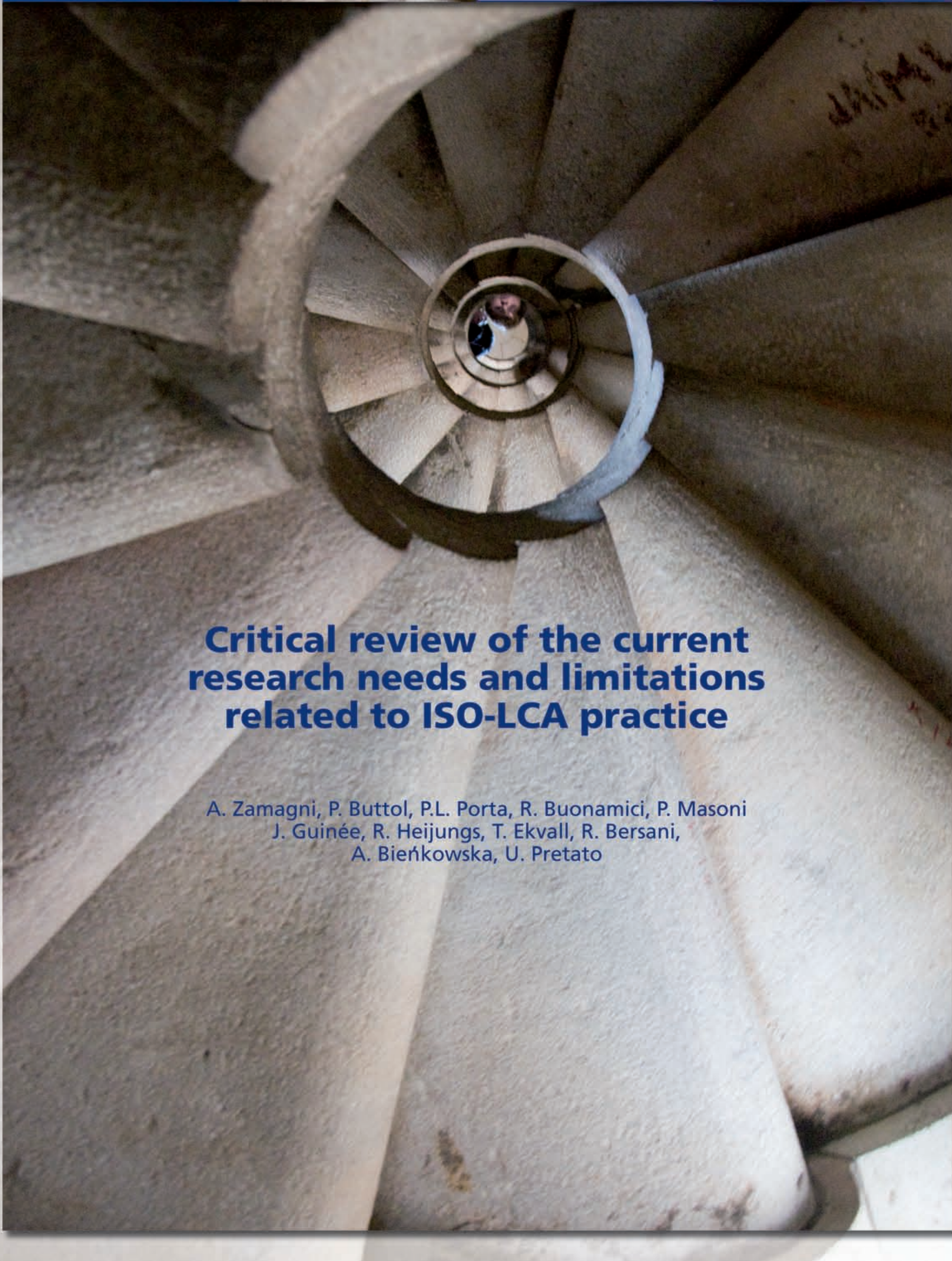
Italian National Agency for New Technologies, Energy and the Environment

Calcas

Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability



Critical review of the current research needs and limitations related to ISO-LCA practice

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A. ZAMAGNI, P. BUTTOL, P.L. PORTA, R. BUONAMICI, P. MASONI,

J. GUINÉE, R. HEIJUNGS, T. EKVALL,

R. BERSANI, A. BIEŃKOWSKA, U. PRETATO

Authors and membership

A. Zamagni, P. Buttol, P.L. Porta, R. Buonamici, P. Masoni

Italian National Agency for New Technologies, Energy and the Environment (ENEA)

J. Guinée, R. Heijungs

Institute of Environmental Sciences, Leiden University (CML)

T. Ekvall

Swedish Environmental Research Institute (IVL)

R. Bersani, A. Bieńkowska and U. Pretato

EC, Directorate General Joint Research Centre (JRC),

Institute for Environment and Sustainability

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Foreword

This volume has been produced within CALCAS project, the EU 6th Framework Program **C**o-ordination **A**ction for innovation in **L**ife-**C**ycle **A**nalysis for **S**ustainability.

The partnership includes outstanding research organisations, science promoter, innovation centre, worldwide scientific society: Institute of Environmental Science (CML) of Leiden University, Swedish Environmental Research Institute (IVL), Wuppertal Institute for Climate, Environment and Energy, Institute for Environment and Sustainability of Joint Research Centre (JRC IES), School of Chemical Engineering Analytical Science, University of Manchester, ARMINES of Ecole Nationale Supèrieure des Mines de Paris, Environmental Policy Research Centre of Freie Universität Berlin, Technical Institute of Lisbon, Institute for Ecological Economy Research (IÖW), European Science Foundation, Chemistry Innovation, Society of Environmental Toxicology and Chemistry (SETAC).

CALCAS is aimed at identifying short- mid- and long-term research lines on life cycle analysis approaches in supporting the sustainability decision making process.
Starting point of the project is Life Cycle Assessment (LCA) standardised by ISO 14040 series: indeed, despite the standardisation process has contributed to its broad acceptance and wide use, there are a number of shortcomings which cannot be resolved through standardisation and which are often considered too restrictive, in particular for meso (e.g. waste treatment, industry sectors, etc.) and macro (e.g. complex technological systems) scale applications, where sustainability problems reside.
CALCAS addresses the question which directions the methodology should move to in order to improve reliability, significance and usability of the LCA applications and to perform analyses more suitable for the broader concept of sustainability. This is achieved by crossing the results of analyses both at the science supply (available knowledge, gaps therein and strategies for filling them) and at the demand side (users' needs in all public and private domains). The most interesting and feasible options for expanding LCA will be pointed out in terms of definition of the scientific framework and identification of the main research lines towards a New LCA.

The present report deals with a review of the scientific literature on ISO-LCA, performed in order to identify present limits and opportunities for LCA, in particular: intrinsic limits of ISO-LCA due to assumptions and simplifications; elements in current ISO standards which are not in line with new scientific developments or best practices; new directions for broadening (e.g.: including economic and social aspects, or covering new environmental aspects) and deepening LCA towards the definition of a New LCA (e.g.: including behavioural aspects in the inventory modelling or including more fate and exposure mechanisms in impact assessment).

Bologna, 18 September 2008

Paolo Masoni (ENEA)
CALCAS co-ordinator

EXECUTIVE SUMMARY

1. Goals and method

The critical review on LCA scientific literature performed within CALCAS is intended to make available a comprehensive and rational state of the art of the methodology, in order to identify areas of controversy and to form the basis for the identification of future research in this area. In particular, the new developments, with respect to the ISO standard LCA, are considered along two main directions:

- broadening, e.g. including economic and social aspects, or covering new environmental aspects;
- deepening, e.g. including behavioural aspects in the inventory modelling or including more fate and exposure mechanisms in impact assessment.

The critical review is part of the work package 5 (WP5) “Deepening and broadening of the standardise LCA”. WP5 objective is the identification of scientific trends and action lines, building on the current state-of-the-art, to improve reliability, significance and usability of the applications of standardised LCA. The developing lines necessary for the improvement of reliability, significance and usability of the applications of standardised LCA will be defined in the second part of WP5.

