ECO-INDICES: WHAT CAN THEY TELL US?

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FOREWORD

Manufacturers are increasingly mindful of the life cycle environmental impact of their products. As part of their strategy to minimize this impact, they pay particular attention to material selection.

A growing number of eco-efficiency models have been developed to help manufacturers evaluate the environmental impact of the materials they use in their products. The approaches used in these “eco-models” range from the Life Cycle Assessment defined by ISO standard 14041, to an ever-increasing number of other life cycle–based models that aim to provide a simpler and more user-friendly tool to measure environmental impacts.

This paper addresses the use of the eco-indicator—or eco-index—approach of identifying, aggregating and collapsing environmental information into one value. The authors examine the premise and mechanics of the eco-index approach and discover key issues associated with this means of measuring environmental impact. They conclude that eco-indices, by their very nature, cannot provide reliable measures of the environmental impacts of a material, particularly because of the arbitrary or subjective nature of valuation approaches for impacts.

This publication aims to support the efforts of developers and users of eco-models to continuously improve the quality of the information that goes into environmental decision making, particularly as it relates to material selection. We hope it will help to further discussions and developments in this important field of endeavour. We welcome any feedback or suggestions from readers.

Gary Nash,
Secretary General
ICME

EXECUTIVE SUMMARY

Almost all models developed to measure environmental performance have in common three basic stages: inventory, classification and characterization, and weighting. There is the potential for error at each of these stages. Since the second and third stages involve multipliers to normalize data, the effect of any errors will usually be magnified.

In the present study, the analysis focuses on single-indicator (also called "eco-indicator" or "eco-index") models that are explicitly based on life cycle methods. The study also looks at the ability of these models to provide reliable information on the specific behaviour and impacts of metals.

One key issue with the use of life cycle inventory (LCI) data in eco-indicator models today is that it appears to differ from the original intent. LCI data are now used to compare materials and products rather than production systems. This confusion is at the source of most limitations in measuring and comparing environmental performance. The lack of complete data sets to describe and quantify environmental performance often adds to these limitations.

Classification involves *identifying* the factors that contribute to a specific environmental problem. Characterization is the process of *quantifying* these contributions. Most issues that arise in the impact assessment phase of the life cycle are associated with the characterization of data. The lack of standardized methods for aggregating data, together with the lack of uniformity in impact assessment methods, are among the key issues.

All of the issues identified above will influence the final result derived from the use of an eco-index. However, a potentially more critical issue as far as biases are concerned may be the use of weighting techniques, which assign weighting factors to each impact category. These factors are intended as indications of relative importance based on the potential or realized impacts associated with each impact category. Aggregating parameters based on their relative importance is difficult at best, even within specific data categories (e.g. global warming potentials of different air emissions).

To combine data that address different environmental problems, one must also be able to determine the relative significance of these impacts. For example, the impact of global warming would need to be compared with that of acid rain. Which is more important? The answer to such a question may not only have to address the relative importance of these factors, it may also have to quantify it. Thus, if global warming is thought to be more important than acid rain, is it three times, 10 times or 100 times more important?

While it is often necessary to make such judgments, it is inappropriate to suggest that they are science-based or unbiased. The key issue is the failure to alert the user or reader to the fact that these judgments have been made, or to thoroughly discuss the basis for making them.

The report concludes that the specific characteristics of metals (e.g. recyclability, complexation, bioavailability, transformation) are inadequately modelled by eco-indicator models. This problem is in fact common to all types of eco-modelling approaches and reflects a lack of information and knowledge by both developers and users of eco-models. For example, the modelling of the impact of metals is generally assumed to be similar to that of the impact of organics.

Even if the characteristics of metals were better known and embedded in the eco-indicator models, problems would still exist. The models would still generate unreliable information on the impact of metals (or of any other material for that matter) because the fundamental flaws inherent in the development of eco-indices would remain.

It is difficult to see how any model could be developed to calculate an eco-index, provide significant information to users, and still meet the tenets of scientific validity and objectivity. With evidence that specific types of materials may be regularly discriminated against by one model or another, it seems prudent to forego the use of any single-indicator model.

PREFACE

To better understand if the environmental measures proposed in eco-indicator models are appropriate for use in product design, ICME contracted Boustead Consulting & Associates Ltd. (BCAL) in August 1998 to review the current situation.

In this report, the authors focused on determining whether eco-indicator models provide reliable information by looking for general problems expressed across various models. Key flaws, both conceptual and real, are identified; where possible, examples are provided.

The individual impact assessment methods employed in each eco-indicator model were not reviewed as it was beyond the time and budget provided for this project. However, the implications of using different impact assessment tools in the context of calculating an eco-indicator value are discussed.

The impetus for this study was a belief that techniques used to assess the environmental impact of specific substances in the environment were discriminating against metals. The initial findings, however, pointed to more fundamental problems as they showed that the methodology used to develop eco-indices cannot provide reliable and consistent results.

The conceptual basis for calculating an eco-index is in itself problematic. Potential errors may in fact overshadow any actual errors in calculating specific impacts for any one class of substances. Even using correct assumptions and calculations for the impact of substances in the environment may not overcome the biases associated with using a design tool such as an eco-indicator model, in which data sets are combined beyond reliable scientific protocol.

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1. INTRODUCTION

The popularization of environmental issues, most recently through the concept of sustainable development, has increased people's awareness about environmental problems around the world. As a result, decision makers from governments, industry and conservation groups, as well as individual product and process designers, are being pushed to increase the use of environmental information in order to make better choices and reduce environmental impacts (i.e. to improve environmental performance).

Over the past few years, a number of organizations and companies have been working to create tools to assist various user groups in evaluating the environmental impact of materials, products and companies. The approaches, models and software programs developed to measure environmental performance have been introduced under several terms, from *eco-efficiency*, to *industrial ecology,* and *design for the environment* (see Appendix A). These tools vary from highly quantitative models to less formal qualitative models.

A number of new approaches[1,2] are also being considered in an effort to include an even wider array of parameters, such as social, economic and esthetic factors that address the broader issues associated with sustainable development. A cursory examination of these new approaches shows that there are still considerable disagreements about what to measure and how to interpret the results.[3,4] These disagreements appear to stem from an attempt to satisfy too many different needs in a single analysis—from physical and chemical requirements to economic and social goals.

One of the trends in eco-tool development has been eco-indicator models that compute a single index value to represent the relative environmental impact of a material or product. Two factors appear to have driven this development. First, there is a feeling that the number of parameters to be successfully incorporated into an environmental analysis is getting so large as to be unmanageable and too expensive, especially when used in applications such as the design process. Second, there is the understandable desire to have a simple way of summarizing all of this information so that the intelligent layperson can get an idea of the overall environmental implication of the system.[5] Eco-indicator models are often marketed as a less expensive and user-friendly alternative to more comprehensive environmental engineering tools. This may inappropriately suggest that they can be used by individuals without proper training in environmental sciences to support design and policy decisions.

It is important that any model designed to measure environmental performance produce results that can be used to make sound decisions. This means understanding exactly what caused the calculated impacts, which impacts were present and at what level, and what trade-offs if any may be associated with using one design over another. Because eco-indicator models calculate a single score of relative potential environmental impact, the control over decisions for targeting specific environmental impacts is no longer in the hands of the user of the model.

For example, life cycle inventory data sets are generally regarded simply as a compilation of information from which the user selects those parts that are needed to address specific problems. Thus, if the intention is to examine the greenhouse gas implications, the users would select CO_2, CH_4, N_2O, etc., from the air emissions set and ignore the rest of the data. Where multiple objectives are to be met (e.g. reduction in global warming, acid rain and fossil fuel consumption), the user can see the type and scale of environmental trade-offs inherent in the system. Models that calculate eco-indices instead aggregate data from all the various impact categories to provide a decision about the relative merits of the material or product under investigation, thus masking the internal trade-offs between impact categories. This does not allow the user to distinguish between specific goals or objectives, such as reductions in global warming gases and ozone-depleting chemicals, that may be important to the organization or to the local and national governments.

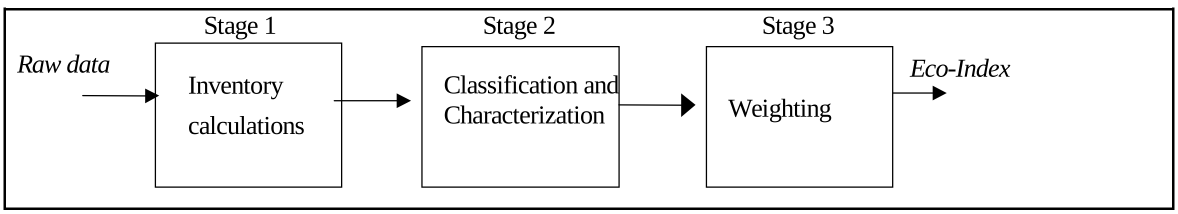
There is a potential for incorrect decisions, which are not only unfortunate, but can also be costly. Incorrect decisions can lead to negative environmental impacts that have significant financial costs over time, not to mention potentially irreversible environmental losses in habitat or biodiversity. Similarly, they can put companies out of business. Since design choices lead to selecting some materials while avoiding others, incorrect calculations by models may significantly harm manufacturers by routinely causing decisions to avoid or discount specific materials. These types of decisions should be made very carefully with a full understanding of the data.

