ANEC POSITION

ENVIRONMENTAL ASSESSMENT GOES ASTRAY

A CRITIQUE OF ENVIRONMENTAL FOOTPRINT METHODOLOGY AND ITS INGREDIENTS

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Foreword

The following paper was prepared in response to the Commission's DG Environment effort to develop "a harmonised methodology for the calculation of the environmental footprint of products (including carbon footprint)"[1] with the aim "to reduce the environmental impacts of goods and services".

This method builds "on the International Reference Life Cycle Data System (ILCD) Handbook as well as other existing methodological standards and guidance documents (ISO 14040-44, PAS 2050, BP X30, WRI/WBCSD GHG protocol, Sustainability Consortium, ISO 14025, Ecological Footprint, etc)".

Consequently, the critique is not focussed on the emerging organisational (OEF) and product (PEF) environmental footprint methodology but addresses the relevant underlying concepts and instruments. In fact, the OEF and PEF methodologies are by no means new, they rather constitute a remix of existing tools and related guidance.

The OEF/PEF initiative of DG Environment was unfortunately not preceded by an in-depth investigation about fundamental limitations of existing approaches (in particular of Life Cycle Assessment, LCA) on the one hand, and a broad discussion about stakeholder perceptions and expectations regarding environmental assessment and related indicators on the other hand. This was a serious omission resulting in a questionable outcome with a potential to constrain environmental assessment and mislead environmental policy.

Any method development should not be seen as an end in itself. A method is suitable only if it fulfils its target – in this case to contribute to environmental policy making in a meaningful manner. Hence, a methodology discussion must have a wider scope – it must be embedded in a system of political target setting and decision making.

Last but not least, instruments must show their value in practical life before existing and well-proven tools are abandoned. Otherwise serious damage is likely to occur.

[1] http://ec.europa.eu/environment/eussd/product_footprint.htm

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Summary

The Commission develops a harmonised methodology for the calculation of the environmental footprint of products, services and organisations with a view to assess, display and benchmark their environmental performance based on a Life Cycle Assessment (LCA) approach. The proposed method is fundamentally flawed and not fit for the purpose for different reasons, which we examine in this paper.

LCA methodology has unique advantages when analysing the environmental performance of products as it allows in principle – based on an accounting of all relevant material flows throughout the entire life cycle – to obtain a complete picture of certain environmental burdens associated with a product. This allows comparisons across technological boundaries and to identify relevant stages in the life cycle, as well as improvement options.

By contrast, LCA methodology features fundamental shortcomings including dependency on numerous subjective choices, lack of adequate data and limited precision. The history of LCA has shown clearly these constraints with heated debates following publications of comparative studies and accusations of manipulation. In some cases European policy was completely misguided based on flawed LCA results (see e.g. biofuels). These limitations cannot be overcome by another layer of rules in addition to existing standards – they are inherent in the system of life cycle assessment.

In addition, LCA is definitely not THE tool which can suitably characterize all environmental impacts. Many impacts cannot be reasonably related to reference flows referring to a functional unit and aggregated throughout the life cycle, because the effects are space, time and threshold dependent. Some of the LCA impact categories are of questionable scientific validity or outdated. Sound environmental assessments require a mix of different tools (environmental impact assessment, human health and environmental risk assessment, technology assessment, etc.) taking due account of their strengths and weaknesses.

Life cycle assessment is a suitable tool for orientation at the onset of indicator development or regulatory requirement setting. However, suitable production, consumption or disposal indicators are typically more robust, in many ways more meaningful or relevant, cheaper; they can be measured and are easier to verify.

Consumer information based on a choice of LCA indicators is useless and a step in the wrong direction – even if linked to rating scales which will often not be possible. The reason is that the poor precision of the method will not allow the establishment of bands comparable to the energy labelling scheme (where, despite well-defined test protocols, tolerances can be as big as the width of one band). Irrespective of this, consumers need a clear indication of a superior product by a traditional type I label. The significance of (several) life cycle indicator results is difficult to assess even for experts, let alone the average consumer. Apart from that, such indicators will be of little interest as they are not related to consumer needs. Bombarding consumers with such information may meet some advertising needs to give some

corporation the glow of sustainability – as in case of questionable carbon footprint labels – but has little to do with provision of sound environmental information to assist purchasing decision making.

Corporate indicators currently used (e.g. in sustainability reports), following the guidelines of the Global Reporting Initiative (GRI) are of little use as they are based on an indication of total amounts (e.g. of energy) per organisation per year and do not allow for comparisons between organisations and benchmarking. The latter is possible only at the process or product level under certain conditions, e.g. using a precise measurement protocol and appropriate metrics by relating environmental burdens to output units (e.g. kg NOx per ton cement). Extending the (questionable) GRI approach by making use of LCA methodology misses the point – it adds complexity to no avail.

A reasonable approach must identify the relevant indicators for the relevant products and organisations using a broad range of assessment methods, and must not follow a one-size-fits-all methodology and collect data for the sake of collecting data. This task cannot be shifted to LCA service providers but must be taken first at the political level. Hence, it is important to develop a framework for indicator development embedded in the system of political decision making translating priority environmental concerns and broad target setting into specific quantified environmental demands at the macro level (EU, MSs), as well as organisational and product level. To this end we suggest a framework for environmental indicator identification which is illustrated using some examples in chapter 7 of this paper.

Finally, this framework and the resulting choice of indicators must be linked to existing policy instruments and applied in a co-ordinated manner.

It would have been useful to start the debate about a harmonised methodology from a broader perspective including a discussion about pros and cons of current practices and – based on that – to identify needs for improvement covering all dimensions of the subject in question. Instead, the European Commission embarked on a detailed methodological development in a rather confined way. This may lead to questionable outcomes – the promotion of a rather one-dimensional tool at the expense of well-established approaches which are in many ways superior to what is suggested. It is time to pause for a rethink.

Introduction

The ANEC reservations, related to the Environmental Footprint (EF) methodology proposed by the European Commission, evolved over the course of many years of intensive examination, research, discussion, political positioning and involvement in standardisation concerning LCA, EPD, carbon footprint, corporate environmental indicators and performance evaluation. Last but not least, it is necessary to look at the real world of LCA including all the controversies its application – particularly in a public policy context - has triggered.

It is far from obvious that LCA methodology is THE method of choice to suitably characterise environmental impacts from products, let alone from corporations. An appropriate all-embracing indicator system must build on all available instruments and methods and should start with a thorough analysis of their strengths and weaknesses. The ANEC criticism of the LCA methodology and related declaration/labelling tools is shared by other commentators. We regret the Commission started the EF method development without a broad stakeholder debate on the indispensable elements of a comprehensive systematic environmental assessment tool embedded in a policy framework**.** A first suggestion for such an approach is included in this paper.

In order to illustrate the alternative approach and related principles we suggest for consideration in chapter 7, we start explaining experience and research related to LCA, EPDs and CFP information underpinning ANEC standpoints (chapter 1). We then continue with other LCA limitation reviews (chapter 2) and demonstration of case studies (chapter 3). Furthermore, we examine whether standardisation can help (chapter 4) and we give our view on corporate indicators, highlighting the need for true benchmarking between companies. We make a specific remark on PEF methodology, to then conclude with the explanation of the basic principles of the alternative approach we hereby propose for consideration.

1. Research based ANEC positions

1.1 LCA methodology

The goal of a study[2] commissioned by ANEC was to investigate LCA methodology more thoroughly with respect to its suitability for labelling, product differentiation and benchmarking, and to give proposals as to how its inherent shortcomings could be solved. The major conclusions from this project are:

Benefits of LCA: The undisputed benefit of LCA is – as the name suggests – providing a complete coverage of environmental impacts throughout the life cycle "from cradle to grave". Thereby LCA allows for comparisons of different technologies delivering similar functions (e.g. different types of fuels). It also allows for identification of the lifecycle stages with the highest contributions to overall lifecycle impacts.

Incompleteness of LCA: The above holds true only for those environmental aspects which are actually covered by an LCA and which can be quantified and summarised (aggregated), such as energy consumption or greenhouse gases. Unfortunately many important aspects do not fall in this category and in a number of cases quantification is not possible. Examples are impacts from agricultural land use such as soil erosion, conservation of soil organic matter, or biodiversity. In some cases potential impacts are unknown but should be avoided following the precautionary principle (e.g. persistent organic chemicals - POPs). Furthermore, many impacts cannot be aggregated as they are site-specific and depend on local concentrations of pollutants, rather than on total life cycle releases (e.g. noise, dust, or indoor air

[2] ANEC study "Environmental product indicators and benchmarks in the context of environmental labels and declarations", performed by Öko-Institut, December 2008 https://www.anec.eu/images/Publications/technical-studies/ANEC-RT-2008-ENV-005final.pdf

pollution). Finally, the impacts may also depend on local conditions (e.g. water consumption in dry areas versus wet areas). Hence, LCA methodology based on a functional unit approach does not and cannot provide for comprehensive environmental assessments.

Limited accuracy of LCA restricts product comparisons: The precision of LCA results is limited by available resources, data gaps and data quality constraints (e.g. temporal and geographical coverage, need to use generic data rather than site-specific data, complex and changing logistics and supply chains). The error margin of an LCA will differ widely and will - in particular for complex products - easily exceed 10% for energy and greenhouse gases and 20% for other impact categories (ideal values which are sometimes mentioned in literature). As a result of the lack of accuracy, LCA does not appear well suited for comparisons of similar products and will typically not allow for product differentiation. Even if only primary data are used (rather than data from generic databases), the physical nature of these production processes makes it likely that the data are so similar that the identified differences are smaller than the error margin. Hence, any labelling scheme will have to focus on issues such as material content or energy consumption in the use phase, meaning that LCA would not give any added value compared to current eco-labelling practices, but would simply require unnecessary efforts for data collection and compilation.

Further complications are related to different methodological choices and data selections by different LCA practitioners, with industry potentially being tempted to 'embellish' data. Hence, methodological conventions, going beyond standards such as ISO 14040/44[3], as well as a common database, would have to be approved by the labelling or criteria-setting institution.

Identification of all significant environmental aspects: LCA needs to be complemented by other assessment tools - referred to as "additional environmental information" in ISO 14025 on Environmental Product Declarations (EPDs). The selection of product categories, their significant environmental aspects, relevant life cycle phases, and the assessment tools and methodological conventions, mentioned above, should be regulated at the political level and should involve relevant stakeholders including consumer and environmental organisations. The current procedure under the Eco-design Directive (2005/32/EC) can be seen as a positive development in that it aims to integrate both scientific input and stakeholder perspectives. A process for a more inclusive environmental assessment of products is suggested, including the determination of "significant" environmental aspects by means of a hot spot analysis (ABC analysis – i.e. grouping in order of their estimated importance).

LCA for orientation and coarse assessments: Comparisons between different product categories are less demanding in terms of accuracy and can be made on

[3] ISO 14040:2006 Environmental Management – LCA –Principles & Framework; ISO 14044: 2006 Environmental Management – LCA –Requirements & Guidelines

the basis of (agreed) generic data. In such cases, product differences are much bigger compared with a narrowly defined product family.

Overall, ANEC concluded among other things that: Environmental indicators and benchmarks used in the traditional (Type I) eco-label schemes, or in Best Available Technique Reference (BREF) documents for specific life cycle phases, will in many cases be superior to LCA indicators – in terms of coverage, data availability, and precision. For similar products, LCA indicators normally will not offer a benefit. This holds even truer when a large proportion of a burden occurs in one phase of the life cycle. The main function of LCA is to identify relevant life cycle stages, "hot spots" and improvement options for certain environmental aspects.