The critical review was organised along four main activities:

a) the identification of the topics to be analysed and the definition of screening criteria for literature selection;
b) the development of an evaluation grid, in order to support the analysis of the scientific literature and to allow for consistency in the analysis performed by different researchers;
c) the analysis itself, including both methodological and practicability aspects (software and databases).
d) the reporting of the results of the analysis.

All in all, more than 40 software and 25 databases have been mapped and about 60 new approaches (in more than 250 papers) with a different degree of maturity have been analysed.

All the work performed will serve as input for WP7 activities, in which the most interesting and feasible options for expanding LCA will be developed, and the main lines of research defined for a New LCA, which is scientifically feasible, relevant for sustainability governance, and practically applicable by stakeholders. These results will be the fruit of the activities of all the WPs, obtained by crossing the results of the other analyses both at the science supply side and at the demand side. From the supply side, the identification of available knowledge, gaps therein and strategies for filling these gaps are in progress. These results will be combined with those coming out from the analysis, on the demand side, related to the identification of requirements and limitations in the full range of situations, in all public and private domains.

2. Main findings

The review is organised according to the steps of LCA procedure (goal and scope definition, inventory, impact assessment and interpretation), and two separate sections are devoted to “cross issues” and to map available software tools and databases. The main findings of each section are summarised below.

2.1 Goal and Scope definition (Section 5.2)

Two main topics have been addressed: System boundaries definition and Scenario analysis.

For the ***system boundaries definition,*** one of the most studied topics in the literature, four main scientific approaches have been identified:

- reducing or eliminating the need for cut-off decisions;
- developing knowledge and methods to improve the basis for cut-off decisions;
- defining other types of system boundaries; and
- finding a more relevant system perspective than the cradle-to-grave perspective.

Despite the numerous studies and publications on this matter, the knotty problem of defining the system boundaries and deciding whether or not to apply the system expansion has not been solved yet. What is clear is that a “one size fits all” solution is not possible, due to the variability of the decision contexts. No one method stands out, but the approach adopted in structuring a consequential LCA opens new thoughts and thus more efforts should be spent on this issue, with more case studies and, mostly, by the development of procedural guidelines.

As regards ***scenario analyses***, a clear classification and categorisation of the different types of scenarios and scenario techniques, as proposed by [Höjer et al., 2008] represents a good starting point in order to improve the use of scenarios in LCA, because it better allows identification of which situation one approach is more suited than another. Furthermore, additional research is needed in this area, both at methodological and practical level, trying to find a balance between the feasibility and the uncertainty related to scenario development. One research line could be devoted to developing pre-defined scenarios, with a defined resolution at different levels, in order to increase their use in analytical tools such as LCA.

2.2 Inventory (Section 5.3)

The analysis of literature on the inventory step has considered the following main issues:

- consequential approach,
- the time dimension,
- hybrid analysis, and
- allocation.

Some of them are strongly interrelated, but we have kept the analysis of each issue separate in order to have a clear structure in the discussion.

Consequential approach

The consequential approach represents a new way to conceive LCA that has direct effects on the majority of the methodological problems. The debate is very vivid in the scientific community and no consensus has been reached on the relevance of the knowledge generated by attributional and consequential LCA, and on its practicability. Indeed, the diatribe between consequential and attributional should be overcome, because depending on the type of sustainability question, the right answer can be found in consequential as well as attributional LCA: asking the right question is the starting point, which will avoid misunderstanding and unrealistic answers. Efforts should be spent on accumulating further experience from successful LCAs and by making available to practitioners guidelines on how to properly deal with the consequential approach. From the practicability viewpoint, several efforts are necessary on the side of marginal data: the present problems are related to the identification of what type of marginal effects (short term or long term) should be included in the consequential LCA and how to identify the marginal technology. Furthermore, there is an apparent need to investigate the feasibility of handling the uncertainties involved in the identification of marginal technologies.

Thinking in consequential way means thinking about the consequences of the actions, to the interrelations and thus it means to project the problem at market level, with all its dynamics. In this context partial equilibrium modelling becomes relevant, since it introduces market mechanisms in LCA models by describing the balance between supply and demand of specific products. Its introduction in the framework of LCA requires still investigation at conceptual level in order to answer the following main questions: when partial equilibrium modelling is relevant, if and how it should be integrated into or used in parallel to LCA, and for what type of goods a change in demand in a life cycle affects the demand in other life cycles. If consensus is reached, efforts will be necessary at practical level: data on price elasticities of demand and supply of several products need to be estimated, and they should be compiled in databases that are posted in connection to ordinary LCI database.

In addition, experience curves represent other important mechanisms, as they make it possible to estimate the possibly huge environmental effects of investments in new technologies, by describing how a technology becomes less expensive as the production of the technology grows more efficient with accumulated experience. Indeed, a decision to invest in a new technology can have a large impact on its future. As the accumulated experience on the technology increases, this makes subsequent investments in the technology less expensive and hence more likely. Over time, the technology will have a much higher market share than it would have had without the initial investment. If this effect is taken into account, it can have a huge effect on the LCA results in some cases.

The issue of experience curves and their relation with LCA still need to be debated, because there are several open questions. Among them: the use of experience curves for making forecasts of future emissions and for estimating the effect of investments in new technologies; whether the concepts of experience curves and learning investments should be integrated into the LCA and, if so, what approach should be used in what circumstances. Regarding the feasibility, to make the combination of LCA and experience curves feasible, experience curves need to be established for more technologies, with related data posted in connection to LCI databases.

In the realm of consequences, the concept of rebound effects has been treated by many authors, but its applicability in LCA is still an open question, because of the complexity, uncertainty and costs involved. It is a large field of research as it includes various types of cause-and-effect relations: indeed, a change that affects the price, quality, functionality, need for maintenance, etc. of a product, can influence the demand for this and for other products [Thiesen et al., 2007]. But it is still methodologically immature: for several rebound effects no method has been found to quantify the effects and, even when established tools (price elasticity, general equilibrium models) can be applied, further research is necessary, because the quantification of rebound effects depends on important subjective methodological choices that add uncertainties to the evaluation.

Introducing time in LCI

The review has analysed the approaches related to time introduction in LCI both in terms of dynamic evolution of time and of scenario analysis, considering the time as a dimension linked to the future states. "True" (i.e. in terms of continuous mathematical function) dynamic approaches are still pioneering, and several efforts are still necessary both at methodological and practical level: indeed, the available software tools do not reflect advances in modelling, because they are based on static relations, and are not supported by databases that could be representative of the future situation.

An accurate balance should be found between the need of having an optimum representation of the reality and the complexity/feasibility of the modelling itself. For decisions related to the long term, the use of scenarios mentioned above (par. 2.1) could be more relevant and feasible: on this aspect, efforts should be spent developing technological scenarios related to the main processes.

Hybrid approaches

During the last few years, the economic discipline of *input output analysis* (IOA) has contributed to the strengthening of LCA. Indeed, with the IO-based LCI approach, process-based LCA can be made more complete by adding environmental IO data for more remote parts of the system, allowing problems of setting boundary conditions and of data availability to be overcome. Several case studies have been generated and at present, performing an IO-based LCI is relatively straightforward and the necessary tools are readily available; however, there is still a need for further research, mainly related to the following aspects:

- Data reliability, in terms of extended environmental intervention databases for IO tables and higher resolution for commodity classification.
- Uncertainty evaluation: no research lines seem to exist on uncertainty factors for IO data but these will be of great interest, due to the repercussion of the methodology on large scale application.

Considering the present state-of-the-art and the inherent limitations of the methodology, we consider IO-based LCI as an intermediate step towards more integrated approaches, like those represented by the *integrated hybrid analysis*. The latter is still considered a complex method, which adds to the cost of already expensive and time- consuming full process LCA, but at the same time it is considered one of the best choices for the future. As the practical implementation is not yet publicly available, more efforts should be spent on drafting case studies, disseminating useful findings and, in parallel, on developing databases and user-friendly software tools, since most commercially available LCA software are not able to handle matrix inversion for LCI computation.