2. DEVELOPMENT OF ECO-INDEX MODELS

Almost all approaches developed to measure environmental performance have in common three basic stages (Figure 1). The general approach followed in developing these tools has been the life cycle assessment (LCA) framework proposed by the Society of Environmental Toxicology and Chemistry (SETAC).[6–8]

1. The raw data collection and analysis, which form the inventory calculations.
2. The classification and characterization stages, where the inventory data are grouped together and summed in an appropriate manner so that each of the grouped data sets describes some environmental facet of the system.
3. These grouped data sets are multiplied by some weighting factor and summed to give an overall index or limited set of indices.

Figure 1. Stages in the Production of an Eco-index



In general, this taxonomy is consistent with that identified by the International Organization for Standardization (ISO) for conducting the life cycle inventory and life cycle impact assessment phases of a complete LCA[9]:

- Inventory—compiling a list of relevant inputs and outputs of a system;
- Classification and Characterization—evaluating the potential environmental impacts associated with the identified inputs and outputs;
- Weighting (previously termed Valuation)—weighting the results of the characterized impacts relative to one another.

3. POTENTIAL SOURCES OF ERROR IN ECO-INDEX CALCULATIONS

It is important to recognize that there is the potential to make errors at each stage of this three-stage procedure. Each error made will be carried through to the final index calculation. Furthermore, since Stages 2 and 3 involve multipliers to normalize data, the effect of any errors will usually be magnified. It is therefore important to examine each of the three stages in turn to see what errors and omissions are likely to occur.

While not all models specifically state the use of LCA methods, most proposed and existing eco-tools contain the same stages as LCA in one form or another, although they may not follow the precise guidelines or standards produced by SETAC or ISO.

In the present study, the analysis was limited to eco-indicator models explicitly based on life cycle methods. The following three sections examine each of the stages identified above.

4. INVENTORY CALCULATIONS

The first systematic procedures for carrying out life cycle inventory (LCI) calculations were published in 1979[10] and were based on energy analyses. One key issue with the use of LCI data in eco-indicator models today is that it appears to differ from the original intent. LCI data are now used to compare materials and products rather than production systems. This confusion is at the source of most limitations in measuring and comparing environmental performance. The lack of complete data sets to describe and quantify environmental performance often adds to these limitations.

4.1 Improper Use of Systems Analysis

To understand whether existing or proposed models properly describe and measure the environmental performance of products and materials, it must be remembered that the models describe a *system* in which a product is produced and used. A production system is not concerned with economic, esthetic or social factors; rather, it is concerned with processing materials, and the operations involved in processing materials are governed by the laws of science. Moreover, in a systems-based analysis, valid comparison can be made only between equivalent systems. Thus, a system whose function is to produce a specific type of material such as steel cannot legitimately be compared with a system whose function is to produce a different type of material such as PVC. Neither the steel system nor the PVC system can ever produce a material other than the one it is designed to produce. Each of these materials is used differently, even to produce the same products (e.g. irrigation pipes), in that the amounts needed to make the same product are not equivalent. Therefore, a material-to-material comparison is incomplete and inaccurate until one can include the amounts needed to manufacture products. Similarly, comparing two products can be and often is incorrect because it takes different amounts of each product to fulfil the same function. In addition, comparing systems that have different boundaries (e.g. cradle-to-gate or cradle-to-grave), even though they have the same function, will produce unsound comparisons. Until all parameters in the system are equivalently satisfied, any comparison is suspect as it can produce erroneous results.

The above considerations provide the first criterion for identifying whether a model produces a biased or unbiased measure of environmental performance. Any model that uses measures other than those associated with the physical parameters of the system introduces elements that may give rise to yet another form of bias. Consequently, the result cannot be regarded as a science-based measure of environmental performance for a production system.

Of the models reviewed, Nortel's Environmental Performance Index (EPI),[11] Dow's Eco-compass.[12] and the metrics put forward by World Business Council for Sustainable Development (WBCSD)[3] all either use or propose to measure parameters other than those described by the physical system.

The indicators tested by the companies involved in the Eco-efficiency Task Force of Canada's National Roundtable on the Environment and the Economy (NRTEE)[2] included parameters other than physical ones to measure environmental performance (e.g. energy and materials measures can be reported either per unit of production output or per unit of revenue). However, in its final draft, NRTEE is careful to point out that using non-physical parameters may not be the best choice. According to the report, participants of the roundtable seemed to agree that measurement parameters should be kept separate and that physical parameters appear to make the most sense. This is further indicated by the fact that the participating companies that tested proposed measures or indicators mostly opted for use of physical parameters without links to the economic system. For example, the energy indicators proposed for examination ranged from total energy to greenhouse gas emissions associated with energy. Most companies participating in this NRTEE exercise chose to use energy as a function of product output rather than as a function of some economic measure such as revenue or profit.

The Eco-Indicator 95[5] model reveals a slightly different aspect of this issue. While the inventory data used in the model were developed using SimaPro—a life cycle model—the data sets are organized for use with eco-indicators by material and process. This allows users to compare materials and processes rather than equivalent systems. There is no default mechanism built into the program that prevents users from comparing non-equivalent systems. In fact, the manual for the model instructs users that the model will help them discriminate between materials choices. There is, however, no scientific basis for making such comparisons.

4.2 Incomplete Analysis

Another criterion that can be used to discriminate between eco-efficiency models is whether they examine or can examine all parameters within the system or only a subset of important or significant ones. Incomplete data sets can occur for two major reasons: either the data do not exist or an *a priori* decision is made to leave the data out. This latter process is often used in life cycle thinking and is commonly found in a number of efforts, including many of the eco-labelling schemes around the world.[13,14] Any evaluation protocol subject to either of these limitations must be deemed inherently biased unless further studies prove the validity of the identified measures.

It is important to recognize that the issue of incomplete data sets is different from that of proper scoping and problem formulation as recommended by SETAC and ISO. Eliminating specific measures at the outset of a study because they are thought to be ancillary and/or inconsequential does not necessarily affect the proper scoping and identification of impact categories. But it can result in calculating inaccurate figures for specific impact categories. An example is the analysis of glass making (Table 1).

Table 1. Glass Making (per kg glass)

	Mass input	Production energy	Energy from supplied input
Sand	0.7 kg	1.0 MJ/kg	0.7x1.0 = 0.7 MJ
Selenium	0.003 kg	350 MJ/kg	0.003x350 = 1.1 MJ

The major input in terms of mass is sand at 0.7 kg, while one of the ancillary inputs is selenium at 0.003 kg. Yet, in terms of energy the ancillary input makes a bigger difference with almost 50% more energy input into the life cycle. Clearly, estimating data or finding appropriate surrogate data would be important, as eliminating selenium altogether would change energy use significantly, which in turn will significantly affect emissions data.

The methods proposed by WBCSD and NRTEE may lead to such a problem, albeit in a more subtle way. Both methods propose appropriate environmental or "sustainability" indicators. In so doing, they give readers the impression that the respective organizations believe some indicators are more significant than others. While this may end up being so for the majority of cases, it lays the groundwork for allowing other important but less-known environmental impacts to be ignored. Whenever a proposed method or system of analysis makes *a priori* decisions about what to measure, or more importantly what *not* to measure, the probability is increased that something of importance will not be accounted for. As many of the proposed measurement indicators (e.g. energy, raw materials, pollutant releases) already require an analysis similar to an LCA to gather the necessary data, it would seem more prudent to support a complete LCI. A complete LCI would provide the measures recommended by NTREE and WBCSD, as well as ensure that the data were collected across the entire life cycle. This would mean that end users of the information would not only be able to determine if a specific factor (e.g. energy) is creating a problem, but also where the problem is, and how a proposed solution might affect other factors.

Another concern associated with the notion of complete (and accurate) LCI is whether all the various aspects of materials are properly accounted for. For example, many metal, plastic and paper products can contain a significant amount of recycled material. Moreover, some of these materials can be recycled numerous times and some extend the useful life of products significantly. Models should be able to account for these types of scenarios if they hope to provide the user with the flexibility needed to accurately measure environmental performance. Using Eco-Indicator 95 as an example once again, recycled content appears to be the only characteristic specifically built into the calculation of eco-indicators. Characteristics such as product life are not included. Even in the recycling schemes, there is no distinction made between products that can be recycled numerous times and those that cannot, or between different types of recycling (open-loop versus closed-loop).

Whether data are estimated incorrectly or are missing, the implications for calculating an eco-index are considerable. If final inventory numbers are significantly off in one or more categories, the error can become quite large by the time the values are multiplied by an equivalency factor, added to other like values, and then multiplied by a weighting factor to complete the life cycle impact assessment. The extent of the error will also be reflected in the final eco-indicator value, potentially causing users to make incorrect decisions.

5. CLASSIFICATION AND CHARACTERIZATION CALCULATIONS

Classification involves *identifying* the factors that contribute toward a specific environmental problem. Characterization is the process of *quantifying* these contributions. Most issues that arise in the impact assessment phase of the life cycle are associated with the characterization of data. The lack of standardized methods for aggregating data, together with the lack of uniformity in impact assessment methods, are among the key issues.