1.2 Questionable benefits of EPD/CFP information

Type III environmental declarations (sometimes referred to as Environmental Product Declarations – EPDs) are unsuitable for consumers and other stakeholders in a similar situation (e.g. public procurement), as this kind of environmental information does not allow for the identification of environmentally-superior products lacking benchmarks and rating scales (colour/letter codes). They are a good marketing instrument pretending environmental superiority where, in fact, there are only (questionable) data. This makes them quite popular among certain industry circles – some kind of environmental label can be purchased without complying with any particular performance requirements.

In another ANEC study[4] published in 2008, the usefulness of EPDs was investigated in more detail. It was suggested to establish so-called "Environmental Data Sheets (EDS)" which combine indicators from various traditional instruments (e.g. energy labelling and type I ecolabels) with LCA indicators. The latter - normalised to the impacts created by an 'average citizen' and expressed as percentage of it – using a graded, colour band scale similar to the EU Energy Label was intended to compare different categories of products from different product families. Hence, the LCA indicators referred to an average product of a certain kind rather than to a specific one. The purpose was – as complementary information - to illustrate the relative contribution of certain products to the total environmental load of a citizen, but not to compare similar products due to the uncertainties of LCA results as explained above. The focus in the EDS approach is, however, on production or use stage indicators using appropriate benchmarks. It includes, for instance, quantitative information on parameters used in eco-label criteria (e.g. the amount of TiO_2 in paints).

Along the same lines, ANEC expressed a strong rejection of the Carbon Footprint labels, such as the one issued by the British Carbon Trust. One of the ANEC studies

[4] ANEC study "Benchmarking and additional environmental information in the context of Type III environmental declarations", performed by Force Technology, December 2007 http://www.anec.eu/attachments/ANEC-R&T-2008-ENV-003final.pdf

looked specifically at carbon labels[5]. The major conclusion: "Single number CO_2 labels make no sense". One reason for this is that a single CO_2 figure allocated to a product reflects a precision and conclusiveness which cannot be achieved using available methodologies. Further flaws include – as in case of EPDs – the absence of efficiency classes and rating (which will, however, be difficult to establish in view of the big uncertainty of results which may be of the same order as performance differences). There is also a risk that the display of such a label makes consumers believe that the product might be better than another without a label. It became clear that climate change issues can be more easily (to a certain extent) addressed by energy efficiency parameters. The latter is cheaper and more reliable as it addresses a key parameter which can be directly measured and is easily verifiable. In the case of other product groups, such as food products, PCF is a good basis for the development of general recommendations addressed to consumers taking into account climate change issues (e.g. "eat regional and seasonal food", "eat less meat" etc.), but these recommendations must not be communicated as PCF. In any case, the preferred option is to incorporate greenhouse gas considerations in type I labels, rather than having a label addressing just a single issue in a questionable way.

Recent developments (e.g. in France) to use a selection of life cycle indicators for consumer information (often 3 different ones) are of major concern. The significance of (several) life cycle indicator results is difficult to assess even for experts, let alone the average consumer. Moreover, such indicators will be of little interest as they are not related to consumer needs. Bombarding consumers with such information may meet some advertising needs to give some corporation a glow of sustainability – as in case of questionable carbon footprint labels – but has little to do with provision of sound environmental information to assist purchasing decision making. Consumer information based on a choice of LCA indicators is useless and a step in the wrong direction – even if linked to rating scales which will often not be possible. In fact, as the poor precision of the method will not allow the establishment of establish bands, as in the case of the energy labelling scheme (which still creates some troubles because of the tolerances of the test methods which can as big as the width of one band). Irrespective of this, consumers need a clear indication of a superior product by a traditional type I label.

A recent study[6] charged by the Commission reinforces these concerns. The purpose of the study was to identify options to communicate EF information to consumers. Based on a literature research, some initial designs were created and subsequently further refined based on the feedback of selected consumer surveys in several countries (Italy, Poland and Sweden). The final result is presented below.

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[5] ANEC study "Requirements on Consumer Information about Product Carbon Footprint", performed by Öko-Institut, February 2010 http://www.anec.eu/attachments/ANEC-R&T-2010-ENV-003final.pdf

[6] "Different options for communicating environmental information for products", BIO Intelligence Service, February 2012 http://ec.europa.eu/environment/eussd/pdf/footprint/ProductsCommunication_Final%20Report.pdf http://ec.europa.eu/environment/eussd/pdf/footprint/ProductsCommunication_Annex.pdf

The consumers could, of course, judge only the layouts but not the technical validity of the approach (and were not asked whether such labels are preferable compared to type I ecolabels). Not surprisingly, they favoured a colour/letter code system in the spirit of the EU energy labelling system, and were in favour of an aggregated indicator. Consumers understand that A is better than B, and green is better than red. However, there is actually no scientific basis for calculating an aggregated overall numerical result. In that sense the label is highly questionable and misleading. In addition, the individual impact category results (whatever they may stand for) are highly confusing. How should the different ratings be interpreted – in particular, when they point in different directions (global warming green but air pollution red)? Should a consumer choose according to his/her preference?

More importantly, this design using different ratings for individual impact categories is a significant step backwards from the EU and national ecolabels awarding only the top performers (the best 10-30%). Such labels give a clear and unambiguous message: this product has an excellent performance. There is no need to complement or even substitute type I labels with highly questionable LCA indicator results – even if presented using nice colours! In real life, such labels would be disregarded by consumers anyway.

Furthermore, the technical feasibility to identify reliably 3-4 distinct classes of performance (using relevant individual life cycle indicators) of similar products with sufficient precision is still to be demonstrated.

1.3 Construction

A good example for an entirely misguided environmental policy based on a LCA indicator system is the construction sector. Based on a mandate of the Commission, the European standardisation committee CEN/TC 350 "Sustainability of construction works" developed standards for the environmental assessment of building products and buildings. These standards were heavily criticised by ANEC in the position paper "Sustainable construction – a building site without end. Alternatives to flawed standards"[7].

One of the key questions is whether an LCA indicator system for buildings is useful. It is a well-known fact that the energy consumption in the use stage of a building outperforms by far the energy consumption in all other life cycle stages. This also applies to other related environmental impacts. The so-called IMPRO-Building study - Environmental Improvement Potentials of Residential Buildings[8] - came, for instance, to the conclusion that the primary energy demand related to the use stage amounts to about 80% of the total energy consumption of new European buildings. It should be noted, however, that the use stage was assumed to be just 40 years in this study. This means that the share of the use stage could be even higher when more realistic service life times are assumed. From this follows that energy efficiency of the use stage is of primary importance. Construction and end of life treatment are of low importance for the total energy balance.

This applies even more to the existing building stock having thermal insulation which is typically much worse than that of new buildings conforming to new building regulations. An LCA approach for existing buildings would make limited sense because the environmental burdens associated with manufacturing of building products and construction is unknown. Beyond that, such burdens are irrelevant because they have occurred in the past and cannot be influenced anyway.

In addition, the improvement potential concerning energy consumption can be assumed to be the highest in the use stage – both for new and old buildings. A meaningful approach in the field of environmental indicators must take into account the options for improvement. If significant efficiency gains are not feasible, indicators are pointless.

Finally, it should be noted that the life cycle energy consumption is irrelevant for the user of the building who is mainly interested in the energy bill.

From this follows that – as far as energy and related impacts are concerned – the use stage indicator is the relevant one to be employed both in a regulatory context as well as in voluntary schemes. For other products and/or environmental aspects, this may be different. In case of paper, for instance, the production is, of course, the most relevant stage of the life cycle with respect to energy consumption,

Generic LCA model studies are highly important e.g. to identify the relevant stages in the life cycle of a product. But there is little, if any, benefit to use life cycle indicators for labelling, certification or law making. On the contrary, this would introduce only additional costs and increasing uncertainty of results, for instance, because of highly subjective choices for establishing scenarios (such as service life

[7] ANEC position paper "Sustainable construction – a building site without end. Alternatives to flawed standards", September 2011
http://www.anec.eu/attachments/ANEC-ENV-2011-G-037.pdf

[8] Environmental Improvement Potentials of Residential Buildings (IMPRO-Building), JRC/IPTS, 2008, 5.3.1, fig. 5.11, pg. 60 https://op.europa.eu/en/publication-detail/-/publication/85dcd023-6800-400a-bb6f-cbbdb3826f9b

time of building and construction products, waste management options, etc). It should be noted that the service life time of a house is not known. Whatever number is chosen – 40 years, 60 years, 80 years, 100 years - is an arbitrary choice (the same applies to the service life time of its components).

It would be much more beneficial to properly enforce existing rules for energy certificates to ensure that the correct values are indicated (currently the situation is unsatisfactory), and to harmonise these rules in Europe, rather than introducing an LCA scheme for buildings which delivers no added value as far as energy consumption is concerned. The most important political target is to strengthen energy consumption requirements for the building stock anyway.

It should be noted that this approach does not address many important other environmental issues, as debunked in the ANEC paper (such as indoor pollution or construction site related noise and dust). But this entails another discussion.

The IMPRO-Building study also showed that there was a good correlation between primary energy consumption and the values for the impact categories global warming, ozone depletion, acidification, eutrophication and photochemical ozone creation. Hence, at least for buildings, these indicators do not provide any substantive additional information. One could say that they just express energy consumption using different headings.

Only few products contribute to the large proportion of energy embedded in building products: essentially basement, walls, floors/ceilings, and perhaps to a lesser extent windows and roofs. This suggests that embedded energy rather than all impact indicators (see reasoning above) should be addressed – and this only for a limited number of construction products or structural elements rather than prescribing this for all products. However, the improvement potential is limited.

All this clearly shows that a one-size-fits-all approach makes little sense. We need to identify the relevant indicators for the relevant products and relevant life cycle stages in a resource efficient manner.

There is also another lesson to be learnt from the construction sector regarding the use of EPDs. Industry goes for averaged "branch EPDs", i.e. the intention is that all manufacturers of a certain construction product provide the same figures to avoid – as they put it – ruinous environmental competition. This means to eliminate a key driver in environmental improvement by making visible the performance differences, and to award the good and to punish the bad.

1.4 LCA Impact assessment

LCA impact assessment relies on the concept of assigning life cycle releases (life cycle inventory results) to selected impact categories (classification), to identify a suitable characterization model and to determine the characterisation factors reflecting the different potencies of the individual contributing compounds.

The most widely used impact assessment categories in LCA studies include global warming, ozone depletion, acidification, eutrophication and photochemical ozone creation. However, the environmental relevance of some of these indicators developed in the late 1980s and early 1990s can be questioned. For instance, the problem of ozone depletion can be regarded as mostly settled under the Montreal Protocol: the relevant substances have been banned and the ozone layer is recovering. Acid rain, resulting in dying of forests and lakes, was an extremely important subject in the 1980s but is no longer considered highly relevant nowadays. Clearly, there are much more pressing needs such as resource (over)consumption (e.g. critical biotic resource extraction, water scarcity, etc.), land use, chemical pollution (including cocktail effect, endocrine disrupters, nanomaterials, POPs, etc.), particles, noise, biodiversity loss and so forth.

