aggregate across different types of causes, but still do not result in one single encompassing category indicator for human toxicity. This can in principle be achieved by performing a further aggregation of the YLL and YLD results in terms of the so-called disability adjusted life years, the DALYs, which weight life shortening and different types of disability against each other (MURRAY and LOPEZ, 1996). The resulting DALY based indicator would indeed be quite manageable, but would not meet the requirement on scientific validity because of the value choices involved in the weighting step.

Another example of modelling at endpoint level concerns **ecotoxicity**. The main problem here concerns the variety of species involved. Comparable to human toxicity, an aggregation on the basis of indicators which include safety factors (because of too small numbers of test species) may not sufficiently meet the requirement on scientific validity (see however also Emans et al., 1993, arguing in favour for the scientific validity of these factors). A step forward concerns the use of indicators based on PNEC data, i.e. the "predicted no effect concentrations" indicating the concentration at which a given percentage (say, for instance, 95%) of the species of an ecosystem is just protected (ALDENBERG and SLOB, 1991). This can be regarded as a common category indicator at midpoint level, with arguably a scientifically valid basis. A recent list of characterisation factors for toxic substances, which is based on the PNEC concept, includes 180 substances (HUIJBREGTS, 1999). A further improvement in scientific validity may well concern the use of the concept of the "potentially affected fraction of species" of an ecosystem (PAF) (KLEPPER and VAN DER MEENT, 1997). Indicators based on this PAF concept concern the endpoint level of the environmental mechanism. Their contribution to scientific validity lies in the fact that also the aggregation itself has a better founded basis, as the non-linearity of the dose-response curve of the impacts on the species composition is taken into account. For the above three reasons, modelling up to the endpoint level is now a major focus for LCIA development (HOFSTETTER, 1998; GOEDKOOP et al., 1998).

6 No Weighting Across Impact Categories

Given the science basis requirement, weighting across impact categories is not allowed for public comparative assertions. Because weighting across impact categories will always, at least in part, be based on value choices, this requirement cannot be removed by further scientific development. This implies that results of LCIA cannot be expressed in terms of a single encompassing eco-indicator (cf. GOEDKOOP, et al., 1998), if they are to be used for public comparative assertions. For specific applications, which do require routine comparisons between large numbers of products, this indeed can be a severe limitation.

As a compromise, the ISO working group now has proposed to accept ranking between categories. This means that in the (normalised) environmental profile of a product (i.e. the characterisation results) the relative importance of the categories can be indicated; for instance, the bars representing the results can be given in different colours indicating their relative importance. However, this does not solve the above mentioned limitation. A solution could be that weighting is not allowed under ISO umbrella, but that the results can be weighted afterwards, outside this umbrella. Of course, it should then be clearly expressed that the weighted results are not obtained under rules of the ISO standard. However, the present text is clearly at variance with this rather artificial possibility. A more preferable option would be to aim at the establishment of a generic global set of weighting values. This would remove arbitrariness, but would also involve a change in the present draft standard that international acceptance is a sufficient condition for a comparative assertion.

7 Discussion

Taken together, to which extent can best current practice meet the coming ISO standard on LCIA? This question is quite relevant, because a standard which would invalidate such best practice would rather be counterproductive than helpful.

In order to give a short overview of the implications for characterisation modelling we follow the list of impact categories which has recently been established by the SETAC-Europe working group on impact assessment in a guiding document towards best available practice (UDO DE HAES et al., 1999).

For *abiotic resources*, there seem to be options which do comply with the standard, as for example, the energy required for their extraction, the exergy content of the resources, or the total material requirement associated with their extraction.

For *biotic resources*, one can think of a measure of depletion, which relates their present availability, their present rate of use and their potential for recovery.

For *land use*, science-based indicators may aim at the area of exclusive land use; or of the vegetation structure as basis for a number of important life support functions. Indicators dealing with biodiversity may well be based on the loss of species diversity (e.g. of vascular plant species), in combination with the area involved (LINDEIJER, 1998). It is interesting to investigate whether a link to the potentially affected fraction of species of an ecosystem (the PAF concept) will be feasible in the near future. Climate change and stratospheric ozone depletion explicitly comply with the present draft standard, although values are involved in the choice of the time horizon.

For *human toxicity*, part of the problems can be solved by choosing the indicators at endpoint level: years of life lost (YLL) or years of life disabled (YLD). These indicators are encompassing in that they combine different types of underlying causes. But they are in fact defined at a subcategory level and need a weighting step for further aggregation (i.e. the disability adjusted life years or DALYs). So no single science-based indicator is as yet available for human toxicity, even if the modelling up to the endpoint level proves feasible. If it is not feasible or only in part feasible, then we must choose a set of subcategory indicators at midpoint level like those based on NOEC or ED10 criteria resulting from animal bio-assays. These have a lower or partial environmental relevance; they will better meet the requirements on scientific validity, but again they cannot easily be combined to one single category indicator.

For *ecotoxicity*, the picture is conceptually somewhat better, as already at midpoint level PNEC based indicators are arguably ISO compatible. Indicators at endpoint level, based on the PAF concept can in principle further add to the scientific validity of the aggregation. And comparable to human toxicity, the PAF concept may enable the inclusion of other category results in the aggregation. We have to keep in mind, however, that for both the toxicity categories there is a very large data problem to address in view of the extreme number of substances involved.

The *smog category* may well be split into human toxic impacts, mainly due to small particles, and the creation of photo-oxidants. For the latter, different indicators are available which may well meet ISO requirements, in particular the POCPs or photo-oxidant creation potentials (DERWENT et al., 1998) and the MIRs or maximum incremental reactivity (CARTER, 1994). Expressing damages to human health due to ozone creation in term of YLL or YLD would allow to compare these with those due to other toxic substances. Acidification will give no severe problems, as the release of protons seems to be an ISO-compatible indicator (POTTING et al., 1998).

For *nutrification*, the most commonly used approach concerns a stoechiometric addition of P and N, implying an addition according to their relative occurrence in biota. This does not take limiting conditions for the nutrients into account. However, at the level of nutrient availability, this can be defended as a sound approach, leaving the uncertainty of impacts to the environmental relevance of the indicator.

8 Conclusion

The framework which has now been presented by the ISO working group, which requires scientific validity of category indicators as strict requirement for public comparative assertions, appears to be useful for guiding aggregation within the majority of the currently distinguished impact categories in LCIA. For most of these categories there is still considerable room for choice. To some extent this is desirable, as the different available options may meet different application requirements. To some extent this is a matter of differences in scientific quality. It is a task of the SETAC-Europe working group on LCIA to identify, where possible, the best available practice in this field. Major difficulties appear to be present to accommodate the toxicity categories under the coming standard. Here the definition of the category indicator at the endpoint level opens perspectives for increasing scientific validity of aggregation within these categories. But particularly for human toxicity, there is still weighting involved if one wants to express the results in terms of one category indicator. We suggest that for the aggregation of interventions within impact categories, as for instance human toxicity, international acceptance of value choices will be established as a sufficient condition for public comparative assertions in the coming ISO standard on LCIA.

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Please note also the Report from the 3rd Conference of the Society of Automotive Engineers (SAE) on Total Life Cycle in Graz, Styria, Austria, December 1998 (→ *pp. 121-122*).