5.1 Classification

Few problems arise with classification of inventory results because the links between most common inputs/outputs and known environmental issues have been identified. Even though a misclassification could yield inaccurate results in terms of an eco-index, this is not very likely to happen.

5.2 Characterization

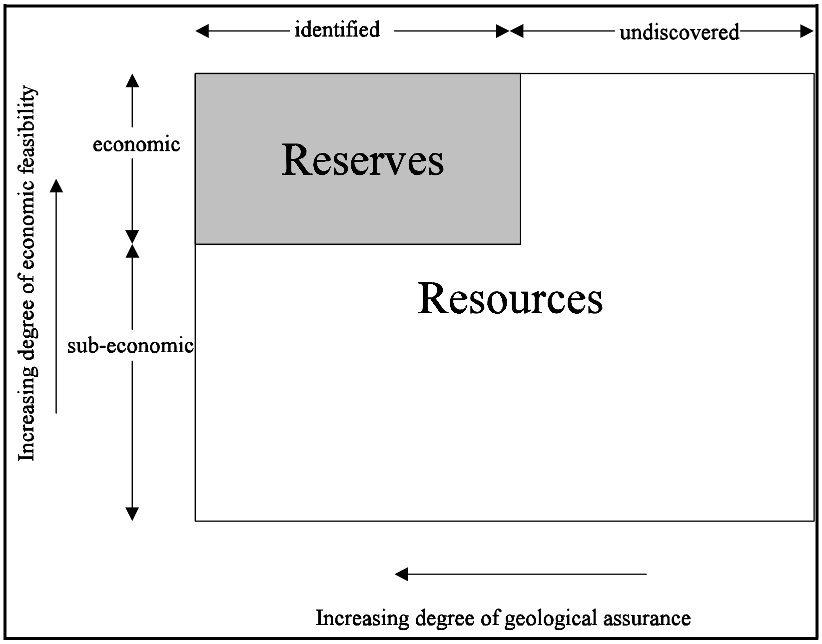
There are a number of proposed approaches for conducting impact assessments in the context of an LCA.[15] These approaches vary in breadth and depth, from aggregating data within impact categories based on equivalency factors, to site-specific exposure/effects assessments.

At present, impact categories associated with global and regional effects have had equivalency factors established that are well documented and have some international acceptance (see Appendix B for specific details). These are:

- Global Warming Potentials
- Ozone Depletion Potentials
- Photochemical Ozone Creation Potentials
- Acidification Potentials
- Eutrophication Potentials

For the category of resource depletion, there is much less agreement about appropriate impact measures. The main measure proposed for raw materials resources has been the rate of depletion of reserves. In theory, this should be possible and straightforward. However, this attractive idea has one major drawback: the numeric value for reserves used in life cycle calculations for any given raw material changes with time.[16] Plotting economic feasibility against geological assuredness (Figure 2), McKelvey made the point that, to be designated as a reserve, the resource has to be identified and has to be economic to extract. As the price of a resource changes or as further supplies are discovered, the numeric value of reserves changes. Thus, using reserves as a base against which to measure depletion of resources is likely to be extremely uncertain, even though there is no physical change in the actual amount in the earth.

Figure 2: The McKelvey Diagram



One of the more complex aspects about the characterization phase of life cycle impact assessment involves the calculations for categories of local effects (e.g. human toxicology and ecotoxicology). The toxicity equivalency factors for these impact categories are the least developed and the most contentious due to the complexity of the numerous factors involved—the fate of substances can depend on degradation rate, bioaccumulation, transformation, complexation, evaporation and deposition. There is also the need to account for acute toxicological effects, irritation, allergenic reactions, genotoxicity, carcinogenicity, neurotoxicity and teratogenicity.

Calculated results vary significantly and provide little consensus about which methods to apply. For example, an analysis was conducted testing the usefulness of four methods for assessing human health impacts of chemical emissions.[17] Each method—toxicity-based scoring, sustainable process index, concentration/toxicity equivalency and human toxicity potential—used a different model with varying levels of complexity and sophistication to calculate a toxic equivalency potential for two toxicants: pentachlorobenzene and styrene emissions. The predicted impacts depended on whether and how exposures were considered by each evaluation method. The relative toxicity scores produced by these methods varied by three orders of magnitude when the same compounds were analysed, which shows that methodological assumptions are critical. Regardless of these types of complications, a number of equivalency conversions have been proposed and used in life cycle impact assessments.[18–21]

5.3 The Example of Eco-Indicator 95 and Metals

With regard to biases resulting from lack of scientific understanding in specific models, several issues associated with the impact assessment of metals in Eco-Indicator 95 have been documented.[22] Examples include:

- Metals are assumed to disperse widely and remain toxic due to their persistence, while organic compounds are assumed to degrade quickly.*
- Metals may be subject to double accounting in that they accumulate eco-points twice if their production contributes to both smog and acidification. This is in contrast to allocating the load between the two impact categories.
- The model rates heavy metals relative to lead and is calibrated to reduce their emissions in order to protect Eastern European children. In fact, the significant exposure pathway for lead is not in air emissions but in the ingestion of dust, food and water.
- There are some inconsistencies in data that are leading to serious overestimations of the impacts of nickel and barium.
- Impact calculations are based on European situations only and are not appropriate for measuring impacts of metals in other geographic locations.
- No consideration is given to the speciation, bioavailability or rates of transformation for arsenic, nickel and chromium.

Although these specific issues focus on metals, several could apply equally to other substances (e.g. inconsistencies in data, no consideration of changes in chemical structure and availability over time, double accounting for impacts).

* Editor's note: In fact, scientific data demonstrate that metals in bioavailable form transform rapidly into non-toxic compounds.

6. WEIGHTING (VALUATION)

All of the issues identified above will influence the final result derived from the use of an eco-index. However, a potentially more critical issue as far as biases are concerned may be the use of weighting techniques. These are techniques that assign weighting factors to each impact category. These weighting factors are intended as indications of relative importance based on the potential or realized impacts associated with each impact category. As discussed under the previous section, aggregating parameters based on their relative importance is difficult at best even within specific data categories (e.g. global warming potentials of different air emissions).

To combine data that address different environmental problems, one must also be able to determine the relative significance of these impacts. For example, the impact of global warming would need to be compared with that of acid rain. Which is more important? The answer to such a question may not only have to address the relative importance of these factors, it may also have to quantify it. Thus, if global warming is thought to be more important than acid rain, is it three times, 10 times or 100 times more important?

While it is often necessary to make such judgments, it is inappropriate to suggest that they are science-based or unbiased. The key issue is the failure to alert the user or reader to the fact that these judgments have been made, or to thoroughly discuss the basis for the judgments.[8]

Appendix B provides an overview of several proposed methods for weighting LCI results. In general, the approaches can be grouped as follows:

- **Proxy Approaches**

  These approaches use only one or a few quantitative measures, stated to be indicative of the total environmental impact or even of (un)sustainability. Most proxy approaches are input-oriented and focus on trying to see where the inputs from two comparative systems are the same. If inputs are the same, the assumption is that they can be eliminated because they will cancel each other out in any comparison. This is primarily done to make the LCI calculations shorter. For example, comparing two systems with equal amounts of energy inputs, one can set up the calculations so that energy is not included. This can and does lead to oversights regarding impacts such as ozone depletion and ecotoxicity. If the energy sources are different, there can be decidedly different emissions that would also be eliminated from the calculations, which in turn can inadvertently eliminate significant differences in impact profiles.

- **Technology Abatement Approaches**

  These approaches tend to favour technological options for reducing environmental burden. For example, these approaches can give increased weighting to technologies put in place to reduce emissions than to practices put in place to improve production efficiencies, even though both may result in equal reductions in environmental burdens.

- **Monetarization**

  The principle of monetarization is to attach monetary values to each specific quantitative value calculated in every impact category. In this approach, economic mechanisms are regarded as a correct guideline for weighting. However, neither a classification nor a terminology has yet to be agreed upon.

- **Authorized Goals or Standards**

  Two examples are the distance-to-target and the critical volumes approaches. The former measures the difference between actual and target values for an impact; the latter divides atmospheric emissions by air quality standards in order to find the volume of air that is potentially polluted up to the critical value.

- **Authoritative Panels (societal approach)**

  The principle behind this approach is to determine the preference of society for establishing priorities with regard to environmental improvement. A panel may be set up to represent lay people, societal groups, "experts," governments or possibly even international bodies. This approach is, by definition, subjective. If the work of the panel is done in a transparent and open manner with a full explanation of the weighting scheme employed, there is scope for data transparency. A lack of transparency would undermine the rationale for having a panel in the first place.

There is the potential for weighting factors to create even greater biases than in the case of the equivalency factors used to normalize impacts during the characterization step. While the multipliers used in the characterization of impacts in the previous phase of calculations are normally based on similarities (e.g. structure, reactivity) among compounds, the multipliers used in this step may not have a basis in science. Most methods rely on some political determination (e.g. national environmental policy) or the judgment of experts. As a consequence, there could be differences of several orders of magnitude in the final results, depending on the method chosen.