One serious disadvantage of these indicators is that they nebulise the origin and contribution of individual compounds to the indicator results – e.g. that NOx from combustion of fuels is a major element for acidification, eutrophication and photochemical ozone creation – leading to a distraction from improvement options. For instance, one can go for "low NOx" burners to reduce combustion related impacts - but not for "low acidification" boilers.

A more fundamental question is whether environmental impacts can be suitably modelled on the basis of life cycle releases, bearing in mind that usually spatial and temporal conditions of releases significantly influence the environmental consequences. Only in case of greenhouse gases one can argue that it does not matter where on the globe the GHG molecules are released – they all contribute to the same global effect. It is also irrelevant – within limits – when the release occurs and what precisely the background concentration is. There are no thresholds below which no effect occurs and the effect increases proportionally with the amount released (whatever functional unit is chosen). Finally, all greenhouse gases share a distinct mechanism and contribute to the same effect: radiative absorption. But these are ideal conditions which are fulfilled only for greenhouse gases. In all other cases one or more of the factors mentioned above have a decisive influence on whether or not an effect occurs, and on its magnitude.

The impact category "human toxicity" may serve as an example. Toxic effects of chemicals are based on quite different mechanisms which neither allows aggregation nor (scientifically) sound classification and application of characterization factors. In other words, any "toxicity number" means to add apples and pears – chemicals with quite different modes of action – and is therefore questionable. Owens[9] proposed a classification scheme for non-cancer endpoints including 13 categories, but failed to identify suitable characterisation factors: "*Importantly, the toxicological critical effects observed, even for the same target organ or system, differed from chemical to chemical and were not equivalent. Using hepatotoxicity as an example, critical effects included changes in organ weight, a variety of different histopathological changes, and changes in circulating hepatic enzyme levels. Therefore, no universal, common basis was identified for biological*

[9] "Chemical toxicity indicators for human health: case study for classification of chronic noncancer chemical hazards in life-cycle assessment", Owens JW., Environ Toxicol Chem. 2002 Jan; 21(1):207-25.

equivalency in order to compare or to aggregate chemicals into an overall toxicity score".

Toxic effects are also strongly related to space and time characteristics which determine whether thresholds are exceeded. It makes a big difference whether, say, a ton of a poison is poured into the ocean or a small lake. Similarly, the life cycle releases of a floor covering may be irrelevant but nonetheless lead to high indoor concentrations. However, such characteristics are normally ignored in LCA.

ANEC has strong doubts about the underlying principles of LCIA as outlined in the part of the ILCD Handbook addressing LCIA models and indicators[10]. In particular, chapter 4.3.1 explaining the differences between regulatory and LCA approaches raises concern: "*The scope and methodology of an LCA differs from that of many approaches adopted for toxicological assessments in a regulatory context. Regulatory assessments of chemical emissions usually have the objective of evaluating whether there will be an unacceptable risk of a toxicological effect to an individual or subpopulation*". This is done by comparing the actual exposure of a population with what is considered to be an acceptable threshold. By contrast, LCA toxicity assessment relies on a different approach: "*Models and factors for toxicological effects in LCA must be based on the relative risk and associated consequences of chemicals that are released into the environment*".

However, this raises the question how a "relative risk" can be determined when essential aspects to adequately determine the risk are completely ignored (exposure relative to threshold). This is all the more worrying as background concentrations are not taken into account: "*However, in LCIA all emissions not related to the evaluated product are deliberately excluded from the assessment, e.g. emission of the same chemicals from other products or from sites unrelated to the product*". This means that 2 substances of similar toxicity (and other factors), released from different product systems, would be considered equivalent even if for one substance – as in case of cadmium – the (overall) exposure of a significant proportion of the population is around or even above the acceptable levels, and any additional exposure must be avoided whilst the exposure to the other substance is far below any threshold. This is dangerous nonsense!

"*Contributions of emissions to short-term/acute and local scale effects are presently not addressed in the recommendation. This includes those associated with indoor exposures, direct exposure to products during their use stage, and to exposures in the work place. The focus here is on the contribution of emissions to the risk of toxicological impacts and associated consequences considering the entire human population and dispersed emissions*". However, direct exposure to chemicals during the production or consumption stage are of highest importance in consumer/worker protection (e.g. release of plasticizers from toys, bisphenol A from baby bottles, chromium VI from leather products, additives from food packaging, etc.). Few people are probably aware of these serious limitations of LCA toxicity assessment as stated above. It seems a bold statement to say that this fundamentally flawed

[10] "Framework and requirements for LCIA models and indicators", JRC, First edition, 2010

approach accounts "for the full extent of the likelihood of an effect … and differences in severity" and the "comparative risk of a chemical" is estimated "considering the entire human population and dispersed emissions". If the method cannot assess effects for individuals or sub populations, it cannot assess effects for the whole world population either.

LCA toxicity impact assessment uses 10% or 50% effect levels from laboratory experiments, rather than "No (Adverse) Effect Levels (NO(A)ELS)" and safety factors which the regulatory approach uses. "*Regulatory-based measures do not necessarily provide a consistent risk basis for comparison, as they were often not developed for use in such a comparative context or to facilitate low dose-response extrapolation*". It is difficult to understand why a 10% or 50% dose-response benchmark from animal experiments should be more appropriate for comparisons and low dose-response extrapolations, given that the slopes of the dose-response curves may be quite different and says little about the no effect level, let alone anything about levels of concern for humans. What is essential is that adverse effects for humans are avoided, i.e. to make <u>relevant</u> comparisons, rather than making comparisons for the sake of comparisons. Relative comparisons are pointless when no statement can be made about the possible real damage. Product A may seem preferable to product B but a risk assessment may conclude the opposite. Similarly, both products may be of concern or none. "*Other differences in data use in LCA and regulatory/based risk assessments include … the consideration of safety factors only as part of the uncertainty assessment, and not as an integral part of the toxicological effects data*". The purpose of applying safety factors in chemical risk assessment (e.g. to derive TDI values from NOAELs) is to extrapolate from animal data to humans - the actual goal of protection measures. This is not just a mere "uncertainty assessment" in terms of possible variation of results. It is an effort to ensure that safe doses for humans are derived. Apparently this is something which is out of the scope of LCA toxicologists.

The latest development in this area is the so-called USEtox model[11] claimed to be based on a "scientific consensus". It is quite interesting to see how much the scientific nature of the undertaking is stressed – something which is rather uncommon in science. It relies on the (questionable) principles discussed above. For the authors, the model "*provides a parsimonious and transparent tool for human health and ecosystem CF estimates*". The calculated characterisation factors are based on fate, exposure and effect modelling. The model and CFs are the result of a collaborative effort of comparing and partly aligning different existing toxicity models. In this process the inter-model variation was significantly reduced: "*Through this process, we were able to reduce intermodel variation from an initial range of up to 13 orders of magnitude down to no more than two orders of*

[11] USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment, Rosenbaum, R.K. et al., Int J Life Cycle Assess (2008), 13:532-546 https://www.researchgate.net/publication/49458514_USEtox_-_The_UNEP-SETAC_toxicity_model_recommended_characterisation_factors_for_human_toxicity_and_freshwater_ecotoxicity_in_Life_Cycle_Impact_Assessment

magnitude for any substance". Thereby, the "*precision of the new characterisation factors (CFs) is within a factor of 100–1,000 for human health*". This means that even according to the authors the uncertainty is rather high. However, these figures are based on comparisons of models. This raises the question how these models are related to the real world. Many assumptions behind the models seem to be far from reality. It is assumed that the whole life cycle release of a substance is evenly distributed in the compartments urban air, rural air, agricultural soil, industrial soil, freshwater and coastal marine water. In real life, the concentrations will be often highest close to the emission point (e.g. car emissions in busy roads). A validation of the models using measurement data would be probably extremely difficult.

The ILCD Handbook states: "*Due to the large number of potential endpoints that involve various mechanisms, there is no true midpoint for toxicological effects where comparisons can be made on a purely natural science basis".* It would be more correct to say that there is NO such basis. *"The midpoint indicator is therefore based on the likelihood of an effect associated with an emission of a quantity of a chemical*". In fact, the midpoint indicator for toxicity does therefore not exist – neither in form of a toxicity score nor in terms of sub scores (cancer/non-cancer effects/respiratory diseases/ impact of ionizing radiation).

But endpoint indicators applied on top of the fragile LCA toxicity house-of-cards raise even more concerns. To assess the actual damage on humans, the so-called DALY-concept (Disability Adjusted Life Years) is used which "*combines information on quality of life and life expectancy in one indicator, deriving the (potential) number of healthy life years lost due to premature mortality or morbidity*". The authors of the source mentioned above admit themselves (chapter 3.1.3) that "*the actual calculation depends on a number of uncertainties, choices and assumptions*". This includes, for example, weighting of disabilities, dependency on location and time (quality of health systems), lack of information on critical effects of chemicals and missing DALYs for health effects.

But there are more severe concerns: how can an exposure to a chemical be translated into a probability to get a certain sickness? This would be quite difficult for a single substance with known (overall) exposure patterns and relevant human thresholds unless good epidemiological data are available (e.g. for cigarette smoking). Owens put it like this: "*For most chemicals, there is no apparent means to convert the critical effects in animal studies into times of human deaths or length and severity of disability necessary for a DALYs approach*". In addition, it appears questionable to assign a proportion of this to a specific product or a functional unit and this for a whole group of chemicals.

The DALY concept itself is debatable from an ethical perspective as it aggregates death and disability. Are 10 years with a 10% disability equivalent to 1 year of premature death?

In an article[12] on carbon footprinting, Mathias Finkbeiner, Chairman of the ISO LCA committee (ISO TC 207 SC5), pointed to the many unsolved issues regarding LCA/CFP inventories: "*Nowadays, we may pretend to know how many life-years humankind is losing because of malaria resulting from a certain amount of GHG emissions, but there are still scientifically unresolved issues, how much GHG emissions we can actually attribute to a certain product. It is a bit like flying to Mars before having invented the wheel (at least one that is more or less circular in shape)*". As the old proverb goes: cobbler, stick to your trade! It is better to produce a good pair of shoes than a bad pair of wings….

Conclusion of ANEC: an aggregated life cycle release amount is a questionable basis for a toxicological impact assessment. These impacts are not "potential" – they are mostly "fictional". ANEC considers that a reasonable method to assess chemicals throughout the life cycle must be based on the same principles as regulatory approaches to ensure that chemicals do not surpass concern levels in any life cycle stage. This is to be accomplished by (simplified) risk assessment techniques. Models such as USEtox can (at best) be used as a screening tool to identify substances of concern which are then investigated in more detail. But there may be easier and more straightforward ways of doing this.

ANEC is even more concerned about approaches resulting in single scores based on aggregation of different impact category scores (=adding apples and elephants). ANEC finds it inappropriate to use such approaches for priority setting in EU policy (e.g. for the ecolabel). It is appalling that that even these approaches appear to be considered "scientific" by some.

1.5 ISO standards for LCA, EPD, PCF

From the above follows that existing standards for LCA (ISO 14040 series) and EPD (ISO 14025) should be revised to remove the inherent bias towards aggregable and quantifiable life cycle indicators, and to strengthen the weight of other instruments such as human and environmental risk assessment, i.e. so-called "additional environmental information". Clear-cut rules must be provided as to which tool is used for which purpose. One option may be to limit LCA methodology to mass and energy balances including greenhouse gases. Alternatively, additional standards could be prepared combining various instruments and traditional LCA for a comprehensive environmental assessment resulting in an "Environmental Data Sheet" (see above).