Allocation

In the debate on how to perform the allocation, in terms of what allocation approach is the most appropriate in different cases and how to identify the most appropriate approach to allocation, the review addressed different types of allocation problems: multi-output processes, multi-input processes, and open-loop recycling. The diversity of views and perspectives in the LCA community is evident. The ISO procedure has also been subject to conflicting interpretations: researchers disagree on what approaches are allowed according to the ISO procedure, and on what approaches are possible.

The system expansion stands out like the most suitable approach to the allocation, however important drawbacks still have to be faced, like the increased data need and the more complicated system to be modelled. The system expansion approach can also introduce new allocation problems in LCA, but the new allocation problems are often less important than the original ones, which means that it is a fair approximation to neglect the new allocation problems or to solve them with a simple approach.

A significant effort is still required to reach a general agreement on the allocation.

2.3 Impact Assessment (Section 5.4)

Impact Assessment has been reviewed according to the following structure:

- Mandatory elements:
  - Improvement of existing characterization models
  - Common framework for the development of mid-point and damage-oriented methods
  - New characterisation methods and new impact categories
- Optional elements:
  - Normalization
  - Weighting

Mandatory elements

a. *Improvement of existing methods of characterization*

The literature analysis shows that different indicators and characterisation models have been proposed to calculate site-dependent characterisation factors (CFs) for various interventions and for several impact categories (acidification, photo-oxidant formation, terrestrial eutrophication and toxicological impacts), but the comparability of different models is still an open question. New suggestions come also from the global fate and exposure model GLOBOX, which provides a methodological framework for the construction of spatially specific characterisation factors, but the list of CFs for different countries has not been published yet.

Besides the spatial differentiation aspects, improvements have been identified for toxicity (human and eco-toxicity) and abiotic resources. Regarding the latter, the focus is moving from resource extraction to the concept that exploited resource come back to the environment in a degraded form, which is no more able to deliver its original functionality. A framework has been proposed for assessing the impacts from resource use, but further elaboration is needed.

On the toxicity side, the development of USEtox model allowed the hazard calculation of more than a thousand chemicals. It will be further developed during the second phase of the UNEP/SETAC Life Cycle Initiative with, e.g., improved CFs for metals.

b. *Common framework for the development of mid-point and damage-oriented methods*

Methods of this type are already available (IMPACT 2002+, LIME) or being developed (Recipe project, LIME2). However, the limited scientific knowledge of certain aspects does not allow defining quantitative impact pathways up to the damage categories for all type of impacts. The UNEP/SETAC Life Cycle Initiative has also nominated a task force aimed at developing a common framework for the two approaches. It is expected to provide a basis for the analysis and the comparison of the existing methods and to facilitate the inclusion of new impact categories, also those particularly suitable for developing countries.

c. *Development of new characterisation methods and new impact categories*

In this section, new characterisation methods for categories that have not yet been elaborated in Jolliet et al. (2004) have been considered.

Two main trends have been identified: one related to the combined approach of LCA with other methods, in particular risk assessment, and one devoted to the development of new impact categories like noise, land use, exergy, ionising radiation, water use, indoor and occupational exposure, and categories for specific production sectors.

On both sides, fully developed approaches are not ready. In relation to RA and LCA, the advantages given by the combination of the two methods need to be clarified and further elaborations are required to identify which specific methods are useful to combine and for which decision-situations.

About the new impact categories, characterisation models (land use, water use) and CFs calculation (land use, ionising radiation, water use) are common research needs for the majority of the proposed approaches. Regarding exergy, different methods have been proposed, at present not immediately comparable, thus the development of general methodological guidelines would increase comparability. Besides, all approaches require more data than conventional LCA and many of these still need to be collected and/or calculated. Research in the field of indoor and occupational exposure is moving towards the treatment of these aspects not as a separate impact category anymore but it is foreseen their inclusion as a compartment in the human toxicity impact category, such as in the current improvement of the USE-tox model for life cycle impact assessment of toxic releases.

Optional elements

Regarding *normalisation*, the review highlighted the importance of bias, suggesting that they should therefore receive further attention resulting into clear guidelines how to deal with them. Consistency should be ensured for methodological and data choices made in working out normalization and valuation/weighting data and in performing LCA case-studies.

For further progresses, it would be useful to draft a list of these issues as checklist for practitioners and normalization data/method developers.

On the *weighting* side, its contribution to the relevance and acceptability of LCA results is matter of discussion. Different proposals have been analysed, like the use of conjoint analysis (see LIME), ecotax method, damage costs/prevention costs, and monetised health impacts. However no method stands out, because controversies at conceptual level still exist, despite recognizing the importance of weighting for communicating results.

2.4 Interpretation (Section 5.5)

A specific remark upon the Interpretation phase is necessary, because it shows different trends compared to the other phases. It seems to be a "free zone", in which the lack of clear procedural guidance in ISO framework has legitimated a scarce development, together with the inherent features of the interpretation itself.

In the framework of Interpretation phase, data quality assessment and uncertainty analysis have been identified as influential elements in order to guarantee the credibility and reliability of the study. Regarding data quality assessment, the review showed that no major new insights or progress have been presented since the end of the 1990's: methods proposed in those years are applied at present. Thus what seems to be necessary is to work on reaching harmonisation and agreement among the methods already available.

Uncertainty analysis has so far not been properly addressed by LCA researchers and practitioners not always perform uncertainty analysis in their LCA applications. At a general level, for all the types of uncertainty identified (parameter, model and scenario), more guidance is needed in terms of guidelines on the definition of uncertainty in LCI and LCIA, together with an increased number of case studies serving as good examples. Due to the complexity and variety of choices and sources of uncertainties, scenario uncertainty is easily treated at qualitative level; nevertheless, major efforts should be spent on it, due to its repercussion on the reliability of final results.

2.5 Cross issues (Section 5.6)

Cross issues are those that cannot be classified as part of a specific step of the LCA standard procedure, because they are horizontal to the methodology, aiming to broaden its scope and/or improve its applicability. In this review they include (Environmental) Life Cycle Costing (LCC), Social Life Cycle Assessment (SLCA), and Simplified LCA.

The forthcoming SETAC publication on *LCC* represents a fundamental step in the improvement of the methodology, since it addresses the question of how costs and environmental aspects can be combined and provides a clear guidance for performing LCC studies. The approach contributes to the development of a code of practice for LCC and leads to a potential standardisation in analogy to ISO 14040 series.

*Social Life Cycle Assessmen*t does not show the same level of development as LCC; however, while in its infancy, it is nevertheless subject of an increasing number of published papers, which demonstrates the existing interest in the methodology and its application. The methodological framework, based on the ISO-LCA structure, was proposed by the taskforce "Integration of social aspects in LCA", nominated in the context of the UNEP-SETAC Life Cycle Initiative. Several aspects need to be discussed, like the *scope* of the analysis, *system boundaries*, selection of *indicators*, formulation of indicators and *data collection*. The forthcoming UNEP-SETAC publication, expected in October 2008, will provide a new state of the art and represent an important step for better addressing the research needs.

Regarding simplified approaches, no recent progress or developments have been identified. Several applications exist, but they are based on the methodology developed in the 1990's. This means that the topic can be considered quite mature and it is time to work on reaching consensus on when and how to simplify the analysis.

2.6 Tools (Section 5.7)

The literature review dealt also with the main LCA tools (software and databases) in order to analyse whether those available on the market have already implemented functions and structures that support upcoming methodological developments highlighted in the critical review.

42 software and 26 databases have been mapped and classified, according to a predefined set of parameters that consider the applicability in traditional ISO-LCA and the possibility to deviate from the standard methodology towards deeper and broader approaches. The results show that, at present, the main LCA software and databases are not sufficiently capable of facilitating broadening and deepening LCA: making them more capable will require a considerable effort in the design of innovative tools as well as in the definition of new data quality requirements and in the collection of new data.