The single-indicator models shown in Table 2 use two different weighting methods: MET (Materials, Energy and Toxics) and Eco-Indicator 95. It is readily apparent that there are differences of several orders of magnitude in a number of factors, such as acidification, ozone depletion and human toxicity/carcinogenicity. Should these factors be used in the same indicator model, they would yield significantly different results. In fact, these differences are not just ones of scale, as relative differences between categories are present as well. Ozone depletion, for example, is the largest factor by 10 times in one method yet the smallest factor according to the other.

Table 2. Weighting Factors Calculated from Two Different Methods

Weighting factors in MET Model		Weighting factors in Eco-Indicator 95	
Resource depletion	0.88	Resource depletion	(not considered)
Greenhouse effects	0.89	Greenhouse effects	2.5
Acidification	0.49	Acidification	10
Smog	0.4	Smog	5
Eutrophication	0.29	Eutrophication	5
Ozone depletion	0.12	Ozone depletion	100
Human toxicity	0.3	Carcinogenicity	10
Ecotoxicity	0.13	Heavy metals in air	5

7. ECO-INDICATOR MODELS IN PRACTICE

Given the problems raised in this and other reports on single-indicator assessment methods, one would expect these types of models to produce inconsistent results. Engineers at Ford Motor Company put this assertion to the test.[23] Three single-indicator models were selected and used: MET, EPS (Environmental Priorities Strategy) and Eco-Indicator 95.

The MET method consists of assigning inventory data to eight impact categories, computing an effect score for each category using the CML (the Centre of Environmental Science of Leiden University) method,[24] and then normalizing each effect score by a country effect score multiplied by a country improvement factor. Finally, these category scores are summed to produce a single-indicator value.

The EPS method was developed in Sweden.[25] It calculates an environmental load unit which essentially comes from taking LCI data, classifying it into five categories known as safeguard areas, and multiplying each item by unit effects expressed as monetary values. These unit effects are estimates of a society's willingness to pay to keep from causing damage to the five safeguard areas. A final environmental load unit (ELU) is calculated and then equated to a currency (e.g. 1 ELU = 1 ECU).

Lastly, Eco-Indicator 95[26] was used. It assigns LCI values to impact categories, and aggregates the LCI values based on equivalencies. Impacts are then weighted using the distance-to-target method.

The EPS method and Eco-Indicator 95 should be expected to provide similar results, everything else being equal, as Eco-Indicator shares some methodological aspects with EPS. MET, on the other hand, appears to have similar impact categories to Eco-Indicator, but the weighting and assessment methods are different.

The Ford engineers then described two sets of vehicles for comparison:

- a spark-ignited vehicle getting 28.3 mpg (miles per gallon) and three compression-ignited vehicles with efficiencies ranging from 58–68 mpg (Table 3A); and
- an electric vehicle versus an internal-combustion vehicle (Table 3B).

Table 3A. Spark-Ignited Vehicle (SIV) vs. Compression-Ignited Vehicle (CIV)

	SIV 1	CIV 1	CIV 4 *	CIV 5 *
Eco-Indicator	280,262	140,493	126,474	123,326
EPS	12,053	7,550	7,133	6,900
MET	608	415	470	395

Table 3B. Electric Vehicle (EV) vs. Internal Combustion Vehicle Spark Ignited (ICV SI)

	EV	ICV SI
Eco-Indicator	206,914	233,406
EPS	13,911	9,930
MET	1,488	449

In the first set of comparisons (3A), the single indicators Eco-Indicator 95 and EPS show the same trend: the SIV is the worst and CIV 5 the best, while CIV 4 is better than CIV 1. MET also shows the SIV as the worst and CIV 5 as the best, but shows that CIV 1 is better than CIV 4.

In the second set of comparisons (3B), the Eco-Indicator model rates the EV as the better vehicle, while EPS and MET suggest the opposite.

Unless more detailed information is available, it is difficult to see how decision makers or policy makers would make use of this information. Depending on the indicator model used, the choices could go in any direction. Furthermore, there is no way to know what factors caused the differences. Were they material differences, process differences, fuel efficiency differences, or simply differences brought about by the different factors used in weighting the results?

Just as difficult to understand is how this information would actually be used to make design decisions. A single index score does not produce what many designers need, which is a breakdown of information to see where improvements can be made. Designers may go back through the data to see if specific changes in materials or processes would produce a more favourable outcome. However, this would be of little benefit since the same model would then be used to test the new assertions. Whether the designer chooses a single index score, or a more detailed analysis, all the calculations performed would still be dependent on the biases built into normalization and weighting. Without standard methods available for normalization and weighting across all environmental parameters, the results could be misleading.

8. CONCLUSION: IS A BETTER ECO-INDEX NEEDED?

There is no question that eco-indices will continue to be developed and used. Work pressures and other constraints provide incentives to simplify the requirements of a scientific and complete environmental analysis. As a result, objectives will continue to be incorrectly defined (systems versus products), information inappropriately applied (economic data used to describe the environmental impacts of physical production systems), and data improperly handled (aggregating beyond recognized scientific standards).

The question that sparked the initiation of this study was whether criteria for eco-efficiency indicators often ignore specific characteristics of metals, thus resulting in poor scores for metals and creating a disincentive for the use of metals in product design.

The answer to this question is yes. The specific characteristics of metals (e.g. recyclability, complexation, bioavailability, transformation) are inadequately modelled by eco-indicator models. This problem is in fact common to all types of eco-modelling approaches and reflects a lack of information and knowledge by both developers and users of eco-models. For example, the modelling of the impact of metals is generally assumed to be similar to that of organics.

The scientific understanding and documentation of the characteristics of metals have expanded significantly over the last few years. The science panels of a number of international organizations, including the Organisation for Economic Co-operation and Development, have now formally acknowledged the specific environmental behaviour of metals and the need to reflect this specificity in impact assessments. Professionals and scientists interested in eco-modelling, as well as their professional associations, have a stake in the quality of information provided by eco-models. They therefore need to be made aware of this new information on metals so as to integrate it, as needed, into their tools.

Even if the characteristics of metals were better known and embedded in the models, problems would still exist with eco-indicator models. They would still generate unreliable information on the impact of metals (or of any other material for that matter) because the fundamental flaws inherent in the development of eco-indices would still remain.

By definition, an eco-index is designed to take numerous sets of data from a variety of environmental impact categories and compress them into a single value to summarize the relative impact of a product. It is difficult to see how any model could be developed to accomplish this task, provide significant information to users, and still meet the tenets of scientific validity and objectivity.

With evidence that specific types of materials may be regularly discriminated against by one model or another, it seems prudent to forego the use of any single-indicator model. Designers and decision makers will need to understand and use more standardized and accepted modelling methods such as LCA without trying to compress or aggregate the data into arbitrary sets of information. Proper studies can be conducted cost-effectively by clearly defining objectives and properly bounding the assessment.

APPENDIX A.
PARTIAL LIST OF ECO-DESIGN TOOLS[27]

LCA/LCI Tools

Boustead
CALA
CUMPAN
Computergestutzte Umweltorientierte Produkt Analyse
ECO-it
EcoManager
EcoPro 1.4
ECO-SCAN 1.0
EDIP LCV tool
GaBi
IDEMAT
JEM-LCA
KCL-ECO
LCAdvantage
LCAiT
LMS Eco-Inv. Tool
NIRE-LCA 2
Öko-Base 2.5 for Windows
Paradox
PEMS 4
REPAQ
SimaPro
TEAM (DEAM)
Umberto

DFX Tools

AMETIDE
Design for Environment (DFE) software tool & DFMA tool
DFR-Recy
DFD/DFR
LASER, Life cycle Assembly Serviceability and Recycling prototype program
PRICE
ReStar

Non-Software Tools

Eco-Indicator manual
Eco-Design sessions
IVF Handbook
Promise Manual (NL)
Promise Manual (UK). *Ecodesign: a promising approach to sustainable production and consumption*
Product Improvement Matrix
Environment compatible products
Design for the Environment (DFE)
Life Cycle Design
DIANA
ECONTROL
EcoPack2000
Ecosys
EcoReDesign program
Green Design Advisor
Green TV
Materials with disposal restriction list
EDIT
Environment Impact Assessment (EIA)

(Non-Software Tools cont'd.)

EMIS

Envision

EPS

Green Manufacturing Shell

Heraklit

IBIS

IDEA (IIASA International Institute for Applied System Analysis, Austria)

IMSS

Koning

LCASys

LCI Procter & Gamble

LIMS

Manufacturing Advisor

PIA

SimBox

TEMIS

TetraSolver

Umcon

PUISSÖCOS

PLA

SEER

SimaTool

Pollution Prevention or Waste Prevention Tools

AWARE
Cage
Clean Process Advisory System (CPAS)
EcoTox
P2-EDGE
SAGE
SWAMI

APPENDIX B.
OVERVIEW OF LIFE CYCLE IMPACT ASSESSMENT MODELS
(Indicator classification, characterization and weighting)

1. SETAC

In the SETAC document *Towards a Methodology for Life Cycle Impact Assessment,*[19] impact assessment is defined as being a quantitative and/or qualitative process to characterize the effect of the environmental interventions identified in the inventory table. The impact assessment component consists in principle of the following three or four elements: classification, characterization, normalization, valuation (i.e. weighting).[28] The following definitions also appear in the same SETAC document.