However, it is important to note that the process oriented LCA standards – which were essentially developed in the 1990s and are outdated today - also suffer from a lack of detail regarding conventional LCA methodology giving a lot of freedom for the LCA practitioners. For instance, it is interesting that the complex and difficult undertaking of establishing a functional unit is addressed in ISO 14044 with a mere two (not necessarily elucidating) requirements: "The functional unit shall be

[12] Finkbeiner, M. Carbon footprinting—opportunities and threats. *Int J Life Cycle Assess* 14, 91–94 (2009). https://doi.org/10.1007/s11367-009-0064-x

consistent with the goal and scope of the study" and "Therefore the functional unit shall be clearly defined and measurable". Such standards cannot provide for comparability – not even for simple mass or energy balances. Therefore it may be useful to specify Product Category Rules (PCRs) for a limited number of priority products. Such development must neither be shifted to standards bodies nor to industry organisations – but must be developed with balanced stakeholder participation under the lead of the Commission. However, there must be first an in-depth discussion about what the expectation of stakeholders regarding the quality and level of detail of such PCRs is. This is still an open question. Existing published PCRs are of rather modest quality. CEN TC 350, for example, developed a standard (EN 15804) for the whole range of construction products – a contradiction in itself.

ANEC was disappointed by the recent endeavour by ISO to develop a standard on the carbon footprint of products (ISO 14067), and has repeatedly expressed its disapproval. CFP specific requirements are vague and allow many choices, e.g. as regards scenarios for use or end of life stages, land use change, soil carbon change, carbon storage in products, non-CO_2 emissions and removals or aircraft emissions can be dealt with in quite different ways. Indirect land use change does not need to be taken into account at all. Credibility is not ensured as third party verification is not a must (a step backward compared to ISO 14025 which requires third party verification in a B-to-C context). PCRs are mandatory only in few communication options. And PCRs could be used without a programme, which is in contradiction to the underlying ISO 14025. Moreover, meaningful consumer information using appropriate colour/letter codes and rating scales is not required (which could be used in some cases subject to specific conditions, e.g. for CO_2 labelling of cars).

2. Other LCA limitation reviews

2.1 General limitations

A literature review entitled "A survey of unresolved problems in life cycle assessment"[13] was published in 2008 highlighting 15 major problem areas identified by the LCA community itself structured by LCA phases. Part 1 addresses goal and scope and inventory analysis, part 2 impact assessment and interpretation.

The result of the first part is summarized as follows: "*Multiple problems occur in each of LCA's four phases and reduce the accuracy of this tool. Considering problem severity and the adequacy of current solutions, six of the 15 discussed problems are of paramount importance. In LCA's first two phases, functional unit definition, boundary selection, and allocation are critical problems requiring particular attention*".

[13] "A survey of unresolved problems in life cycle assessment. Part 1: goal and scope and inventory analysis", John Reap et al., Int J Life Cycle Assess (2008) 13:290–300
"A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation", John Reap et al., Int J Life Cycle Assess (2008) 13:374–388

As an example the authors rightly emphasize the key importance of adequately selecting a functional unit (in particular for product comparisons) and the difficulties this poses.

The authors point to "*multiple potential sources of error*" which "*can stem from inaccurate reflection of the product system reality when identifying and prioritizing functions, defining the functional unit and defining the reference flow*". But is this just a question of error and truth? Is there the one and only correct way of doing things? Using the example the authors themselves give – getting the news from different media such as a newspaper, TV, and the Internet – the question is what a suitable comparison basis is. Using only the impacts relating to the production of the pages of paper containing the news story for comparison seems questionable as a reader has to buy a complete newspaper including all pages it consists of. By contrast, one could argue that a reader will read more articles than just the one (and must accept all the useless advertising too!). There are probably many plausible ways to handle the issue – and they will give quite different results.

Products have often multiple functions (different types of cars, smartphones) or functions which are difficult to measure (e.g. aesthetics) which make comparisons and assignments of reference flows difficult. Is drying hands using paper/cotton towels equivalent to using a fan which may be less convenient (time-consuming, sometimes impractical position)?

A critical step is the definition of reference flows to functional units. This is among other determined by assumptions related to the product life time and product use. The issue is only briefly addressed in the paper. In fact, it is one of the most important contributors to uncertainty. In particular, for long-lived products, it is difficult to anticipate a service life time. The life time of a house can only be determined once it is demolished which may be as low as 40 years from now or more than 120 years. A large proportion of the inputs and outputs of buildings will occur a long time in future and are, by definition, just more or less guesses. T-shirts differ quite significantly with respect to durability and may be kept for short or prolonged periods (which are not known) – not least influenced by fashion habits. Washing and drying of such garments involves quite a few variables (efficiency of washing machine, wash loads, temperature programmes, wash cycles, drying in air or tumble drier). All related assumptions may be close to or far from reality.

Similarly, the definition of system boundaries can be made in quite different ways. It is difficult to justify any cut-off criterion because, by definition, the environmental impacts outside the boundary remain unknown. Process-based LCAs have been shown to omit significant proportions of the impacts. Hence, Input-Output (IO) LCA was employed to overcome these difficulties. However, also IO-LCAs have significant methodological drawbacks as shown in the paper.

LCA studies rely on scenarios (e.g. for end-of-life treatment) which have a significant influence on the results. "*The inherent difficulty with any formal scenario analysis framework is that of trying to predict with confidence the future*". Hence, the authors recommend that "*any LCA practitioner should pause and carefully reflect on the scenarios selected*". But is it enough to model such scenarios carefully? Maybe it is a better choice to avoid such foreseeable failures altogether or, at least, make clear that the results are just guesses.

"*The allocation problem has the distinction of being called one of the most controversial issues of LCA*". Allocation is defined in ISO 14044 as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems". This can be done in different ways – again there is not the "correct" or "incorrect" allocation rule which the authors seem to believe (although a conclusion of a one of the reviewed studies is provided that "*no single method provides a general solution*").

Impact assessment is – according to the authors – "*the most challenging of LCA's four phases*". The problems are "*associated with impact category selection, spatial variation, local uniqueness, environmental dynamics, and decision time horizons*". Lack of standardisation, significant data gaps, lack of consensus, diverging results depending on the method used are some of the problems relating to the impact category selection. "*Unlike global impacts such as stratospheric ozone depletion and global warming, those affecting local, regional and continental scales require spatial information in order to accurately associate sources with receiving environments of variable sensitivity*". The sensitivity varies – "*each local environment is uniquely sensitive to the stresses placed upon it by a particular product system's life cycle*". Similarly, "*Temporal factors such as timing of emissions, rate of release, and time-dependent environmental processes affect the impact of pollution*". However, spatial, temporal and sensitivity variation is typically ignored in LCA studies. This raises the question whether LCA is the instrument to address such issues adequately.

Key issues in the interpretation phase are related to weighting, valuation and uncertainty management. Weighting of different impacts is a precondition to derive an overall judgement of overall superiority – but this is a value choice which poses a number of challenges.

Main types of uncertainty include "*badly measured data ('data inaccuracy'), data gaps, unrepresentative (proxy) data, model uncertainty, and uncertainty about LCA methodological choices*". Even though methods for uncertainty and sensitivity analysis are available LCA results may deliver shaky results: "*Inaccuracies in other phases and variability inherent in modelled systems result in a high degree of aggregated uncertainty by the time one reaches an LCA's interpretation phase. Making meaningful decisions under this potentially severe level of uncertainty is challenging*". There is nothing to add.

2.2 LCA and Risk Assessment

Even the LCA family admits that LCA and risk assessment (RA) are complementary tools. The ILCD Handbook says: "*Thus, site specific regulatory assessments, chemical related regulatory assessments and toxicity aspects in LCIA are to be seen* *complementary in their nature*". In fact, LCA and RA are often combined in various ways as pointed out in a Swedish study[14]: "*When comparing risk assessment and LCA there are five different, alternative solutions or approaches; they could be seen as completely separated, overlapped i.e. there is an intersection between them, RA could be a subset of LCA, LCA could be a subset of RA and finally they could be seen as complementary tools where they both are needed to get the whole picture…..*". However, it remains unclear how the interaction should look like and which tool is responsible for what. The authors of the Swedish study conclude: "*One straightforward view is to regard data and knowledge from risk assessment as input to model the impact assessment of chemical substances of LCA. Another straightforward view is to regard LCA as a strategic tool to prioritise the data to acquire and the risk assessments to perform. Based on the result of an LCA the prioritising may, for example be based on the location of the emissions, the functionality of the process, product or emissions or the amount of emissions etc. In this way prioritising and relevance may be an LCA input into risk assessment, and to thereby provide environmentally relevant cost efficiency to the prioritisation of risk assessments*". The second option raises the question whether complex and contestable impact assessment models are required to identify substances for a more in-depth assessment following the risk assessment approach. There may be simpler and more cost efficient ways to identify priority substances such as available chemical ranking and scoring systems which may have to be adapted to the information requirements of REACH, the more so as some of the most relevant exposure paths such as direct exposure during production and consumption are not covered by LCA anyway. Along the same lines other human health and environmental risks can be covered.

3. Case examples – the real world of LCA

The following examples are intended to illustrate typical controversies following the publication of a LCA study highlighting some aspects of the debate. It is beyond the scope of this paper to make comprehensive assessments of the studies quoted. However, one could raise many more issues on the various aspects of the methods employed.

3.1 Packaging

Packaging was one of the first areas where LCA was applied in order to support environmental decision making (e.g. in Germany from the early 1990s). The subject has triggered many disputes and still is a controversial issue, as the following extract from a press release of the German environmental organisation DUH (Deutsche Umwelthilfe) on the occasion of the 3rd ReUse Conference in Brussels in October 2010 shows[15]:

[14] "Relationships between Life Cycle Assessment and Risk Assessment - Potentials and Obstacles. Swedish EPA, June 2004 https://5dok.org/document/oy8xr5qr-relationships-between-life-cycle-assessment-and-risk-assessment.html

[15] "Reusable Packaging in Europe: Waste Reduction and Resource Efficiency", DUH, 2010-10-07 http://www.duh.de/pressemitteilung.html?&tx_ttnews[tt_news]=2406

"*In order to reach the goal of market domination the one-way industry needs arguments. So it buys them. Namely in the form of life cycle assessments based on extreme one-sided and unrealistic figures leading to misleadingly positive results for PET one-way bottles and cans*", Jürgen Resch, DUH Managing Director, explained. The plastics industry as well as the Beverage Can Makers Europe (BCME) has commissioned LCAs to the German IFEU Institute, feeding the contractor with their own figures, requests and assumptions. "*An LCA is like a black box: if you enter false and invalid data and misleading assumptions into the calculations, you end up with the wrong results. And this is what happened with the LCAs recently published by the plastics and beverage can industry*", so Resch. Based on the LCA on beverage cans for beer, BCME lobby communicated broadly to the general public that the beverage would be on par with the environmentally-friendly reusable glass bottle.

Similarly, the German IFEU Institute prepared a comparative study on certain beverage containers in Austria, published in early 2011. It concluded that one-way PET bottles are equivalent to reusable glass bottles. The study came under attack by various organisations which questioned a number of methodological assumptions and conclusions of the study – for instance the assumed transport scenarios or low number of refills of reusable bottles (just 30 rather than the assumed correct number of 40) was challenged[16]. At least there was agreement that the most preferable solution would be reusable PET bottles.