2.7 Closure notes

The work performed has shown a great diversity of new thoughts in particular in the inventory and impact assessment phases. Developments with a different degree of "hardness" have been identified, starting from the consequential school, that has given rise to a new mode to conceive LCA, with consequences for many methodological issues, such as allocation, system boundaries, modelling changes over time, etc. Other approaches see an increasing use of different methodologies combined with LCA, like Input Output Analysis, and the combination/integration with other tools put the question of how far we should go in "improving" LCA.

The new insights identified show different levels of maturity: some of them can be already considered ***quite mature***, like:

- system boundary,
- allocation and
- data quality assessment.

Indeed, the debate will never end but some elements could be already made available to experts for reaching consensus, working on the procedural side more than on the analytical one. Regarding this aspect, the harmonization work of the ILCD Handbook, as coordinated by the European Platform on LCA, should help to fix the presently best-available methodology recommendation for LCA use in business and public policy context.

On the contrary, the issue related to the application of the *consequential approach* and the *hybrid analysis* still require ***further research and developments*** before being extensively used within the LCA framework: many efforts are necessary, both at conceptual and practical level, and they require the involvement of expertise also outside LCA community.

1. Introduction

The standardization of LCA in the ISO 14040 series has contributed to its broad acceptance and wide use, but at the same time the simplifications are often considered too restrictive, in particular for meso (analysis of e.g. waste treatment, industry sectors, etc) and macro (e.g. complex technological systems) scale applications.

In the last years many authors in their applications of LCA tried to go beyond the ISO standard, raising the question on which directions the methodology should move to in order to improve reliability, significance and usability of the applications of LCA, performing analyses more suitable for the concept of sustainability. Thus, several methodological issues have been discussed, ranging from time and space modelling, to rebound effects and new efforts towards social and economic evaluations, just to mention some. This state of ferment is a clear symptom that the methodological framework, as defined in ISO standard, is often judged too narrow for the applications needed, i.e. broad systems, with complex interrelations and dynamics. CALCAS1 (Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability) project starts from these considerations.

Within CALCAS, we reviewed the scientific literature on ISO-LCA, in order to identify new directions for broadening (e.g.: including economic and social aspects, or covering new environmental aspects) and deepening (e.g.: including behavioural aspects in the inventory modelling or including more fate and exposure mechanisms in impact assessment) LCA. In the review were also of interest the identification of intrinsic limits of ISO-LCA, due to assumptions and simplifications; identifications of elements in current ISO standards which are not in line with new scientific developments or best practices, or just are missing and hence enough guidance is not given.

The work has been organised along four main activities:

- the identification of the topics to be analyzed and the definition of screening criteria for literature selection;
- the development of an evaluation grid, in order to support the analysis of the scientific literature and to allow for consistency in the analysis performed by different researchers;
- the analysis itself, including both methodological and practicability aspects (software and databases).
- the reporting of the results of the analysis.

All in all, more than 40 software and 25 databases have been mapped, and about 60 new approaches (in more than 250 papers) with a different degree of maturity have been analyzed.

The work has been jointly conducted by a team of CALCAS partners, in particular: Reinout Heijungs (CML) is the author of par. 5.2 "*Framework, scientific foundation and definition of ISO-LCA*"; Tomas Ekvall (IVL) analysed the literature on

[1] www.calcasproject.net

Consequential LCI (and related items), *System Boundary* and *Allocation*; Jeroen Guinée (CML) analysed the literature on *Impact Assessment;* Raffaella Bersani, Agata Bieńkowska and Ugo Pretato (JRC-IES) analysed the available tools. ENEA analysed the literature related to all the remaining subjects, further elaborated the analyses and is the editor of the present report. Because this work was performed by different authors, some unhomogeneity is present in the report. We consider not necessary to proceed further in harmonising styles and contents in the different sections, as this is an intermediate report

The present report is deliverable D7 of work package 5 (WP5) of the CALCAS project and summarises the first phase of the activities of WP5. The WP5 activities will continue with the 2nd phase, in which the more promising developing lines will be identified (deliverable D14) together with standardisation guidance on topics were a consensus could be easily reached (deliverable D18).
All the work performed will serve as input for WP7 activities, in which the most interesting and feasible options for expanding LCA will be developed, and the main lines of a scientifically feasible, a relevant for sustainability governance, and an applicable by stakeholders New LCA will be defined. These results will be the fruit of the activities of all the WPs, by crossing the results of the other analyses both at the science supply side and at the demand side. From the supply side, the available knowledge, gaps therein and strategies for filling these gaps will be identified; from the demand side requirements and limitations in the full range of situations will be specified, in all public and private domains.

2. READING GUIDE

The present report, after a short description in Chapter 3 and 4 of the main purposes of the review process and of the method adopted for reviewing the scientific literature, describes in Chapter 5 the core of the analysis. It is structured in paragraphs, organised according to the ISO 14040 structure as much as possible: Par. 5.2 presents an outline of the framework, foundation and definition of ISO-LCA; par. 5.3 to 5.6 are related to the four phases of the LCA methodology, namely Goal and Scope definition, Inventory, Impact Assessment and Interpretation. Par. 5.7 deals with what we defined “cross issues”, i.e. issues that either are outside of the present ISO framework, as (Environmental) Life Cycle Costing (LCC) and Social Life Cycle Assessment (SLCA), or affect the overall methodology, as simplified LCA. The last paragraph of the chapter is dedicated to the analysis of LCA software and databases, in particular to understand whether the main tools available on the market may have already implemented functionalities that support upcoming methodological developments highlighted in the critical review. Chapter 6 summarises the main results, highlighting to what extent the present developments of LCA cover the broad spectrum of mechanisms needed for broader evaluation like those related to the field of sustainability.

3. BACKGROUND

A central question in the research for sustainability is how to support public and private decisions by assessing existing and future product systems. A common understanding in the scientific community is that any sustainability assessment requires a “life cycle” approach, as the only way to avoid problem shifting. Presently, the most mature life cycle based assessment method is Life Cycle Assessment. However, LCA has been developed and standardized for evaluating the environmental potential impacts of goods and services, by applying a “simple” linear static model based on the technological relations of the product system and without taking into account the social and economic effects[2]. Indeed, when the assessment is required for systems with high impacts on the economy and society, ISO-LCA shows its limits, because many possible relevant effects are not even considered. While these limitations are also due to the useful intention to keep LCA operational, to limit its complexity and structural uncertainty by focussing on the main and direct effects along the life cycle, the need exists to evaluate in which respect and how far to go beyond to further improve decision support. Also, there is a sort of “standard paradox” as ISO 14040 and 14044 from one side provide limits to the analysis due to the above mentioned model limitations, but on the other side they do not provide enough guidance for many practical aspects of the LCA procedure. For this reason, many researchers and LCA practitioners in their applications have proposed a very large number of approaches[3], trying to overcome these limits and to provide guidance on how to implement practically some methodological issues not fully addressed by the standards.

The goal of the CALCAS project is to expand life cycle analysis to mend shortcomings of current LCA and improve its broader applicability, by developing new approaches and models and especially by indicating research lines for development and specifying road maps for their implementation. This expansion starts out in WP5 by exploring the new directions in LCA with focus on broadening and deepening:

- broadening, such as including economic and social aspects, or covering new environmental aspects;
- deepening, such as including behavioural aspects in the inventory modelling, or including more fate and exposure mechanisms in impact assessment.

This exploration has been conducted analysing the available scientific literature, adopting the scheme of LCA phases defined in ISO 14040.

The critical review is intended to make it available a comprehensive and rational state of the art of LCA literature, in order to identify areas of controversy and to forms the basis for the identification of future research in this area.

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[2] For a more extensive and comprehensive description of this statement please see: D1(Heijungs, R. et al., 2007) and D15 (Heijungs, R. et al., 2008).