1.1 Classification

First element within impact assessment, which attributes the environmental interventions listed in the inventory table to a number of selected impact categories.

Note: Environmental interventions contributing to more than one impact category are listed more than once.

Note: The selection of relevant impact categories can be made either in the classification prior to the attribution, of course in line with the goal and scope definition of the study under consideration, or it can be made in the goal and scope definition itself. The current tendency in ISO is to split these two elements into two distinct steps: definition and classification.

1.2 Characterization

Second element within impact assessment succeeding the classification element and preceding valuation, in which analysis/quantification and aggregation of the impacts within the chosen impact categories take place. This element results in impact scores of the impact score profile.

1.3 Normalization

An optional element within impact assessment which is relating all impact scores of a functional unit in the impact score profile to a reference situation. The reference situation may differ per impact category, and is the contribution of a certain region in a certain period of time to the problem type at hand. Normalization results in a normalized impact score profile, which consists of normalized impact scores.

1.4 Valuation

Last element within impact assessment following the characterization/normalization element, in which the results of the characterization/normalization, in particular the (normalized) impact scores, are weighted against each other in a quantitative and/or qualitative way in order to be able to make the impact information more decision friendly. This is an element that necessarily involves qualitative or quantitative valuations, which are not based only on natural sciences. For instance, political and/or ethical values can be used in this element. The valuation can result in an environmental index.

2. Impact Categories

There is no international consensus on impact categories or on methods of treating them. A preliminary list was developed by the ISO based on a consensus process involving practitioners from around the world. The list includes:

- Abiotic resources
- Biotic resources
- Land use
- Global warming
- Stratospheric ozone depletion
- Photochemical oxidant creation
- Acidification
- Eutrophication
- Ecotoxicology
- Human toxicology

In addition to this list, SETAC documents published prior to the ISO process also recognize the potential for a few additional impact categories:

- Odour
- Noise
- Radiation
- Casualties

Several authors also proposed a further category of "Work environment."

It must be emphasized that the above-named categories do not all exhibit the same "sphere of influence"—effects can be global, regional or even local, with many of the categories covering more than one spatial domain. Table B-1 shows how SETAC lists the spatial scope of these categories.

Table B-1. Impact Categories Listed by SETAC

Input-related categories	Abiotic resources	Global
	Biotic resources	Global
	Land	Local
Output-related categories	Global warming	Global
	Stratospheric ozone depletion	Global
	Human toxicological	Global/regional/local
	Ecotoxicological	Global/regional/local
	Photochemical-oxidant formation	Regional/local
	Acidification	Regional/local

(continued on next page)

(Table B-1 cont'd.)		
Output-related categories	Eutrophication	Regional/local
	Odour	Local
	Noise	Local
	Radiation	Regional/local
	Casualties	Local

A description of each impact category as well as a proposed method of treatment is given below:

2.1 Abiotic Resources

The term "abiotic resources" covers three subcategories:

- deposit (fossil fuels, mineral ores, gravel, peat, etc.)
- funds (ground water, lakes, soil)
- natural flow resources (air, water, solar radiation and ocean currents)

Proposed methods for determining characterization factors (W) for deposits look at T, the total estimated global amount of the resource, with some linking this to A, the total annual consumption. Suggested methods of determining W include $W=A/T^2$,[19] $W=1/T$[29] and $W=A/T$.[30] The characterization factors are then summed to provide a single aggregated figure.

For a fund with an annual replenishment rate P (same units as A), the characterization factor is given by $W=(A-P)/T^2$.[3,4]

2.2 Biotic Resources

Biotic resources has only one subcategory: funds (flora and fauna). Characterization factors may be calculated by the equation $W=(A-P)/T^2$ and then summed.

2.3 Land Use

Until recently, land use has been largely ignored by LCA methodology. This area is now receiving attention. Several methods have been suggested; it remains to be seen whether they actually contribute any useful information to impact assessment. One idea presented is to regard land use in two different ways: in terms of production of food and materials for humankind on the one hand, and in terms of the implications on biodiversity on the other. Many of the ideas proposed are not based on assumptions or mechanisms that are the subject of rigorous demonstration. An example of this is the idea of comparing the effects of land use to other energy aspects in the life cycle by expressing the solar energy loss due to the land use as the difference in solar energy utilization between the actual land use and the alternative land use with the "highest possible efficiency." The fact that it may be possible to do this does not mean that the exercise serves any purpose.

2.4 Global Warming

Values for global warming potentials for many compounds have been determined by the Intergovernmental Panel on Climate Change (IPCC). These global warming potentials express the warming effect of a compound as an equivalent quantity of CO_2 that would produce the same effect. Global warming potentials (GWP) are expressed for time horizons of 20, 100 and 500 years. Table B-2 below shows the IPCC GWP as published in 1996.

Table B-2. Global Warming Potentials as Determined by the IPCC

Compound	Formula	GWP/20 yrs	GWP/100 yrs	GWP/500 yrs	Lifetime/yrs
Carbon dioxide	CO_2	1	1	1	150
Methane	CH_4	62	25	7.5	10
Nitrogen dioxide	NO_2	290	320	180	120
Tetrachloromethane	CCl_4	2400	1400	500	42
Trichloromethane	$CHCl_3$	15	5	1	0.55
Dichloromethane	CH_2Cl_2	28	9	3	0.41
Chloromethane	CH_3Cl	92	25	9	0.7
111 Trichloromethane	CH_3CCl_3	360	110	35	5.4
Tetrafluoromethane	CF_4	4,100	6,300	9,800	50000
Hexafluoroethane	C_2F_6	8,200	12,500	19,100	10000
CFC-11	$CFCl_3$	5,000	4,000	1,400	50
CFC-12	CF_2Cl_2	7,900	8,500	4,200	102
CFC-13	CF_3Cl	8,100	11,700	13,600	640
CFC-113	$CF_2ClCFCl_2$	5,000	5,000	2,300	85
CFC-114	CF_2ClCF_2Cl	6,900	9,300	8,300	300
CFC-115	CF_2ClCF_3	6200	9,300	13,000	1700
HCFC-22	CHF_2Cl	4,300	1,700	520	13
HCFC-123	CF_3CHCl_2	300	93	29	1.4
HCFC-124	CF_3CHFCl	1,500	480	150	5.9
HCFC-141b	$CFCl_2CH_3$	1,800	630	200	9.4
HCFC-142b	CF_2ClCH_3	4,200	2,000	630	19.5
HCFC-225ca	$C_3F_5HCl_2$	550	170	52	2.5
HCFC-225cb	$C_3F_5HCl_2$	1,700	530	170	6.6
HFC-23	CHF_3	9,200	12,100	9,900	250
HFC-32	CH_2F_2	1,800	580	180	6
HFC-43-10me	$C_5H_2F_{10}$	3,300	1,600	520	21
HFC-125	CF_3CHF_2	4,800	3,200	1,100	36
HFC-134	CHF_2CHF_2	3,100	1,200	370	12
HFC-134a	CH_2FCF_3	3,300	1,300	420	14
HFC-143	CHF_2CH_2F	950	290	90	3.5
HFC-143a	CF_3CH_3	5,200	4,400	1,600	55
HFC-152a	CHF_2CH_3	460	140	44	1.5
HFC-227ea	C_3HF_7	4,500	3,300	1,100	41
HFC-236fa	$C_3H_2F_6$	6,100	8,000	6,600	250
HFC-245ca	$C_3H_3F_5$	1,900	610	190	7
Halon 1301	CF_3Br	6,200	5,600	2,200	65
Sulphur hexafluoride	SF_6	16,500	24,900	36,500	3200
Carbon monoxide	CO	—	—	—	months
Non-methane VOC	—	—	—	—	days/months
Nitrogen oxides	NO_x	—	—	—	days

2.5 Stratospheric Ozone Depletion

The World Meteorological Organization (WMO) has produced a list of ozone depletion potentials (ODP), similar in function to the IPCC's GWP, whereby halogenated compounds are expressed relative to the effect caused by an equal mass of CFC-11 ($CFCl_3$). As with GWPs, ODPs are presented for a series of time horizons. Table B-3 below shows ODPs for CFCs, HCFCs and halons.