A statement[17] from the brewery industry (Derek McKernan, head of packaging for group technical at SABMiller): "*Unfortunately there are various parties who use their studies of packaging to simply make their own materials look better, by excluding bits of the supply chain. This makes them quite difficult to rely on. They all depend on the various assumptions that are made, which is why we do our own analysis*". "*It all depends on who initiates the reports,*" says Roland Folz, head of brewing & beverage science and applications at VLB Berlin. "*It seems that today you can't introduce any new packaging without an LCA supporting it so they are bound to be used as marketing tools.*"

3.2 Nappies

Many studies have been performed to compare disposable and reusable diapers with quite different conclusions – one of the classical LCA battles. For instance, a study[18] commissioned by the National Association of Diaper Services (NADS) published in early 1991 concluded: "*Considering the overall environmental burdens, and most notably the higher volumes of solid waste produced and energy and raw materials consumed by single-use diapers, reusable diapers are determined to be*

[16] PET-Mehrweg wäre die beste Flasche, derStandard.at, 2011-02-18 http://derstandard.at/1297818327464/Oekobilanz-Studie-PET-Mehrweg-waere-die-beste-Flasche
[17] Packaging's green debate, Brewer's Guardian, September/October 2011 http://www.petengineering.com/sites/default/files/pdf/Brewers%20September_October%202011%20p30-34.pdf
[18] Lehrburger/Mullen/Jones, "Diapers: Environmental Impacts and Lifecycle Analysis," January 1991

superior from an environmental perspective." However, studies commissioned by the disposable diaper industry – not surprisingly - came to opposite conclusions. For instance, a study[19] commissioned by Procter & Gamble to Arthur D. Little in 1990 found significant advantages of disposable diapers compared to cloth diapers with respect to energy consumption, water use, air pollution and water emissions.

A LCA study[20] of "Disposable and Reusable Nappies in the UK" was published by DEFRA in May 2005. The (solomon-like) conclusion provided within this study was: "*There is no significant difference between any of the environmental impacts of the disposable, home use reusable and commercial laundry systems that were assessed. None of the systems studied is more or less environmentally preferable*". However, also this conclusion was contested by various parties challenging various assumptions made, in particular, relating to the use scenarios. An update of the study[21] was published in 2008. Although the main conclusion was the same, the report better highlighted that the results for reusable nappies strongly depend on the assumptions concerning washing and drying: "*Combining three of the beneficial scenarios (washing nappies in a fuller load, outdoor line drying all of the time, and reusing nappies on a second child) would lower the global warming impact by 40 per cent from the baseline scenario*" whilst, by contrast "*the study indicated that if a consumer tumble-dried all their reusable nappies, it would produce a global warming impact 43 per cent higher than the baseline scenario*". So, it depends…as usual.

The US Real Diaper Association (RDA) was not convinced that both kinds of diapers are equivalent and published in response to the UK studies a "Flawed Impact Studies Review"[22] which stated that "the data and assumptions are flawed", i.e. were at the disadvantage of reusable diapers (e.g. no inclusion of commonly used prefolded diapers with lower impacts, not representative production data). Their conclusion regarding LCA: "*When LCA is used for comparison, there are too many variables to result in an accurate comparison. The UK studies tried to control for these variables, but those controls don't resolve the issues of what impacts count. These are the foundational assumptions inevitable in any study. Compare two such different groups of products, and the assumptions will determine outcomes*".

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19 Arthur D. Little, Inc., "Disposable Versus Reusable Diapers: Health, Environmental and Economic Comparisons", March 1990

20 "Disposable and Reusable Nappies in the UK", Environment Agency, May 2005 https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/290683/scho0505bjcw-e-e.pdf

21 "An updated lifecycle assessment study for disposable and reusable nappies", Environment Agency, October 2008 https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/291130/scho0808boir-e-e.pdf

22 "Flawed Impact Studies Review", Real Diaper Association, accessed 2012-03-04 http://www.ecobabysteps.com/2010/04/12/lower-environmental-impact-of-cloth-diapers/

3.3 Hand drying

The German Öko-Institut published a study[23] which compared a continuous cotton roll system and a paper towel system (made from virgin luxury paper and from 50 % recycled fibres), commissioned by the European Textile Service Association (E.T.S.A.) in 2006. The functional unit of the study was specified with 10,000 hand-dryings: 10,000 pulls of the cotton roll system were compared with 20,000 paper towels. It was assumed that the cotton roll would be washed 100 times. Although there were some caveats considering different use scenarios (different life times for the cotton rolls, using more or less cotton or paper towels), the main conclusion was: "*As conclusion and seeing that the use behaviour of the washroom clients will influence the environmental assessment of both systems significantly, it can be stated that the cotton roll system for standard use causes less environmental impacts than the paper towel system*". The commissioning organisation was pleased and published a statement[24] claiming that "*scientific comparison leaves no doubt*" that "*cotton towels outperform the paper alternative*".

Not surprisingly the European Tissue Symposium (ETS), a Brussels based trade association of the European tissue paper industry, was not pleased with the result and prepared the following counter-statement[25]:

"*ETS has also analysed the Öko report and has come to the conclusion that cotton roll towels do not outperform paper towels in environmental aspects. The arguments can be summarised as follows.*

1. The weight parameter that is used in the research for paper towel is about 4 grams. Paper towels sold in Europe weight approx. 2-3 grams.

2. Another parameter that highly influences the outcome of the research is the assumption that for hand drying with cotton roll towels only 1 pull per hand drying is used whereas 2 paper towels are taken per hand drying. Observations show that on average 1,5- 2 pulls per hand drying is more realistic for a cotton roll.

3. The report has calculated the outcome for 100% virgin towels and 50% virgin/50% recycled. In reality a big part of the towels in Europe are made out of 100% recycled fibres.

4. The exclusion of key environmental impacts associated with fertilizer and pesticide runoff and volatilization in cotton culture can highly influence the environmental impact of cotton rolls".

Later, ETS also commissioned a study[26] to the environmental consultancy PE International to review the study by Ökoinstitut. Their conclusion: "*The conclusion*

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[23] "Life Cycle Analysis of hand-drying systems - A comparison of cotton towels and paper towels", Ökoinstitut, June 2006 https://www.oeko.de/en/publications/p-details/life-cycle-analysis-of-hand-drying-systems-supplement =

[24] "Continuous cotton roll towels - Top Environmental Performance", July 2006

[25] "Tissue and other hand drying systems. Their environmental impact". ETS, April 2008 http://www.europeantissue.com/pdfs/080503-etsEnvironmental%20Impact%20Drying%20Systems%20-%20042008.pdf

of the study was that paper towels are the less preferable option for hand drying in washrooms. This conclusion however, is based on a number of assumptions that do not represent state of the art knowledge or can be challenged as an oversimplification. Once corrected the study conclusions are reversed".

3.4 Biofuels

One of the most controversial environmental discussions of the last years was related to the political support of biofuels inspired by a number of LCA studies showing (or not) benefits in terms of greenhouse gas reductions. The EU adopted, for instance, the Renewable Energy Directive (RED)[27] in 2009 which provides that 20% of all energy used in the EU has to come from so-called "renewable sources", including biomass, bioliquids and biogas, by 2020 (different targets for different Member States). It also stipulates a 10% share of renewable energy in the transport sector to be complied with by each Member State by 2020. A large proportion of this share is believed to be accomplished through biofuels. Greenhouse gas emission savings from the use of biofuels and bioliquids must be at least 35% compared with fossil fuels. From 2017, this value is increased to 50%. From 2018, a value of 60% applies, but only if production started in 2017. To calculate the savings, different options exist including default greenhouse emission saving values for various biofuels given in Annex V (e.g. for rape seed biodiesel 38%) which may be used subject to conditions not elaborated here. Most of the listed biofuels (but not necessarily all related production pathways) have default values better than required. Producers may also calculate values using the indicated methodology to demonstrate compliance. The EU Fuel Quality Directive follows a similar approach.

It turned out that this policy is built on sand. The environmental savings (other aspects are not discussed in this paper) have been challenged by various institutions. The OECD, for example, published a paper in 2007 tellingly entitled "Biofuels: is the cure worse than the disease?"[28] which concluded: "*The conclusion must be that the potential of the current technologies of choice — ethanol and biodiesel — to deliver a major contribution to the energy demands of the transport sector without compromising food prices and the environment is very limited*".

Also a joint publication[29] of several environmental NGOs warned that the EU policy will cause more harm than good: "*One of the most important reasons for this is the failure to account for the environmental impact of indirect land use change (ILUC). When agricultural land is converted for biofuel production, land elsewhere will be converted for agriculture, releasing lots of CO_2 emissions, hence the term 'indirect'*

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[26] "Critique of the LCA Study 'Life Cycle Analysis of hand-drying systems' by U. Eberle and M.Möller Öko-Institut E.V., 2006", PE International, April 2010

[27] Directive 2009/28/EC on the promotion of the use of energy from renewable sources

[28] "Biofuels: is the cure worse than the disease?", OECD, September 2007 http://www.oecd.org/dataoecd/15/46/39348696.pdf

[29] "Biofuels. Handle with care", Various ENGOS, November 2009 https://www.transportenvironment.org/wp-content/uploads/2021/07/2009%2011_biofuels_handle_with_care.pdf

land use change. Assessing the impact of ILUC and incorporating it in biofuels policy is critically important to ensuring biofuels really do reduce carbon emissions and do not indirectly increase them".

The effects of LUC were estimated by various studies. For example, the International Food Policy Institute (IFPRI)[30] published a figure of 38-40 g CO_2 equivalent per MJ and stated: "Overall, land use emissions for the entire EU biofuels additional mandate eliminate more than two-thirds of the direct emission savings". Bad enough! However, another study by the Institute for European Environmental Policy (IEEP) found that RED leads to an increase rather than a decrease of greenhouse gas emissions and "*would lead to between 80.5% and 167% more GHG emissions than meeting the same need through fossil fuel use*".

The European Environment Agency (EEA) Scientific Committee expressed concern in an opinion[31] that EU policies in support of renewable energy derived from plant biomass "*inaccurately assess the greenhouse gas consequences of different forms of bioenergy*" and identified a "*serious accounting error*" which "*could have serious adverse consequences on a range of environmental concerns*". In particular, the omission of ILUC is deplored.

There are more issues to be solved when applying LCA methodology to crop/plant-based biofuels. In a paper[32] entitled "Grand challenges for Life-Cycle Assessment of biofuels" the authors sound a note of caution with respect to the uncertainty and variability of LCA results: "*Addressing uncertainty is among the greatest of the grand challenges, not only for biofuels LCA, but for other LCA efforts*". This uncertainty – some of which is irreducible - must be explicitly taken into account in policy making. Their advice: "*Decision makers who work in real time and often cannot wait for precise results must recognize that LCA can provide valuable insight but it is not necessarily a "truthgenerating machine". Effective LCA can guide and inform decisions, but it cannot replace the wisdom, balance, and responsibility exhibited by effective decision-makers*".