[3] It should be noted that in this report we changed the terminology with respect to the description given in the Annex I to the CALCAS contract: we think more appropriate to refer here to “R&D approaches” instead of “R&D initiatives”.

It is out of the scope of the present work any judgements about the validity of the analysed approaches and discussions on the achievement of the declared goal by the author(s) of the approaches. Anyway, despite the critical review has been objective in its purposes, the papers published reflect the subjectivity of the authors. Thus, since in scientific journals papers that analyse and criticize a methodological approach proposed by other authors are rarely present, this review could partially reflect only subjective elements, intended as “representative of the judgement of one or few authors”.

The main features of this review, the criteria for the literature selection and the main aspects analysed, are described in the following chapters.

4. METHOD OF ANALYSIS

The analysis of the state-of-the-art has been broken down into three main lines:

- Rationale, i.e. the basic principle of ISO-LCA foundations. It describes the essential scientific and procedural characteristics of LCA, starting out by the context in which the conception of ISO standards has taken place.
- Procedure, i.e. how the methodology is implemented. It describes the main findings on current practice, aimed at setting up a consistent framework for analysing deviations/developments (both new aspects not covered by the standard and issues already covered but not sufficiently detailed).
- Tools, i.e. software and database. It aims at identifying the tools available on the market that are in line with the methodological developments highlighted in the survey.

For each phase of LCA methodology, the analysis has been broken down into “topics” (the main issue from the methodological viewpoint, e.g. allocation, simplified methods, etc.) and “approaches” (solutions proposed by the authors in order to deal with the problem). For example, one topic is the improvement of existing methods in the impact assessment; the related approaches are which developments exist for the impact categories (e.g. eutrophication, climate change, etc.).

Despite some topics belonging to the impact assessment require also a different LCI approach, in order to better manage the literature review we fully allocated them to the impact assessment phase, and we treated there – as far as possible – also the related problems that occur in inventory phase.

The fig. 1 shows the final tree of topics and approaches identified for the review.

For each approach, the relevant literature has been identified according to the following selection criteria:

- only officially peer-reviewed references published after 2000 until July 2007. More recent references (until May 2008) have been considered regarding the following issues: scenario analysis, uncertainty (parameter, model and scenario) analysis, time introduction in LCI modelling, LCC and SLCA.
- case studies only in so far they present or illustrate a new development;
- max. 5 references per heading; if there are more than 5 references; further selection should focus on review-papers, justifying the selection in view of complementarity of authors and approaches.

For LCIA, an added criterion is:

- if only 1-2 references from the LCIA domain could be found, 1-2 from the non-LCIA domain may be added when available.

Several journals have been consulted, not only in the domain of life cycle assessment but also related to other disciplines, like risk assessment, economic system, etc.

The principal journals considered were:

- International Journal of Life Cycle Assessment
- Journal of Industrial Ecology
- Journal of Cleaner Production
- Ecological Economics
- Waste Management
- SETAC publications
- Chemosphere.

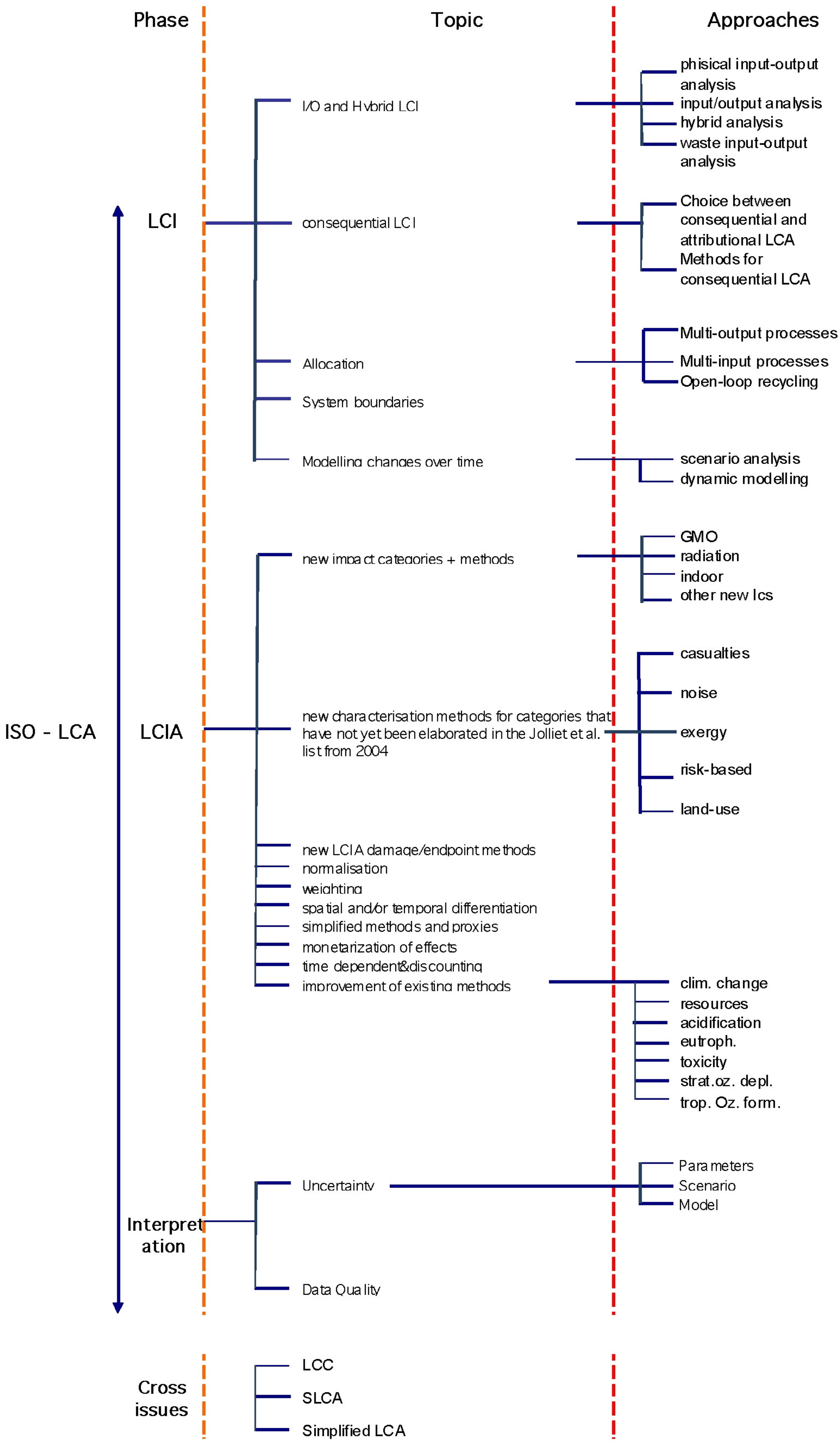

Fig. 1 - Topics and approaches identified

Other sources:

- Environmental Impact Assessment Review
- Environment International
- Energy Policy
- Resources, Conservation and Recycling
- Environmental Science & Technology
- Economic System Research
- Science of the Total Environment
- Ecological Modelling
- Environmental Modelling & Software
- Risk Analysis
- International Journal of Business Environment
- Environmental Progress
- Exergy, an International Journal
- Energy conversion and Management
- Ecological Risk Assessment

The criteria for the selection have not been completely fulfilled for all topics. Some exceptions have been made, due to the peculiarities of the issue and to the different degree of development and implementation. In particular, case studies have been analysed if they illustrated new developments; other references besides peer reviewed articles (like books and conference proceedings) have been also consulted; references before 2000 have been considered when they presented interesting approaches not taken into account in the subsequent scientific literature.

In order to manage the huge amount of literature available and make it possible to divide the analysis among different researchers assuring consistency, an evaluation grid has been drawn up, structured along three levels: a) *Generalities*, which include the description of the analysed topic/approach, the deviations/developments with respect to ISO standards and the relevant references. b) *Analysis*, the core of the grid, with a description of the rationale (key principles of the approach analyzed, why the approach is implemented, which needs the authors addressed), main advantages, open questions and practicability aspects. c) *Comments*, in terms of R&D needs and trends.