Table B-3. Ozone-Depleting Potentials as Determined by WMO

Compound	Formula	ODP/20 yrs	ODP/100 yrs	ODP/500 yrs	Total ODP	Lifetime/yrs
Tetrachloromethane	CCl_4	1.23	1.14	1.08	1.20	42
111	CH_3CCl_3	0.45	0.15	0.12	0.12	5.4 ± 5
Methylbromide	CH_3Br	2.3	0.69	0.57	0.64	1.3
CFC-11	$CFCl_3$	1.00	1.00	1.00	1.00	50 ± 5
CFC-12	CF_2Cl_2	—	—	—	0.82	102
CFC-113	$CF_2ClCFCl_2$	0.59	0.78	1.09	0.90	85
CFC-114	CF_2ClCF_2Cl	—	—	—	0.85	300
CFC-115	CF_2ClCF_3	—	—	—	0.40	1700
HCFC-22	CHF_2Cl	0.14	0.07	0.05	0.04	13.3
HCFC-123	CF_3CHCl_2	0.08	0.03	0.02	0.014	1.4
HCFC-124	CF_3CHFCl	0.08	0.03	0.02	0.03	5.9
HCFC-141b	$CFCl_2CH_3$	0.33	0.13	0.11	0.10	9.4
HCFC-142b	CF_2ClCH_3	0.14	0.08	0.07	0.05	19.5
HCFC-225ca	$C_3F_5HCl_2$	0.10	0.03	0.02	0.02	2.5
HCFC-225cb	$C_3F_5HCl_2$	0.11	0.0	0.03	0.02	6.6
Halon 1301	CF_3Br	10.5	11.5	12.5	12	65
Halon 1211	CF_2ClBr	9.0	4.9	4.1	5.1	20
Halon 1202	CF_2Br_2	11.0	7.0	5.9	~1.25	
Halon 2402	CF_2BrCF_2Br	—	—	—	~7	25
HBFC 1201	CF_2HBr	—	—	—	~1.4	
HBFC 2401	CF_3CHFBr	—	—	—	~0.25	
HBFC 2311	$CF_3CHClBr$	—	—	—	~0.14	

2.6 Photochemical Ozone Creation Potentials

Photochemical ozone creation potentials (POCPs) are expressed as ethylene (C_2H_4) equivalents (i.e. they are listed as the equivalent mass of ethylene that causes the same effect as one kilogram of the respective organic compound). POCPs can, however, be determined in different ways, as is evident by the approaches of Andersson-Sköld et al.[31] and Heijungs[24] (Table B-4). In the former approach, the authors provide POCPs for three scenarios (maximum differences in concentration, ordinary Swedish background concentration of NO_X in a 0–4 day period, high NO_X background period in a 0–4 day period); in the latter, POCPs are determined as the contribution to ozone formation during peak ozone formation based on average data from three different locations in Europe.

Table B-4. Photochemical Ozone Creation Potentials Expressed as Ethylene Equivalents					
	Andersson-Sköld et al.[31]			Heijungs et al.[32]	
Compound	**Max. difference in concentration**	**Ordinary Swedish background during 0–4 days**	**High NO_x background during 0–4 days**	**Average for three European sites**	**Range**
Alkanes					
Methane	—	—	—	0.007	0.000–0.030
Ethane	0.173	0.126	0.121	0.082	0.020–0.300
Propane	0.604	0.503	0.518	0.420	0.016–1.200
n-Butane	0.554	0.467	0.485	0.410	0.150–1.150
i-Butane	0.331	0.411	0.389	0.315	0.190–0.590
n-Pentane	0.612	0.298	0.387	0.408	0.090–1.050
i-Pentane	0.360	0.314	0.345	0.296	0.120–0.680
n-Hexane	0.784	0.452	0.495	0.421	0.100–1.510
2-Methylpentane	0.712	0.529	0.565	0.524	0.190–1.400
3-Methylpentane	0.647	0.409	0.457	0.431	0.110–1.250
2,2-Dimethylbutane	—	—	—	0.251	0.120–0.490
2,3-Dimethylbutane	—	—	—	0.384	0.250–0.650
n-Heptane	0.791	0.518	0.592	0.529	0.130-–1.650
2-Methylhexane	—	—	—	0.492	0.110–1.590
3-Methylhexane	—	—	—	0.492	0.110–1.570
n-Octane	0.698	0.461	0.544	0.493	0.120–1.510
2-Methylheptane	0.691	0.457	0.524	0.469	0.120–1.460
n-Nonane	0.633	0.351	0.463	0.469	0.100–1.480
2-Methyloctane	0.669	0.454	0.523	0.505	0.120–1.470
n-Decane	0.719	0.422	0.509	0.464	0.080–1.560
2-Methylnonane	0.719	0.423	0.498	0.448	0.080–1.530
n-Undecane	0.662	0.386	0.476	0.436	0.080–1.440
n-Duodecane	0.576	0.311	0.452	0.412	0.080–1.380
Average	—	—	—	*0.398*	*0.114–1.173*
Halogenated HCs					
Methylene chloride	0.000	0.000	0.000	0.010	0.000–0.030
Chloroform	0.007	0.004	0.003	—	—
Methylchloroform	0.007	0.002	0.001	0.001	0.000–0.010
Trichloroethylene	0.086	0.111	0.091	0.066	0.010–0.130
Tetrachloroethylene	0.014	0.014	0.010	0.005	0.000–0.020
Allyl chloride	0.561	0.483	0.667	—	—
Average	—	—	—	*0.021*	*0.003–0.048*
Alcohols					
Methanol	0.165	0.213	0.178	0.123	0.090–0.210
Ethanol	0.446	0.225	0.317	0.268	0.040–0.890
i-Propanol	0.173	0.203	0.188	—	—
Butanol	0.655	0.214	0.404	—	—
i-Butanol	0.388	0.255	0.29	—	—
Butane-2-diol	0.288	0.066	0.216	—	—
Average	—	—	—	*0.196*	*0.065–0.550*
Aldehydes					
Formaldehyde	0.424	0.261	0.379	0.421	0.220–0.580
Acetylaldehyde	0.532	0.186	0.615	0.527	0.330–1.220
Propionaldehyde	0.655	0.170	0.652	0.603	0.280–1.600
Butyraldehyde	0.640	0.171	0.597	0.568	0.160–1.600
					(Cont'd.)

(Table B-4 cont'd.)					
	Andersson-Sköld et al.[31]			**Heijungs et al.[32]**	
Compound	**Max. difference in concentration**	**Ordinary Swedish background during 0–4 days**	**High NO_x background during 0–4 days**	**Average for three European sites**	**Range**
i-Butyraldehyde	0.583	0.300	0.677	0.631	0.380–1.280
Valeraldehyde	0.612	0.321	0.686	0.686	0.000–2.680
Acroleine	1.201	0.832	0.827	—	—
Benzaldehyde	—	—	—	-0.33	(-0.82)-(-0.12)
Average	—	—	—	*0.443*	*0.079–1.263*
Ketones					
Acetone	0.173	0.124	0.160	0.178	0.100–0.270
Methyl ethyl ketone	0.388	0.178	0.346	0.473	0.170–0.800
Methyl *i*-butyl ketone	0.676	0.318	0.666	—	—
Average	—	—	—	*0.326*	*0.135–0.535*
Esters					
Dimethylester	0.058	0.067	0.046	—	—
Methyl acrylate	—	—	—	0.025	0.000–0.070
Ethyl acetate	0.295	0.294	0.286	0.218	0.110–0.560
i-Propyl acetate	—	—	—	0.215	0.140–0.360
n-Butyl acetate	0.439	0.320	0.367	0.323	0.140–0.910
i-Butyl acetate	0.288	0.353	0.345	0.332	0.210–0.590
Average	—	—	—	*0.223*	*0.120–0.498*
Olefins					
Ethene	1.000	1.000	1.000	1.000	1
Propene	0.734	0.599	1.060	1.030	0.750–1.630
1-Butene	0.799	0.495	0.983	0.959	0.570–1.850
2-Butene	0.784	0.436	1.021	0.992	0.820–1.570
1-Pentene	0.727	0.424	0.833	1.059	0.400–2.880
2-Pentene	0.770	0.381	0.965	0.930	0.650–1.600
2-Methyl-1-butene	0.691	0.181	0.717	0.777	0.520–1.130
2-Methyl-2-butene	0.935	0.453	0.784	0.779	0.610–1.020
3-Methyl-1-butene	—	—	—	0.895	0.600–1.540
Isobutene	0.791	0.580	0.648	0.634	0.580–0.760
Average	—	—	—	*0.906*	*0.650–1.498*
Acetylenes					
Acetylene	0.273	0.368	0.291	0.168	0.100–0.420
Aromatics					
Benzene	0.317	0.402	0.318	0.189	0.110–0.450
Toluene	0.446	0.470	0.565	0.563	0.410–0.830
o-Xylene	0.424	0.167	0.598	0.666	0.410–0.970
m-Xylene	0.583	0.474	0.884	0.993	0.780–1.350
p-Xylene	0.612	0.472	0.796	0.888	0.630–1.800
Ethylbenzene	0.532	0.504	0.621	0.593	0.350–1.140
1,2,3-Trimethylbenzene	0.698	0.292	0.868	1.170	0.760–1.750
1,2,4-Trimethylbenzene	0.683	0.330	0.938	1.200	0.860–1.760
1,3,5-Trimethylbenzene	0.691	0.330	0.989	1.150	0.740–1.740
					(Con'td.)

(Table B-4 cont'd.)					
	Andersson-Sköld et al.[31]			Heijungs et al.[32]	
Compound	**Max. difference in concentration**	**Ordinary Swedish background during 0–4 days**	**High N0$_x$ background during 0–4 days**	**Average for three European sites**	**Range**
o-Ethyltoluene	0.597	0.408	0.637	0.668	0.310–1.300
m-Ethyltoluene	0.626	0.401	0.729	0.794	0.410–1.400
p-Ethylotoluene	0.626	0.443	0.682	0.725	0.360–1.350
n-Propylbenzene	0.511	0.454	0.531	0.492	0.250–1.100
i-Propylbenzene	0.511	0.523	0.594	0.565	0.350–1.050
Average	—	—	—	*0.761*	*0.481–1.258*
Other					
Methylcyclohexane	0.403	0.386	0.392	—	—
Isoprene	0.532	0.583	0.768	—	—
Dimethylether	0.288	0.343	0.286	—	—
Propylene glycol m.e.	0.770	0.491	0.497	—	—
Propylene glycol m.e.a.	0.309	0.157	0.143	—	—
Carbon monoxide	0.036	0.040	0.032	—	—

It should be noted that in many cases the exact nature of volatile organic compounds (VOCs) that are released is unknown, in which case calculating an overall POCP is not possible. In this case, the suggestion has been made to aggregate data according to the subcategories: oxides of nitrogen; hydrocarbons or VOCs; carbon monoxide; methane.