In his analysis[33] John M. DeCicco arrives at a more radical conclusion: "*While it may be discomfiting to some readers, the conclusion is that LCA is inappropriate for specifying regulations. Although LCA may be a useful research tool and can helpfully inform policy discussions, its literal application for policy specification is a mistake. Disputes over LCA regulatory outcomes are unproductive and ultimately unresolvable*". Following the principle "what gets measured, gets managed", he

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[30] Assessing the Land Use Change Consequences of European Biofuel Policies, IFPRI, October 2011 https://www.ifpri.org/publication/assessing-land-use-change-consequences-european-biofuel-policies

[31] Opinion of the EEA Scientific Committee on Greenhouse Gas Accounting in Relation to Bioenergy EEA, September 2011 http://www.eea.europa.eu/about-us/governance/scientific-committee/sc-opinions/opinions-on-scientific-issues/sc-opinion-on-greenhouse-gas

[32] Grand Challenges for Life-Cycle Assessment of Biofuels, T. E. McKone et al., January 2011 http://pubs.acs.org/doi/abs/10.1021/es103579c

[33] Biofuels and Carbon Management, John M. DeCicco, July 2011 http://deepblue.lib.umich.edu/bitstream/2027.42/86104/1/Biofuels%20and%20Carbon%20Management%20FINAL%20for%20CC%202011.pdf

proposes as an alternative a method using annual basis carbon (ABC) accounting to track the stocks and flows of carbon and other relevant GHGs throughout fuel supply chains, focusing on real emissions of fuel and feedstock production facilities without any biogenic carbon bonus.

The lesson to be learnt from this is that, first, biofuels should never have received regulatory support in the EU and, second, LCA indicator results are not a solid base for decision making. It would have been much more sensible to support CO_2 reduction by enhancing energy efficiency in the transport sector (e.g. by more demanding CO_2 emission limits for cars) and by traffic reduction measures.

4. Does standardisation help?

4.1 Enhancing precision

Standardisation may reduce the variability of LCA results to some extent – however, this does not necessarily mean that the results become more reliable in the context of setting relevant environmental indicators or regulatory requirements. Normative provisions in this regard may be simply wrong or not applicable for a specific case or impossible to back with data.

Using one of the examples above – number of trips of reusable beverage bottles – may illustrate the dilemma. One could, of course, define this number in a PCR (Product Category Rules) and stipulate that the number of trips is assumed to be 35. Then all LCA practitioners would use this figure and – seemingly - this would reduce the variability of results and, thereby, increase the validity of the results. However, this would mean to punish manufacturers which can achieve more trips (e.g. more than 40) and reward those which are below the agreed number (e.g. less than 30). Understandably, this would not be acceptable in particular by those with a high number of trips and the results would be challenged again. The only alternative option would be to agree in a multistakeholder process on a number of trips (or any other parameter) for any of the specific systems or individual manufacturers involved in the study. One may call this a study specific PCR which would have to be prepared in advance of any LCA study in a public policy context - a quite laborious undertaking.

In the above example, at least the number of trips is in principle accessible (after some time of operation of the system). This does not need to be so. For instance, in many cases the relevant service life time is not known. As an example, T-shirts may differ strongly in terms of quality and durability, and may be kept for short or prolonged periods of times and, consequently, laundered a different number of times using washing machines of quite different efficiencies, wash loads, temperature programmes etc. Any assumption could be hardly backed by data (at best, one might collect some average data with a lot of effort) and – as in the case above – would not be appropriate for a specific product. For long-lived products – such as a house - the service life time is only known when the building is demolished (which may be less than 40 years or more than 120 years from now).

This makes it virtually impossible to determine any service life time ex ante – not even with a very high effort.

Results can differ widely depending on the chosen scenarios for use, transport, waste management etc. There is not necessarily a right or wrong. In case of the nappies example above, significant improvements can be achieved when the appropriate laundering techniques are used. The only conclusion from this is that – as a matter of principle – ranges should always be given and best/worst cases calculated. But still one would have to agree on what these scenarios are.

Scenarios can be moving targets. One example for this is the electricity mix and related emission factors. Although the electricity consumption of a washing machine, a refrigerator or a computer can be measured, it is not straightforward to calculate resulting CO_2 emissions during operation. ISO 14067 calls for the use of national grid mixes. This would mean to calculate the CO_2 emissions for any country separately. However, any consumer can choose and change the energy supplier with a few mouse clicks. The emission factors differ substantially – from almost 100% renewable share (with low CO_2) to a huge proportion of electricity from fossil-fuel power plants. Again, any predefined mix and emission factor would be arbitrary - only the indications of all possibilities are probably useful. Any CO_2 figures complementing the energy label scheme would, therefore, be rather difficult to implement. It would be either complex and confusing (many numbers) or simplistic (just one number based on a European mix) which would not give any relevant information in addition to the electricity consumption based rating.

Conclusion: the above examples show clearly that strategies to enhance the precision of LCA are limited. It would mean a tremendous effort to find agreement of all parties involved, affected and interested to stipulate the many choices to be made in the conduct of a LCA in advance, and still would deliver numbers with a lot of uncertainty. The uncertainty may still be of the order of magnitude of the performance difference between products to be compared, which would make the identification of superior products virtually impossible. For some products, this may be a suitable way forward (where differences are very big) but clearly this cannot be a general approach. The development of appropriate PCRs would take many years if a high quality is the aim. It can be doubted that controversies, such as the ones mentioned above, would be avoided.

4.2 Other aspects

As pointed out above, limitations of LCA with respect to human health and environmental risks are of principle nature. They can be overcome only by using instruments that are fit for the purpose. But this requires broad discussion involving all stakeholders concerned and a political decision before any standardisation can be initiated.

5. Corporate indicators

When the Commission presented a proposal for a revised EMAS scheme in 2008 (EMAS III), the obligation to make use of general "core indicators" both in the environmental statement and the environmental performance report was introduced. These indicators covered energy, materials, water, waste, biodiversity, and emissions. In addition to the total amounts (e.g. of water use per company and year), normalised figures relating to economic output - total annual gross value added (for big industry) and total annual turnover or number of employees (for small organisations) - was required.

In a joint position paper[34], ANEC, ECOS and EEB rejected this approach: "*However, generic indicators such as total energy consumption are normally not meaningful as they do not allow for reasonable comparisons between organisations. Even if such data are related to the physical or monetary output, including the value added or number of employees, they say very little, and could be equated with the results of comparing apples and pears. A prerequisite for serious assessments of performance and benchmarking is to compare comparable activities or processes*". One can, for instance, compare the energy intensities of the production of 1t of cement and related pollutant emissions, but not the energy consumption of different construction products manufacturers of different sizes with quite different product portfolios (let alone the energy consumption of any other producers or service providers). Indications of total tonnages are also promoted by the Global Reporting Initiative (GRI) guidelines. Maybe these guidelines are so popular among industry just because they do not permit performance comparisons and benchmarking. However, they serve as a good decoration in CSR or sustainability reports, and give these reports a touch of seriousness and objectivity. Apart from that, these numbers – which nobody can verify anyway - are pointless.

Instead ANEC, ECOS and EEB proposed: "*Hence, ANEC, ECOS and EEB consider the proposed general indicators to be of little use. Instead, we believe the focus should be on the development of a limited number of relevant and comparable (sub) sector-specific indicators*". Indeed, one of the few positive changes in EMAS III was the introduction of sector reference documents. The current pilot projects directed by JRC IPTS are also based on the philosophy of comparable process based indicators and identification of best practice. In fact, these documents are an equivalent to BREF documents for non-industrial sectors. ANEC is quite pleased with the progress in this area.

ISO 14031 on environmental performance evaluation[35] is in the process of being revised. To our great delight a clause (4.2.2.5) on "Selecting sector-specific operational performance indicators for comparison" has been inserted (with significant input from ANEC). This document makes also clear that comparisons of

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[34] Joint ANEC/ECOS/EEB position on "Commission proposal for a revised EMAS (EMAS III)", October 2008 http://www.anec.eu/attachments/ANEC-ENV-2008-G-037final.pdf

[35] ISO/DIS 14031 Environmental management — Environmental performance evaluation — Guidelines, January 2012

operational performance indicators (OPIs), based on quantities per unit of time relating to an entire organization or to its sub-units, are normally not possible.

Sometimes, the use of such indicators is justified by arguing that an organisation can monitor performance changes over time. But according to the draft ISO standard, this is possible only to a limited extent: "*Similarly, while monitoring OPIs over a period of time can identify performance trends for an organization, increases or decreases of environmental burdens are not necessarily related to performance changes alone, but may be due to other reasons such as organizational expansion/reduction of production or outsourcing/relocations of certain activities. Hence, even internal performance comparisons within the same organization present difficulties that need to be taken into account when doing comparisons*".

Hence, the ISO draft suggests using environmental efficiency indicators at the process or product level for comparisons: "*These relative values will allow - under specific, controlled conditions – qualified comparisons of processes, products or services from different organizations, as well as for the identification of benchmarks, and best and worst practices or ratings*". By contrast, "*comparisons of the overall environmental performance of whole organizations are normally difficult or even impossible to achieve*".

In addition, it is suggested to focus on the important issues: "*Furthermore, comparisons can be made easier by focusing only on the most significant aspects - the Key Performance Indicators (KPIs)*". Such indicators should be based on a consensus among materially interested parties.

A methodology to develop sector specific environmental indicators is provided as well as examples to illustrate the approach. It constitutes a suitable starting point for the development a European methodology.

The Organisational Environmental Footprint (OEF) builds upon the concept to calculate total inputs (energy, materials) and outputs (emissions) across organisational boundaries (as the GRI and EMAS III core indicators), but extends site-level flows to include supply chain activities (and optionally downstream activities), and uses the same life cycle impact categories as PEF (rather than energy, materials, etc.). To expand corporate indicators in this way is pointless. It is rather absurd to claim that this approach can be used for comparisons and benchmarking – at the very best internal company comparisons can be made, and with limitations (see above). Clearly, this approach points to the wrong direction. Instead, we need a system which allows true benchmarking between corporations.

6. PEF specific remarks

The proposed EF methodology may lead to some improvements but will not address the fundamental problems and inherent shortcomings of LCA. It is in our view essential to first address these and set the frame for the overall assessment scheme before entering into details.

The draft method is widely based on existing standards (e.g. ISO 14040/44) and constitutes a remix of normative requirements with some additional elements. As an example, the need for Product Environmental Footprint Category Rules (PFCRs) is stressed (though it is unclear whether this means that they are compulsory). This is not a new concept as it is already included in ISO 14025 on type III environmental declarations (here they are obligatory). However, it is an open question what the quality expectations are (existing PCRs leave much to be desired) and, more importantly, who will prepare these PFCRs (industry, standards bodies or in analogy to the EU ecolabel system). But the main concern is that the PFCRs are focused on LCA methodology (by contrast, ISO 14025 considers at least human and environmental risk assessment as part of "additional environmental information").

7. The alternative approach: tailor-made environmental Key Performance Indicators

7.1 Basic principles

The alternative approach for developing indicators and related benchmarks is based on the following principles:

- The relevant environmental indicators for the relevant products, services, organisations and the macro level (global, European, national) must be selected in a political process involving all stakeholders resulting in a limited number of tailor-made environmental Key Performance Indicators (KPIs).
- Similarly, method selection must address environmental concerns identified in the political process rather than the other way round. It would mean to put the cart before the horse to derive political priorities from given methodologies.
- No scientific method can be a substitute for political priority setting as the selection of relevant aspects is by definition a value choice.
- There is no point in following a one-size-fits-all approach collecting an endless number of data (data collection for the sake of data collection) for all kinds of activities. This would be highly inefficient and expensive.
- No single method is able to suitably characterize the environmental performance of activities. Different methods have strengths and weaknesses which must be analysed and, combined in a meaningful and cost-efficient manner.
- Preference must be given to simple, transparent, reliable, measurable, easy to verify and cheap approaches.
- By contrast, methods with a high level of uncertainty which rely on a countless number of assumptions and subjective choices, which can be easily tuned to get the desired results and which are in practice very difficult to verify, should not play a dominant role.