All in all, more than 40 software and 25 databases have been mapped, and about 60 new approaches (in about 250 papers) with a different degree of maturity have been analysed.

5. CRITICAL REVIEW

5.1. Framework, foundation and definitions of ISO-LCA

In this paragraph the ISO-framework will be briefly described, adding discussions that relate to the scientific foundations of LCA, the deepening and broadening of LCA, and the context in which the conception of the ISO standards has taken place.

The history of LCA has been described by several authors (Hunt et el, 1996; Oberbacher et al., 1996; Boustead, 1996; Gabatuler, 1997) as well as by Fava et al. (1991) and Baumann & Tillman (2004). The need for a standardization of methods and/or procedures and/or terminology had become evident in the beginning of the nineties of the last century following the discrepancies in conclusions of different case studies on similar products (Guinée et al., 1993). The Society of Environmental Toxicology and Chemistry (SETAC) organized a number of workshops that produced documents aimed at fulfilling the role of a standard (Fava et al., 1991; Fava et al., 1993, Consoli et al., 1993). Meanwhile, in several countries national initiatives were undertaken that lead to national or regional methods, most notably in The Netherlands (Heijungs et al., 1992), in Scandinavia (Lindfors et al., 1995), and in the US (Vigon et al., 1993).

The proliferation of mutually incompatible methods and terminologies created a new problem, and soon the industry-driven initiative of creating an ISO-standard was born. The inception of such a standard has been described elsewhere (see, e.g., several contributions to the International Journal of Life Cycle Assessment, Volume 2, Number 1 (1997), and follow-ups in later issues of this journal). In any case, the gradual evolution of LCA is reflected in the publication of a whole series of standards and technical reports by the ISO; see Table 1.

Number	Type*	Title	Year of issue
14040	IS	Principles and framework	1997, 2006
14041	IS	Goal and scope definition and inventory analysis	1998
14042	IS	Life cycle impact assessment	2000
14043	IS	Life cycle interpretation	2000
14044†	IS	Requirements and guidelines	2006
14047	TR	Examples of application of ISO 14042	2003
14048	TS	Data documentation format	2001
14049	TR	Examples of application of ISO 14041 to goal and scope definition and inventory analysis	2000

* IS = International Standard; TR = Technical Report; TS = Technical Specification
† Replaces 14041, 14042 and 14043

Table 1 - Overview of the ISO standards and technical reports

Meanwhile, it has been recognized that the entire ambition of international standardization may at times be too tedious for practical purposes, given the seminal state of LCA. For instance, while the ISO standards provide a framework for life cycle impact assessment, they do not recommend concrete methods for carrying out this step, let alone tabulate characterisation factors to be used in such a step. Several SETAC working groups have worked in the 1990ies and early 2000s on several topics to provide more concrete guidance on a number of issues, ranging from nomenclature of elementary flows through LCIA methods and framework, and life cycle working environment to data quality, to name a few. The UNEP/SETAC Life Cycle Initiative (http://lcinitiative.unep.fr/) has been established for a variety of reasons, including to "Identify best practice indicators and communication strategies for life cycle management". At the moment of writing, contributions to a best practice that have been developed from 2001 to 2005 have been published for some topics, but it has also been recognized that it was early to formulate a best practice for all aspects of LCA.
The European Commission, through the European Platform on LCA, co-ordinated by the JRC IES, is now working in close consultation with a number of third countries with National LCA projects, the 27 EU Member States, and with UNEP to provide by the end of 2008 the International Life Cycle Data System (ILCD) in support of good practice in LCA and its applications in business and government. The development of the ILCD is furthermore carried out in consultation with three distinct Advisory Groups composed of presently 14 EU-level industry associations, 15 leading LCA software and database developers, and 6 LCIA method developers.
The ILCD consists of a Handbook and a Data Network. The ILCD Handbook is a series of technical guidance documents to the ISO 14040-44 standards, including explicit and goal-specific methodological recommendations, a multi-language terminology, a nomenclature, a detailed verification/review frame and further supporting documents and tools. It will serve as a basis for comparable and quality-assured LCA studies and applications in business and the public in general as well as in the European Union for the implementation of key EU policies: the two Thematic Strategies on Resources and Waste and the forthcoming Sustainable Consumption and Production (SCP) Action Plan.
The online ILCD Data Network will be made of independently managed and published data sets that meet the common requirements on methodology, quality, nomenclature, documentation, and review of the ILCD Handbook. The ILCD Data Network will include the European Reference Life Cycle Database (ELCD), composed of LCI data sets for core materials, energy carriers, transportation activities and waste treatment services representing the EU market, provided or approved by relevant industry associations as far as possible.
Moreover, as a common basis for LCA work, a comprehensive set of elementary flows, unit conversion data sets, and global default impact assessment methods and factors will be recommended and maintained.

Perhaps, this long way towards agreed practice is a ”normal” process: Physics has a history of 500 years. Its body of knowledge can be described as consisting of a well-established core (including topics like Newtonian mechanics, optics and thermodynamics), and an area of frontier science in which most knowledge is labelled as tentative or even speculative. This applies, for instance, to string theory and the theory of elementary particles. For several reasons, the fraction of knowledge that belongs to core science is much lower for LCA. Reasons include:

- LCA is a field of academic research and use in industry practice for not more than 25 years.
- The research effort that has been put into LCA is much smaller than that put into a field like particle physics (as an example, the Large Hadron Collider at the CERN laboratories costs about 3 billion euro).
- The procedures of empirical science (hypothesis testing, validation, etc.) are impossible or problematic in LCA (for theoretical and practical reasons).
- There is rarely an objective truth in LCA, nor in other kinds of systems analysis of socio-technological systems. The LCA results will always depend on a choice of perspective. In other words, it is not just that we have not yet found the truth; but that we cannot expect it to exist (Heijungs, 2001). Thus, no argument can be expected to prove scientifically that an allocation method is correct, etc.

As a result, we see more speculative and “badly founded” methods and proposals in LCA than in physics. There is not yet an “Ohm’s law” for LCA. This does not mean that LCA is without foundations. But it does mean that it is difficult to say what is generally accepted in LCA, and what’s not.

As an interlude to discussing the foundations of LCA, it is worthwhile to reflect one moment on the scientific status of LCA. In the development of sciences, periods of steady progress are interrupted by periods of turbulent Kuhnian “paradigm shift” that are instigated by an accumulation of paradoxical results. As has been noted by Bertrand Russell in his Introduction to mathematical philosophy, “Mathematics is a study which ... may be pursued in either of two opposite directions. The more familiar direction is constructive, towards gradually increasing complexity: from integers to fractions, real numbers, complex numbers; from addition to multiplication to differentiation and integration, and on to higher mathematics. The other direction, which is less familiar, proceeds ... to greater and greater abstractness and logical simplicity.” Indeed, as late as 1900, the great mathematician Giuseppe Peano published on the foundations of mathematics with postulates like “0 is a number” and “The successor of any number is a number”.

It is natural that, at a certain stage of development, researchers are not only exploring the deeper and broader areas of LCA, but are also addressing the things that seemed obvious at first, but that turn out to be more perplexing on a second thought.

For instance, while the first LCA studies just started with data and made calculations on that basis, a later development started to introduce the distinction between prospective and retrospective, or marginal and average, or change-oriented and descriptive LCA. And whereas the *Code of Practice* stated that in 1993 of the inventory analysis that it was "defined and understood; needs some further work" (Consoli et al., 1993, p.7), the debate on the principles for inventory analysis have been revitalized on many topics. One reason for this is that many "old" contributions to LCA do meet the normal criteria for scientific practice. For instance, the ISO 14041 standard gives a hierarchy of approaches to deal with allocation and recycling, but does not give arguments, let alone proofs. In fact, dissatisfaction with the allocation problem has been a rich source of development of LCA, with spin-offs to the marginal-average debate, the system boundaries debate, and other issues as well. This does not only apply to the allocation topic: the ISO standards lack a serious scientific foundation altogether. So, at the end of the nineties, we see a sudden rise in the interest of the foundations of LCA.
Contributions to the foundations of LCA have been laid by several groups, mainly by academic researchers, and often in the context of PhD-thesis research. We mention several main directions in this respect:

- links with decision theory;
- links with systems analysis;
- links with economics;
- LCA as a deductive science.