2.7 Acidification

Acidification potentials (AP) are expressed as SO_2 equivalents or alternatively as moles of H^+. Table B-5 below lists acidification potentials for several acidifying substances.[5]

Table B-5. Acidification Potentials Expressed as SO_2 Equivalents

Compound	Formula	Reaction	m.mass g/mole	AP kg SO_2/kg
Sulphur dioxide	SO_2	$SO_2 + H_2O$ ® H_2SO_3 ® $2H^+ + SO_3^{2-}$	64.06	1
Sulphur trioxide	SO_3	$SO_3 + H_2O$ ® H_2SO_4 ® $2H^+ + SO_4^{2-}$	80.06	0.8
Nitrogen dioxide	NO_2	$NO_2 + ½H_2O + ¼O_2$ ® $H^+ + NO_3H^-$	46.01	0.7
Nitrogen oxide	NO	$NO + ½H_2O + O_3$ ® $H^+ + NO_3^- + ¾O_2$	30.01	1.07
Hydrogen chloride	HCl	HCl ® $H^+ + Cl^-$	36.46	0.88
Hydrogen nitrate	HNO_3	HNO_3 ® $H^+ + NO_3^-$	63.01	0.51
Hydrogen sulphate	H_2SO_4	H_2SO_4 ® $2H^+ + SO_4^{2-}$	98.07	0.65
Hydrogen phosphate	H_3PO_4	H_3PO_4 ® $3H^+ + PO_4^{3-}$	98.00	0.98
Hydrogen fluoride	HF	HF ® $H^+ + F^-$	20.01	1.6
Ammonia	NH_3	$NH_3 + H_2O$ ® $NH_4^+ + OH^-$	17.03	1.88

Example calculation: 1 mole NO releases same quantity of H^+ as ½ mole SO_2, thus 1 kg NO has same effect as ½×mol. mass SO_2÷mol. mass NO = ½×64.06÷30.01 = 1.07 kg SO_2.

It is unclear whether acidification potentials are applied to emissions to water or whether they are applied to airborne emissions only: undoubtedly, releases to water should be accounted for.

2.8 Eutrophication

Lindfors et al.[21] suggest two methods of determining eutrophication potentials (EP). The first method separates emissions into the following four categories: organic material released to water (measured as Biological Oxygen Demand [BOD_5]; total nitrogen released to water (as kg N); total phosphorus released to water (as kg P); total nitrogen released to air (as kg N). The second method divides eutrophication into two subcategories: aquatic ecosystems and terrestrial (aquatic) ecosystems, respectively. The stated reason behind this second approach is to try to take account of the conditions into which the releases occur, as in some circumstances the release of phosphorus is taken to be the limiting factor, and in others it is the release of nitrogen. In this second method, each of the two subcategories is then further split into the following five sectors: total nitrogen released to air (terrestrial effects); total phosphorous emissions and the emission of organic material to water; total nitrogen emissions and the emission of organic material to water; total nitrogen emissions and the emission of organic material to water summed with total nitrogen released to air; total phosphorus, nitrogen and organic matter releases (air and water). Eutrophication potentials for this second method appear in Table B-6 below.

Table B-6. Eutrophication Potentials

Compound	N to air kg O_2 /kg	P-limited kg O_2 /kg	N-limited kg O_2 /kg	N-limited + N to air kg O_2 /kg	Maximum kg O_2 /kg	Maximum kg PO_4 eq. /kg
N to air	20	0	0	20	20	0.42
NO_X to air	6	0	0	6	6	0.13
NH_3 to air	16	0	0	16	16	0.35
N to water	0	0	20	20	20	0.42
NO_3^- to water	0	0	4.4	4.4	4.4	0.1
NH_4^- to water	0	0	15	15	15	0.33
P to water	0	140	0	0	140	3.06
PO_4^{3-} to water	0	46	0	0	46	1
COD	0	1	1	1	1	0.022

Eutrophication potentials have also been calculated[21] in terms of total nitrogen, total phosphorus and total nitrate emissions, as shown in Table B-7 below.

Table B-7. Eutrophication Potentials in Terms of Total Nitrogen, Total Phosphorus, and Total Nitrate

Compound	M_m g/mole	EP(N) kg N /kg	EP(P) kg P /kg	EP kg NO_3 /kg
NO_3^-	62	0.23	0	1
NO_2	46	0.30	0	1.35
NO_2^-	46	0.30	0	1.35
NO	30	0.47	0	2.07
NH_3	17	0.82	0	3.64
CN^-	26	0.54	0	2.38
Total N	14	1	0	4.43
PO_4^{3-}	95	0	0.33	10.45
$P_2O_7^{2-}$	174	0	0.35	11.41
Total P	31	0	1	32.03

Although the increased supply of surplus nitrogen, phosphorus and dead organic matter can undoubtedly lead to algal blooms in water, there seems to be little information of the effect of, for example, acidifying substances on the eutrophication process. While increased acidity in itself is probably undesirable, would it have a countering effect for eutrophication?

2.9 Ecotoxicology

A number of different methods addressing chemical fate, route of exposure and toxicological effect have been developed, as summarized in Table B-8 below.

Table B-8. Various Methods Used in Assessing Ecotoxicological Impacts

Method	Effects concerned	Criteria/comments	Reference
Quantitative approach with partial fate analysis based on EU directives	Acute toxicity Acute toxicity for not readily degradable compounds Potential bioconcentration Potential bioconcentration for not readily degradable compounds	EU criteria for degradability	Lindfors et al.[21] Finnveden et al.[33]
MUP-method	Acute toxicity Potential bioconcentration Biodegradability	EU criteria for classification of substances as dangerous for the environment	Jensen et al.[34]
Quantitative approach with partial fate analysis	Acute, aquatic toxicity Chronic, aquatic toxicity Chronic, terrestrial toxicity Acute toxicity to wastewater treatment plants	Critical volume. The fate analysis includes evaporation, deposition and degradation. The ecotoxicity values are based on PEC (Predicted Environmental Concentration) for acute, chronic aquatic and terrestrial toxicity, and LOEC for micro-organisms in wastewater treatment plants	Hauschild et al.[35]
The “ecotoxicity potential approach”	Terrestrial ecotoxicity Aquatic ecotoxicity		Guineé & Heijungs[36]
The “provisional method”	Terrestrial ecotoxicity Aquatic ecotoxicity	The provisional classification factors for ecotoxicity are derived from NOEC or LC_{50} multiplied by a safety factor. The output is the PEC/PNEC (Predicted No Effect Concentration Level)	Heijungs et al.[30]

International consensus on specific methods for assessing ecotoxicological impacts has not yet been reached and development of some of the methods is still in progress. It is therefore recommended that different methods be used when assessing potential ecotoxicological impacts for a specific data set.

2.10 Human Toxicology

This is another impact category which is extremely difficult to address. Several methods have been devised which attempt to account for chemical fate, route of exposure and toxicological impact. The present methods are described briefly in Table B-9 below.

Table B-9. Some Examples of Methods Used for Calculating Human Toxicological Impacts

Method	Effects concerned	Criteria/comments	Reference
Critical volumes	Water and air pollution	The critical air volumes are based on MIK (Maximale Immissions-Konzentration) or MAK (Maximale Arbeitsplatzkonzentration) values. The critical water volumes are based on Swiss directives for emissions to surface water	BUS[37]/ Habbersatter[38]
The "provisional method"	Human toxicity (not specified), exposure by air, water and soil	The provisional classification factors for human toxicity are derived from TCL (acceptable concentration in air), AQG (air quality guideline), TDI (tolerable daily intake) or ADI (acceptable daily intake). The classification factors are expressed by kg body mass/kg substance	Heijungs et al.[32]
The Tellus method	Carcinogenic potency Non-carcinogenic effects Combined	Classification factors for carcinogenic potency are expressed as "isophorone equivalents" and for non-carcinogenic effects as "xylene equivalents" Classification factors for the combined effects are derived from permissible exposure levels for the two effects.	Tellus Institute[39]
The MUP-method	Irritation, allergenic reactions, organotoxicity, genotoxicity, carcinogenicity, neurotoxicity, teratogenicity	This method is based on exposure estimated. The method includes a screening LCA process with qualitative results	Jelnes et al.[40] (Danish paper)/ Schmidt et al.[41]
			(Cont'd. next page)

(Table B-9 cont'd.)