- Methods that use theoretical concepts which are in stark contrast to established regulatory practices, proven approaches and traditional scientific concepts lacking broad agreement, should be avoided. By contrast, method alignment among different existing regulatory and voluntary instruments and established practices should be strived for.
- LCA, including the EF approach suggested by the Commission, may be a useful element for orientation in the initial phase of a comprehensive assessment of environmental performance, in particular for identifying hot spots, relevant life cycle stages and improvement options. However, due to its significant shortcomings, LCA needs complementary assessment tools. It does not seem to be the instrument of choice to address a number of environmental aspects (e.g. when impacts are dependent on space, time and background levels). It is, however, a suitable instrument for energy, greenhouse gas and mass balances.
- LCA indicator results are normally inappropriate for performance comparisons of similar products and, therefore, inadequate for communication such as labels or declarations or as basis for regulatory limits. Preference must be given to relevant raw material extraction, production, consumption and end of life indicators using appropriate metrics depending on the issue in question and benchmarks, rating scales and colour/letter codes.
- Good and robust process or product data are a prerequisite for meaningful indicators and for establishing sound environmental requirements. For instance, petrol consumption data as indicated by cars manufacturers have little to do with real life consumption. Hence, efforts should be made to develop appropriate test protocols.
- Corporate indicators should be used which allow for comparisons between organisations and benchmarking, i.e. they must focus on the process or product level rather than organisation level.
- BREF documents identifying the state-of-the-art are needed not only for the industrial sector but also for the non-industrial sector (the EMAS sector reference documents are a good start). Product BREFs could be also envisaged.
- Human health and environmental risks should be screened using appropriate tools, followed by simplified risk assessments using the precautionary principle. This means, for example, to red-flag CMR or other hazardous substances which may lead to critical exposures.
- Quantitative indicators should also use proxy indicators to address e.g. the level of consumption. For instance, meat consumption per capita may be a much more useful and easier to measure indicator than any CFP value. Durability of resource intensive products (service life time) may be better than resource efficiency.
- Indicators are useful only if there are significant improvement potentials.

- Redundancy in indicators should be avoided (i.e. where possible an environmental issue should not be expressed using different correlating indicators).
- Qualitative indicators are also an essential element of environmental information. This includes issues difficult to quantify e.g. compliance with organic farming or compliance with most advanced industrial practices (e.g. to prevent oil platform accidents).
- Not only regular operation must be covered but also accidental releases.

7.2 3-level framework for tailor-made environmental KPI identification

The "**Framework for environmental indicator identification**" is shown in figure 1. The basic idea is that an identification of environmental concerns (e.g. availability of resources), followed by the establishment of broad (non-quantified) environmental targets (e.g. less energy consumption), takes place at the policy level (making use, where available, of existing targets such as on energy efficiency). In a further step the main contributors to the various environmental burdens (or to meet the adopted environmental targets) are determined (e.g. in case of water consumption: food production, cotton production, certain industry sectors and construction). This is a first step to narrow the list of potential issues to be investigated in detail.

The specific quantified targets are then (along general lines) allocated to different sections – Area (global, European, national and local) – Organisations (enterprises) – Products. Here a further subdivision and prioritisation takes place. As an example, air pollution targets (NOx, particles) are established at the European level with specific national ceilings and specific local measures (traffic reduction measures). The corresponding indicators measure the relevant pollutants in the air. Complementary measures appropriate to meet the targets are implemented for key industrial emitters (e.g. energy providers, cement factories) and key products (e.g. cars, lorries). Relevant indicators include production and consumption related pollutant releases using appropriate metrics (e.g. amount of NOx per ton cement or km drive). All this should be accomplished in a coordinated way to make sure that the specific quantified targets can be actually met.

So the important point here is to start with a problem, to proceed to broad and specific targets and to identify the relevant indicators for the purpose of assisting problem solution rather than starting with a method uniformly applied to all areas ("putting the cart before the horse").

Although this paper supports a tailored approach using a choice of specific environmental aspects, indicators, benchmarks and associated methodologies, it also advocates a set of indicators for the macro level (national, European, global). These indicators should reflect the overall consumption level of our society which desperately needs to be reduced. The list includes:

- direct and indirect[36] consumption of energy per capita,
- direct and indirect consumption of water per capita,
- direct and indirect consumption of key materials (to be defined – e.g. scarce biotic materials such as fish, forest or critical abiotic materials) per capita,
- consumption of key products (such as meat) per capita,
- production of waste per capita,
- direct and indirect land use per capita.

The associated political targets such as reduction of energy use by 20% by 2020 and benchmarks e.g. primary energy use corresponding to world average 2008 (~21 kWh per capita, i.e. ~50% of current levels in Europe) and so forth - need to be defined.

The detailing of the results of the political process described above takes place at the study level. The boundaries between the 2 levels are, of course, fluid. This means e.g. that not all relevant product related environmental aspects can be derived from political targets - to some extent the relevant issues will have to be identified at the study level. A broad range of instruments is used to identify key processes, to suitably characterize them and to identify improvement options making use of the existing knowledge base and established practices. The scope and method definition can only be done in a balanced stakeholder process with final political responsibility (adoption of measures and indicators).

Figure 2 illustrates the framework using the **example direct and indirect water consumption** as an example. Water scarcity has been recognised as an issue of big concern, particularly in arid areas. Water saving measures seems to be warranted. As stated above, main contributors to direct and indirect water consumption are food production (particularly meat and notably beef, but also food of plant origin), cotton production, certain industry sectors and construction (toilets, bath, and shower). Key processes include farming practices (e.g. livestock, irrigation), industrial processes including cooling and water use in lavatories. Improvement options can be identified based on a sound technology assessment showing best practices (BAT) such as most efficient irrigation techniques or water saving equipment (waterless urinals) but are also consumption behaviour related (meat consumption). Although at the "macro" level (country, regional - Europe, global) indicators such as overall direct and indirect water or meat consumption per capita are useful, the process level will focus on indicators such as litre per kg produced food. Benchmarks for the former would have to be politically agreed – for the latter they are given by the most efficient technologies in the relevant production areas. For local areas, other indicators such as ground water level change may be relevant.

[36] indirect meaning the impact/resource use embedded in products consumed. For example the energy used in the production or delivery stage of a product, which may be beyond national/regional boundaries.

There is no point in identifying water consumption measures and indicators for all kinds of processes and products when an overwhelming part of the water use is related to just a few activities. This reinforces the need for tailored approaches and for early prioritisation. However, at the macro level water consumption will be always relevant.

The study level is more detailed in figure 3 using **buildings as example**. The example is related to products – but the basic elements are equally important for studies on organisations and area related matters (classical environmental media policies). The focus of the studies will differ, of course. There will be almost always a knowledge base to begin with. A stakeholder analysis of the status quo should be the starting point for the definition of study scopes and methods taking into account the established political targets rather than leaving all choices to the Commission or consultancies or methodologies. In case of buildings an orientational LCA of the most relevant types of new buildings focusing on material and energy flows will tell us that an overwhelming proportion of energy consumption occurs in the use stage with a minor contribution from certain construction products such as walls, basement, ceilings, etc. On that basis, indicators for energy consumption in the operational stage of the building and embedded energy for a limited number of products seem promising. However, the improvement potentials must be established by a sound technology assessment including BAT for the identified construction products as well as building technology. At the building level, passive or zero energy houses will be identified as state-of-the art giving an enormous saving potential, in particular as regards the building stock. However, the review will also reveal that calculation methods differ largely and need improvement as well as harmonisation. Hence, additional studies may be commissioned to establish sound calculation procedures providing the necessary detail to calculate reliable energy consumption figures. The method could also be subject of a comparative field test. The indicator is expressed as (primary or final) total energy consumption per m^2 and year.

Note: it is significant that LCA standards for buildings do not even arrive at this level of detail.

Risk assessment methodology is used in a simplified manner using the precautionary principle for issues related to noise, dust, particles, chemicals (e.g. indoor air emissions, nanomaterials) and radiation. Again, a state-of-the-art analysis will identify best practices (such as low emission construction machines or lorries, dust attenuation measures, low emission construction products) for suitable indicators, benchmarks and requirements.

Qualitative indicators will be also helpful. Many issues cannot be covered using quantitative indicators or quantitative impact assessment. For instance, big construction projects need environmental impact assessments based on expert judgement. Compliance with sustainable production methods (e.g. for wood) or guidelines for (design for) selective demolition to facilitate recycling are other examples.

8. Links to political instruments

At the end of the day, indicators and benchmarks will be used to establish requirements in regulatory measures – be it for establishing baseline requirements for all actors on the market or for voluntary tools. Figure 4 shows the **links of the suggested framework to political instruments.**

It goes beyond the scope of this document to address the management of political instruments, but it is essential to point to the need to align the various pieces of legislation to ensure consistent application of the principles outlined in this document as well as, where possible, harmonised indicator development and use.

As an example, the indicators forming the basis of requirements for a specific category of products such as TV sets should be the same – irrespective of whether such requirements are incorporated in the Energy Related Products Directive (or any extended SCP framework), energy label (as far as the indicators are applicable), eco-label or GPP criteria. The ambition level will be, of course, normally different.

Along the same lines, approaches at the corporate level (BREF and EMAS sector reference documents and, where applicable, GPP criteria) need to be aligned.

Corporate indicators can be used in setting of product specifications vice versa. This calls for alignment of corporate and product level indicators.

Finally, product and corporate requirements must be adequate to meet environmental media related regulatory demands and macro level targets.