First and foremost are the attempts to connect LCA to **decision theory**. Hofstetter (1998), Hertwich et al. (2000), Seppälä et al. (2001), Rahimi et al. (2004) provide examples of this line of research. Decision theory is supposed to provide a framework and methods that can help to found LCA, or at least to be useful in the context of some aspects of LCA (such as choosing impact categories or setting weighting factors).

Another approach starts from systems analysis or general systems theory, or from a general field of study that is based on a **systems approach**, such as thermodynamics. Azapagic & Clift (1999) provide an important example of this approach.

A third line is the connection with **economic theory**, for instance with input-output analysis (Suh, 2005), production functions (Heijungs, 2001), or marginal theory (Ekvall & Weidema, 2004). Here, LCA is seen as extending on ordinary economic analysis, just like environmental satellites may be added to economic accounts.

A fourth approach is to consider **LCA as an analytic, formal science**, like mathematics and logic, and to base it on definitions and axioms, which provide the material to deduce further theorems. Although it may seem illogical to construct a science with actual empirical content on purely formal axioms, it should be noted that this has been done in other similar cases as well. Newton did this for mechanics and optics and so did Debreu for economic value. Heijungs (1998) and Heijungs & Suh (2002) provide attempts to construct a theory of LCA on the basis of an axiomatic system.

It should be noted that these four different angles of founding the science of LCA are not necessarily exclusive in the sense that at most one of them can be the good one. The science of thermodynamics can be founded in different ways: on the basis of a statistical approach to the kinetics of gas molecules, on the basis of a few empirically grounded axioms ("the laws of thermodynamics"), etc. Likewise, the edifice of the science of LCA may be erected in different ways and using different approaches. As long as they provide compatible results, this will only add to an enhanced scientific foundation and understanding of LCA. Moreover, because LCA is diverse as to its disciplinary content, the foundations may well be as diverse. The decision-theoretical line, the systems-analytical line, the economic line, and the formal deductive line are all expected to further contribute to the foundations.

Part of the fruits of founding LCA is the elicitation of implicit assumptions. Thus, the debate between the proponents of using marginal data and those using average data pointed towards different interpretations of the role of LCA. Likewise, Guinée et al. (2002) suggest that three "modes of LCA", addressing occasional, structural and strategic questions, might require different operational methods. Discussing the foundations of LCA will make LCA more scientific, it will provide a means to judge between "good" and "bad" methods, but it will also clarify the place of good but competing methods, for instance in different decision contexts.

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5.2. Goal and Scope

5.2.1. System boundaries

ISO 14040 (section 3.42) defines the concept of system boundary as “a set of criteria specifying which unit process are part of the product system”. The product system includes the life cycle from raw material acquisition to final disposal (section 4.4).
ISO 14040 states that the system should ideally include all processes that are directly or indirectly connected by physical flows to the product or its function. This would imply (Raynolds et al., 2000) that all processes in the global economy should ideally be included in the study, but, according to ISO 14040 (section 5.2.3), processes that will not significantly change the overall conclusions of the study can be excluded from the system. For this reason, the choice of system boundary depends on the goal and scope of the LCA. To be more specific, it depends on the question to which the LCA should respond.
The international standard does not give any specific guidance regarding how the goal of the study affects the system boundary. Here, the distinction between attributional and consequential LCA is a useful example. Attributional LCA aims at describing the environmentally relevant physical flows to and from a life cycle. Consequential LCA aims at describing how the environmentally relevant flows, from the technological system as a whole, change in response to possible changes in the life cycle (Ekvall & Weidema, 2004). In an attributional LCA, the system investigated should include all environmentally significant processes in the life cycle, from raw material acquisition to final disposal. In a consequential LCA, the system boundaries should include those unit processes that significantly affect environmental impact, regardless of whether these processes are inside or outside the life cycle (Tillman, 2000; Rebitzer et al., 2004). In particular, the requirement of deciding which processes could be excluded from the inventory can be rather difficult to meet because many excluded processes have often never been assessed by the practitioner, and therefore, their negligibility cannot be guaranteed (Suh et al., 2004).
ISO 14044 (Section 4.2.3.3) states that the criteria that specifies which unit processes are part of the product system should take into account the mass of the physical flow, the energy demand of the unit processes, and the environmental significance of the unit processes. ISO/TR 14049 says, as an example, that the criteria could state that the system boundary should include unit processes responsible for at least 99% of the mass flow, 99% of the total energy demand, and 90% of each environmental impact category. The rest of the processes can be cut off from the system. A problem with such criteria is that it is not possible to know when 90% of the environmental impact or 99% of the energy demand is covered, unless data are collected for the full system, i.e. for all processes in the global economy (Raynolds et al., 2000 Part I).

The topic of system boundary definition has been one of the most studied in literature and four main scientific approaches have been identified in our review:

a) reducing or eliminating the need for cut-off decisions;

b) developing knowledge and methods to improve the basis for cut-off decisions;
c) defining other types of system boundaries;
d) finding a more relevant system perspective than the cradle-to-grave perspective

a) *Scientific efforts towards reducing or eliminating the need for cut-off decisions*.
The scientifically most significant approach in this effort is probably the application of input-output tables (Suh et al., 2004). This approach potentially eliminates the need for upstream cut-off, because the input-output table is an aggregated model of all activities in the economy: the advantage is, however, that at any time the assessment is complete in terms of upstream requirements.
LCA studies utilizing economic input-output analysis have indicated that, in many cases, excluded processes may contribute as much to the product system under study as included processes; thus, the subjective determination of the system boundary may lead to invalid results. Input-output analysis automatically takes into account capital goods and overheads as inputs to a product system, which are often deliberately left out by most of process LCIs for the reason of real or assumed limited relevance. From this viewpoint, the introduction of hybrid analysis, in which the strengths of both methods are combined, represents a potentially promising approach also beyond the issue of system boundary (see par. 5.4.4).
On the other side, other efforts to reduce the need for cut-off include the development of process-based databases and default data aiming to simplify data collection. It should be noted, anyway, that excluding unit processes from system boundaries because of data unavailability is an unacceptable method: it is not repeatable, has no scientific justification, and is not rigorous.

b) Scientific efforts towards developing knowledge and methods to improve the basis for cut-off decisions
The selection of the system boundary affects the completeness or scope of the life cycle system. Thus, to be efficient and provide a repeatable and rigorous comparison between systems, a system boundary selection method should (Raynolds et al., 2000 Part I):
i) be quantitative;
ii) not require data collection for all processes in the global economy;
iii) be easy to apply and possible to use also in streamlined LCAs;
iv) still consider the significance of processes and flows relative to the system as a whole, and
v) facilitate measurements of the system completeness.
Raynolds et al., (2000 Part I) proposed a quantitative method called RMEE (Relative Mass-Energy-Economy) that focuses on the mass, energy content and economic value of the flows rather than unit processes. The cut-off criteria are defined as a percentage of the mass, energy content and economic value of the flow

that relates to the functional unit of the system. As the percentage of the cut-off increases, the uncertainty introduced to the overall results increases as well (Raynolds et al., 2000 Part II).