Method	Effects concerned	Criteria/comments	Reference
The "toxicity potential approach"			Guineé & Heijungs[36]
The "critical surface-time"			Jolliet[42]/Jolliet & Crettaz[43]
Quantitative approach with partial fate analysis	Acute toxicity (inhalation) Acute toxicity (oral intake)	Critical volume. The fate analysis includes evaporation, deposition and degradation. The human toxicology factors are based on LC_{50}/LD_{50} LC_{Lo}/LD_{Lo} (lethal concentration/dose low), or lowest observed adverse effect level (LOAEL), on animals and humans	Hauschild et al.[35]

As with ecotoxicological impacts, international consensus on specific methods for assessing toxicological impacts has not yet been reached and development of some of the methods is still in progress. It is therefore recommended that different methods be used when assessing potential toxicological impacts for a specific data set.

2.11 Work Environment

This category covers the same effects as human toxicological impacts, but also includes non-chemical effects, such as hearing impairment, psychological damage and muscular/joint pain. Almost all of the work in this area has taken place in the Scandinavian countries.

2.12 Summary

In all of the various equivalency methods described above, each of the potential values listed assumes that all of the substance emitted contributes to the particular effect. While this is good in that it lists the maximum possible effect, there is a serious flaw when these effects are grouped together via a weighting procedure into a single number (or a few numbers). The problem arises when an individual emission may contribute to more than one impact category. For example, NO_X air emissions can potentially contribute to global warming (GWP), acidification (AP) and eutrophication (EP), but not at all maximum values at the same time. While each impact category states the maximum effect (with the exception of NO_X/NH_3 for acidification) of an emission with regard to that category, the emissions cannot all have the maximum values in all categories simultaneously. Thus, depending on the number of categories aggregated, the distortion can be threefold, fourfold or higher.

3. Weighting Approaches

While there may be some consensus regarding the treatment of impact categories, the same cannot be said of weighting procedures. Weighting is a qualitative or quantitative step that is not necessarily based on natural science: often, political or ethical values are also considered. The term "valuation" had previously been applied to the weighting procedure; however, in ISO/CD 14042 (1997.01.15) "weighting" is the adopted name. An overview of the various weighting methods employed up to and including 1996 appears in Table B-10 below.[44]

Table B-10. Selected Examples of Weighting Methods Used in Life Cycle Impact Analysis

Weighting method indication	Year	Organization	Equivalency principle (when are impacts or systems considered environmentally equivalent?)	Characteristics
Energy requirement		Franklin, USA	Equal energy requirement	Proxy
Material Intensity Per Service unit	1994	Wuppertal Institute, D	Equal material displacement	Proxy
Sustainable Process Index		Technical University of Graz, A	Equal space consumption	Proxy Technology
Abatement energy	1993	TNO MEP, NL	Equal total energy requirement, including energy for abatement of environmental burden	Technology
Abatement costs	1994	ECN, NL	Equal modelled costs for abating emissions according to national goals	Technology Monetarization Authorized targets
Abatement costs	1992	Tellus Institute, USA	Equal costs for abating emissions, with most human toxic emissions abatement costs extrapolated from characterization factors via lead (combining carcinogenic/non-carcinogenic substances via permitted exposure level values)	Monetarization Authorized standards
Decision Model for Environmental Strategies of corporations	1992	TME, NL	Equal projected generic costs for abatement of burden according to national goals, derived per impact category	Technology Monetarization Authorized targets
Environmental Priority Strategies system	1992	IVL, S	Equal willingness to pay for restoring biodiversity, production, human health, resources and esthetic values after changes due to the system	Monetarization TechnologyMole fraction
Mole fraction	1991	University of Basle, CH	Equal critical volume scores, with the volume of each medium weighted according to their mole density	Authorized standards
Critical volume	1992	EPF, D	Equal critical volume scores, weighted subjectively	Authorized standardsCritical surface time
Critical surface time	1994	EPFL, CH	Equal critical emission scores, weighted subjectively	Authorized standards
Ecoscarcity	1990	BUWAL, CH	Equal scores of over proportional distances to political targets	Authorized standards

(Cont'd. next page)

(Table B-10 cont'd.)

Weighting method indication	Year	Organization	Equivalency principle (when are impacts or systems considered environmentally equivalent?)	Characteristics
Distance to target	1994	VROM, NL	Equal scores of distances to political targets, optionally additionally weighted subjectively	Authorized targets
No Significant Adverse Effect Level	1994	CE, NL	Equal scores of overshoots of sustainable targets, optionally weighted subjectively	Authorized targets
Eco-indicator 95	1995	Pre, NL	Equal scores of distances to science-political targets contributing to the equally weighted safeguard subjects 1 on a million lives, 95% of ecosystems and human health complaints due to smog	Authorized targets
Ecoscarcity Iso-utility functions	1994	TNO STB, NL	Equal panel scores on relative (negative) utilities of actual impact scores	Panel
Iso-preferences approach	1994	CML, NL	Equal panel preferences for elasticities in relative impact scenarios	Panel
Delphi technique	1994	Landbank, GB	Equal expert panel scores on actual impacts	Panel
Questionnaire	1995	Waseda University	Equal industry/science panel scores on actual impacts	Panel
Panel questionnaire	1994	IVAM ER UvA, NL	Equal societal group panel scores on actual impacts	Panel
Structured dialogue	1994	PI DTU, DK	Panel agreement on weights based on argumentation	Panel
Argumentative evaluation	1994	UBA, D	Societal group consensus on the interpretation of product systems comparison, with inputs from normalization, environmental problem weights by a political panel and sensitivity analysis	Panel
Expert panel prioritization	1996	CAU, D	Qualitative valuation of normalization data and expert panel scores on time, space and hazard	Panel

APPENDIX C. HISTORICAL DEVELOPMENT OF LIFE CYCLE INVENTORY CALCULATIONS

At first, most industrial decisions were based on economics and tempered by social and political considerations. Environmental considerations were usually limited to compliance with local and national pollution regulations. Little attention was paid to environmental aspects of entire systems. With the emergence of a green movement, and the beginning of the world modelling exercises,[45–47] companies became aware of how little they really knew about the wider environmental implications of the processes they operated. This awareness, coupled with legislative moves to control the way certain industries could operate,[48] provided an impetus for examining the wider implications of industrial operations. One of the initial and most notable exercises was that carried out by Harold Smith at ICI.[49]

It was quickly discovered that comparing the energy required to manufacture different items missed a number of significant impacts and therefore was no indication of the overall effect of the various production, use and disposal systems. The result was a need to develop a method to examine extended systems stretching from the extraction of raw materials from the earth through to the final disposal of materials back into the earth.

The first systematic procedures for carrying out this work were published in 1979.[28] The treatise mainly focused on energy analyses, but by necessity also discussed the flow of materials as they relate to energy production. A clear distinction was made between industrial systems and products. One of the key problems associated with LCIs today is the misunderstanding of this distinction. Most systems produce multiple products, with the life cycle burdens associated with the entire system. When users attempt to assign burdens to individual products in multi-product systems, they have to make weighting decisions on assigning burdens. These decisions are not scientifically based, but are judgments made by the user to facilitate specific calculations—burdens associated with one product and not the system.

In the 1980s, the technique was extended to air emissions, emissions to water and a more detailed treatment of solid wastes. The calculation method employed was identical to that which had been used for fuel resources, so that the inventory methods were the same and presented no new problems.

However, the new parameters included in the analysis are not all global in scope. Some (such as toxics) were highly localized. Others (such as SO_X, which causes acid rain) were regional in effect. Yet others such as CO_2 (a greenhouse gas) were global in effect. As a result, it was no longer possible to calculate aggregated results for all parameters as it had become extremely doubtful that aggregated data for some parameters (e.g. local water emissions) would yield a result that had any physical significance.

In 1990, the first conference of the Society of Environmental Toxicology and Chemistry (SETAC)[20] took place in Vermont, USA. As a toxicological organization, SETAC placed emphasis on the air and water emissions of industrial systems rather than on energy and materials flows, which had been the predominant drivers up to that time. The main effect was the introduction of the concept of *life cycle assessment*. Now, instead of calculating inventories alone, the concept was extended to include impact assessment calculations and methods for combining data on disparate environmental impacts for the purpose of comparing and prioritizing the impacts.

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William Dove gained a Ph.D. in Physics at the Open University, United Kingdom, in 1992, where he was also a part-time tutor for the Open University's T234/T334 Environmental Pollution and Public Health courses. After a successful post-doctoral research post at the University of Kent, he left to join Dr. Ian Boustead in 1992. Since that time he has been responsible for software development and support within Boustead Consulting Limited and has also conducted life cycle inventory or ecoprofile studies for the following materials and products: packaging steel (including container production and filling), aerosol containers, plastic and glass yogurt containers, primary zinc production, aluminum sulphate production and primary copper production.

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Mr. Yaros is a partner in Boustead Consulting & Associates in the United States. As a life cycle engineer and practitioner for the past 12 years, Mr. Yaros has conducted LCA studies on a wide variety of processes, from chemical manufacturing to buildings and construction. As one of the leading engineering experts in LCA, Mr. Yaros has participated in SETAC conferences and workshops to help formulate and codify the procedures for conducting technically competent LCAs. Prior to joining BCAL, Mr. Yaros was a lead engineer at Scott Paper, working on technology management and innovation. At Scott Paper, Mr. Yaros became one of the first industrial users of LCA software in the United States, utilizing LCA techniques to foster continuous environmental improvements in paper manufacturing.