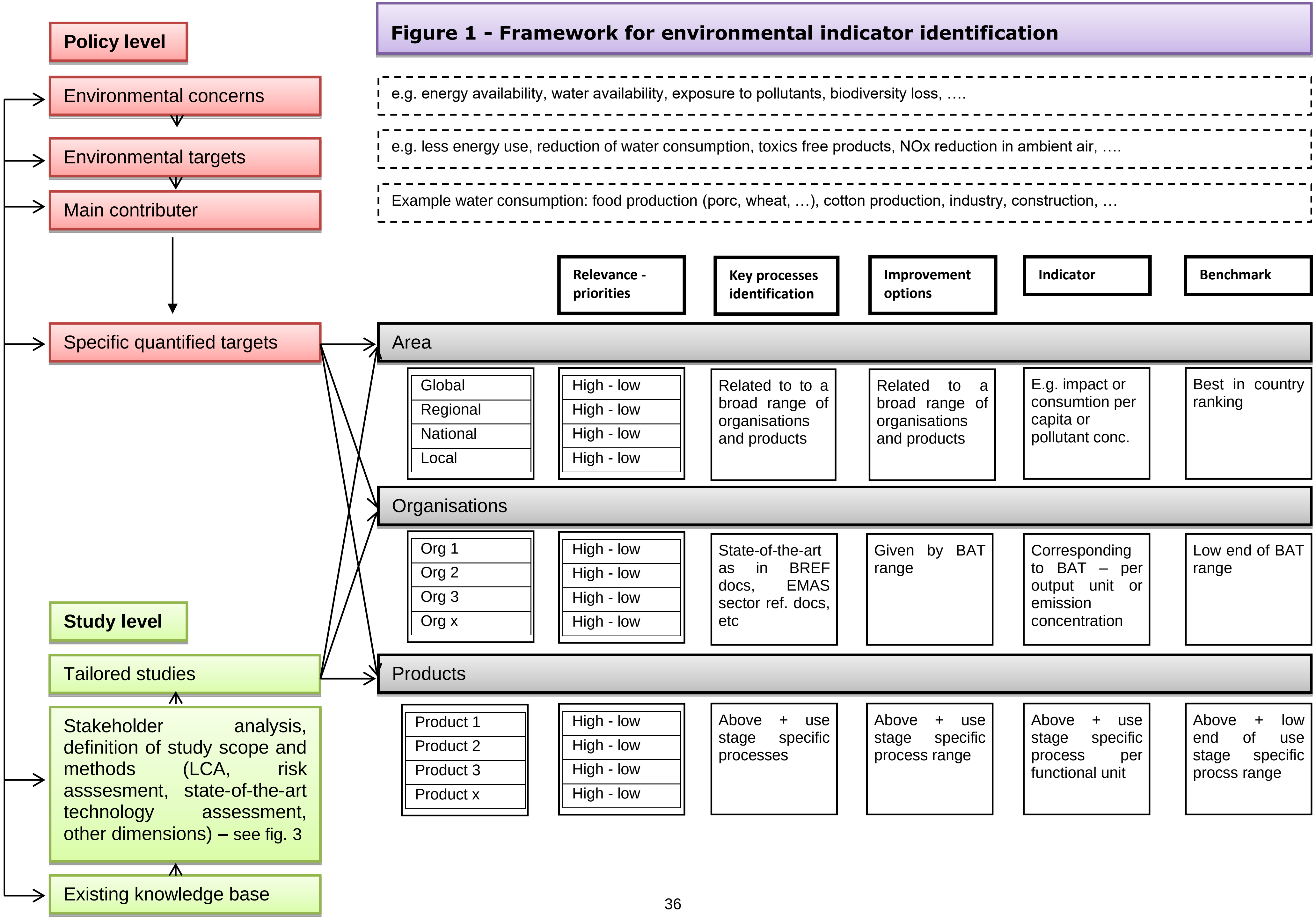

Figure 1 - Framework for environmental indicator identification

Figure 2 - Framework for environmental indicator identification, Example: Direct and indirect WATER USE

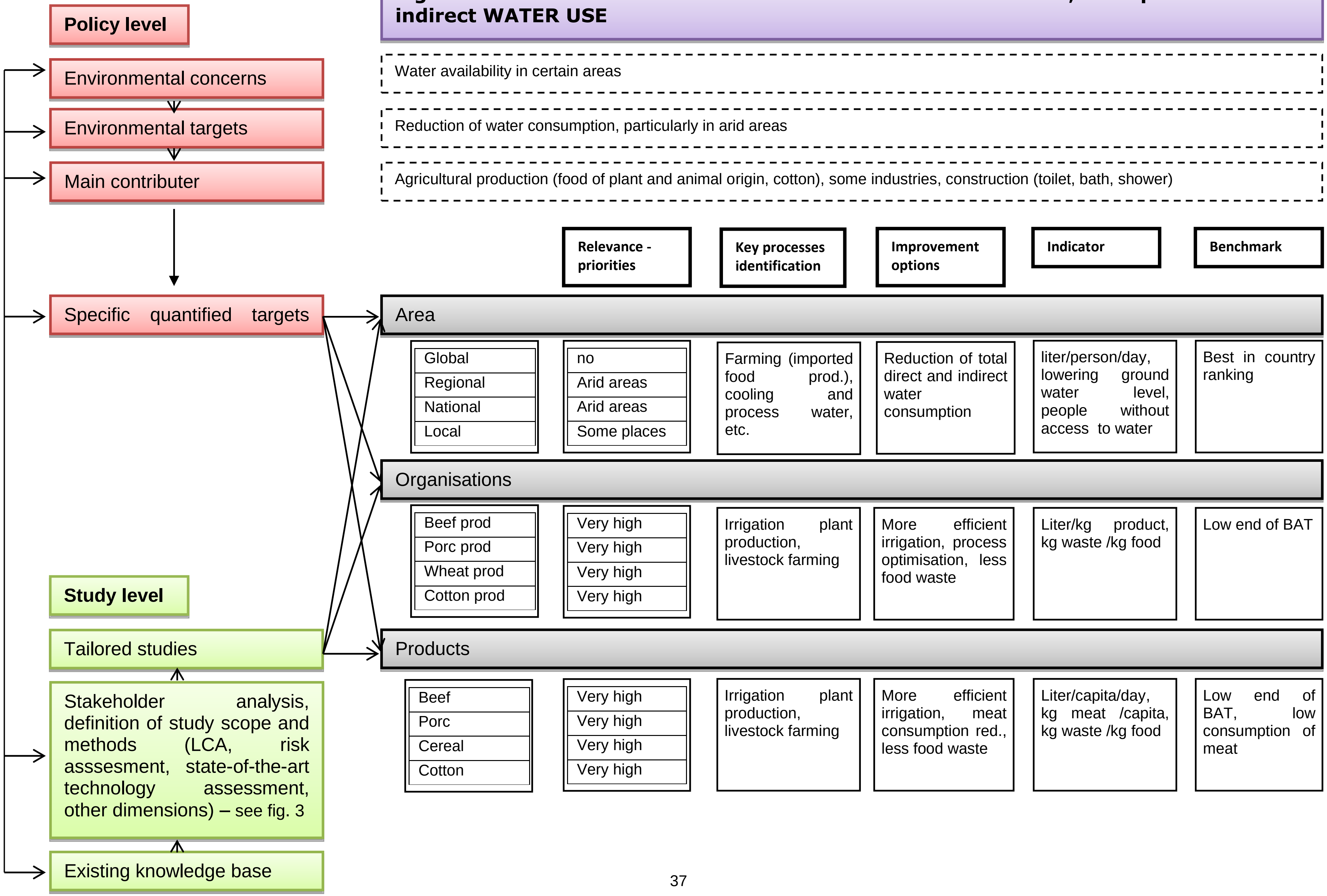

Figure 3 - Framework for environmental indicator identification, Example study level products BUILDINGS

Study level

Tailored studies → Buildings

Existing knowledge base

Stakeholder analysis, study scope (product group), env. aspects and methods identification

Study level	Key processes identification	Improvement options	Indicator	Benchmark
Methods identification – **other dimensions**, qualitative aspects such as recyclability, eco-design, persistant substances,	End-of-life, (selective) demolition, reuse, recycling	More recycling through proper separation of materials	Design for recycling, "recyclability"	Compliance with best practice
Methods identification – **State-of-the-art technology assessment** – BAT or product BAT	Heating, cooling energy in use stage	Passive house or zero energy house	Heat-final-primary energy per m^2	0 kWh/m^2
Methods identification - **Risk Assessment (simplified)**, mainly for chemical risks, noise, particles, other local effects	Indoor air emissions, dust and noise on building site	Low emission materials, trucks and const machines	Chamber conc., vehicle emiss. g/kWh, noise level	Natureplus limits, Euro VI class for lorries, 78 dB
Methods identification - **LCA** mainly for material and energy balance, relevant stages in the life cycle, "orientation"	Indicates that >80% energy consumption in use stage	Focus on use stage	LCA indicator not useful	LCA indicator not useful

Buildings	Relevance
Commercial	Very high
Residential	Very high
Educational	Medium
Government	Medium
Industrial	High
Agricultural	Low
Military	Low
Parking	Low
Religious	Low
Other	Depends

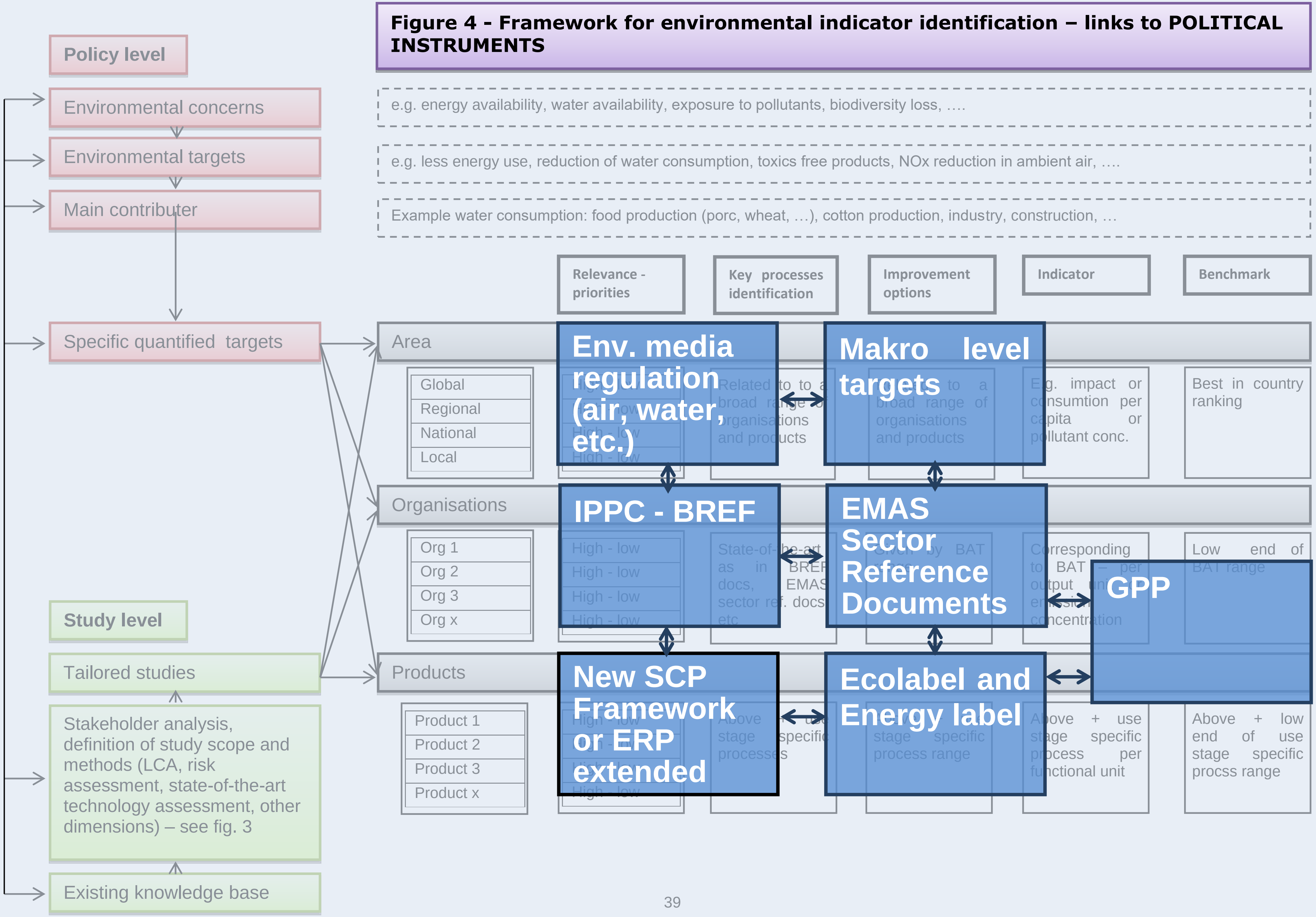

Figure 4 - Framework for environmental indicator identification – links to POLITICAL INSTRUMENTS