Among the several aspects involved in selecting system boundaries, also the significance of capital goods has been investigated (Frischknecht et al., 2007). Many life cycle assessment case studies neglect the production of capital goods and it is still unclear whether or not capital goods can be excluded or must be included; the only exception is LCAs of metals, where capital goods have a substantial effect only on the land-use impacts. Generally it was found that a general inclusion is not necessary while a general exclusion leads to very relevant gaps in many cases, i.e. a case-wise decision may be most appropriate, including capital goods fully and more systematically into the application of cut-off criteria. Furthermore, the authors provide a useful synthesis matrix in which the importance of capital goods is classified, distinguishing between economic sectors and environmental impacts, with recommendation regarding their inclusion in LCA case studies.

It should be added that a similar situation exists for the cut-off of services, especially more indirectly related ones such as advertisement.

For closing these data gaps of capital goods and services, the use of modelled or estimated process-LCI data was suggested.

c) *Other types of system boundaries*

There are several dimensions involved in system boundaries, like geographical boundaries, boundaries in time and boundaries between the technological system and nature.

Regarding *geographical boundaries*, studies have investigated how different geographical boundaries in the electricity supply system affect LCA results (Koch & Harnisch, 2002).

Boundaries in time become relevant when emissions occur over a very long time, which can be the case with emissions from landfills or soil. For example, in environmental product declaration (EPD) of landfills (Del Borghi et al., 2007), the system boundary has been defined at 30 years after the closure of the landfill, in accordance with the product-specific rules defined for such EPDs. Indeed, the issue of the length of time emissions from the landfill is the subject of discussion from long time, and some authors suggested also that long time period should at least be considered in a complete LCA in order not to miss any important impacts. For example, Finnveden (1999) in his work considered a hypothetical, infinite time period, in order to get the maximum, potential impacts. In other cases a separate inventory for emissions occurring with 100 years after depositing and beyond are suggested, considering exclusively the first 100 years directly for decision support but considering the long-term emissions as additional information on a more general level as evidence-basis. The assumption behind the 100 years is that if long-term emissions play a relevant role, the society will, at least after 100 years – given the progress in environmental policies and technology – treat the whole waste deposit or recover even secondary raw materials. .

Boundaries between the technological system and nature are relevant to discuss, e.g., in the context of forestry processes, agriculture and landfills. It is related to the boundary in time. If the boundary in time is set to be 30 years after the closure of a landfill, the landfill site is regarded as part of nature after that time.

d) <u>Scientific efforts to find a more relevant systems perspective than the cradle-to-grave perspective</u>

Ekvall & Weidema (2004) argue that an assessment of environmental consequences should start with the decision at hand, and that it should trace chains of cause and effect originating at this decision. The resulting model does not resemble the cradle-to-grave model in a traditional LCI.

In another approach, Ny et al. (2006) argue that a sustainability assessment should adopt a top-down approach, with essentially no system boundaries but the ones that apply for the whole biosphere, focussing on activities that contribute significantly to society's violation of the sustainability principles in The Natural Step Framework.

Despite the numerous studies and publications on this matter, the knotty problem of defining the system boundaries and deciding whether or not to apply the system expansion has not been solved yet. What is clear is that a solution one size fits all is not possible, due to the variability of the decision contexts. No one method stands out but the consequential thinking opens new thoughts into the approach to the methodology and thus more efforts should be spent on this issue, with more case studies and, mostly, by developing procedural guidelines.

5.2.2. Scenario analysis

Scenarios in LCA cover all the procedure's steps: in "Goal and scope definition" the elements relevant for the scenario analysis are defined, while the modelling of scenarios is done in LCI and LCIA and, finally, in the Interpretation conclusions and results are discussed. Despite scenarios in LCA are very relevant, because the inherent decision-support nature of LCA (and decisions relate to future), the literature analysed does not show such evidence.

Indeed, the issue of scenarios is a complicated matter for two main reasons:

i) they deal with the future, that is uncertain by definition, and

ii) they involve expertise in different disciplines.

When translated into the LCA field, these principal features, pose several questions:

- Scenario definition and scenario categories.
- Techniques for scenario development.
- Data availability.
- Relevance: when the use of scenarios is relevant in LCA?
- Uncertainty: how to evaluate the inherent uncertainty of future evaluations?

- Expertise: the use of scenarios requires to go beyond the LCA domain, and to involve expertise belonging to the field of economy, planning, etc.

The paper presented by Höjer et al. (2008), addresses these questions: its main purpose is to analyse how different types of scenarios can be used in connection with different environmental systems analysis tools, among which LCA. This paper was very useful for this review, as it is an analysis of the state-of-the-art of the applications and role of scenarios into LCA. For this reason, this paragraph is mainly based on that paper. The authors refer to the work of Börjeson et al. (2006) that presented a guide to help users in selecting the appropriate scenario types for a specific situation and in understanding which specific category of scenario can be used for. A general framework for scenario development in LCA can be found in Pesonen et al. (2000) and Weidema et al. (2003a), but a first structured framework for scenario-based LCA was proposed by Fukushima (2002). Pesonen et al. (2000) introduced two basic approaches: what-if-scenarios and cornerstone-scenarios, also in relation to the time frame of the analysis. What-if-scenarios are used in situation with a short time horizon when the researcher is familiar with the decision problem and can set defined hypothesis on the basis of existing data; cornerstone scenarios are more suited to long term planning and give potential direction of future developments.
Börjeson et al. (2006) further detailed this classification by proposing the scheme showed in Fig. 1, in which three main categories of scenarios are distinguished, namely predictive (what will happen?), explorative (what can happen?) and normative (how can a specific target be reached?), each of them containing two scenarios types, respectively: Forecast and What-if, External and Strategic, Preserving and Transforming. What-if and cornerstone, in this scheme, belong respectively to predictive and explorative scenarios.

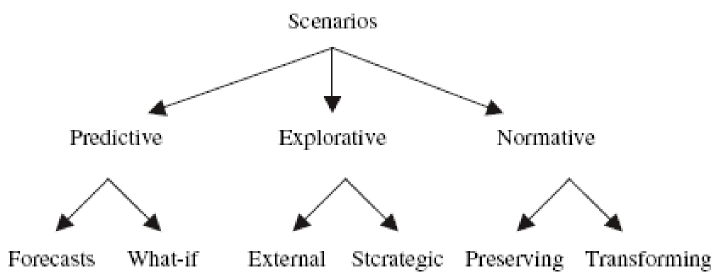

Fig. 1 – Scenario typology. *Source: Börjeson et al. (2006)*

The paper by Höjer et al. (2008) highlights other main aspects:

- *Data.*
  Data availability and reliability represent a hot spot that hampers the application of scenarios to LCA. Indeed, the data uncertainty can be very large in scenarios that refer to technologies not yet in use and, furthermore, most LCAs are based on input data that refer to several years ago, an element that makes their representativeness questionable in prospective studies.

- *Uncertainty*
  Uncertainty is inherent to future evaluations and reduces the value of long-term forecasts. Despite the authors state that there is no exact time frame when forecasts are considered too uncertain, because it depends on the aim of the study as well as on worldviews and perceptions, the knowledge generated by scenarios should be evaluated together with the associated uncertainty. Indeed, scenarios' user should be aware of the uncertainty related to the scenario, in order to make the most suitable choice, clearly also in accordance with other boundary conditions. Thus, dealing with scenario increases the need for methods for uncertainties treatment.
- *Relevance*, i.e. in which situation one approach is more suited than others, in terms of its ability to make the knowledge required available.
  The authors showed that predictive and explorative scenarios could be useful in analytical tools like LCA to describe future situations; on the other hand, transformative scenarios, in which large changes in the overall structure are involved, seem to be of little use in LCA due to the difficulty in making available reliable input data for LCA.

The work performed by Höjer et al. (2008) represents a good starting point in order to improve the use of scenarios in LCA, but further research is needed in this area, both at methodological and practical level, trying to find a balance between the feasibility and the uncertainty related to scenario development. One research line could be devoted to develop a set of forecasts, with a number of different parameters, which could be used as general input for many LCA studies (Höjer et al., 2008). Indeed, this contradict, as pointed out by Höjer et al. (2008), what the scenario literature often recommend, that scenarios should be tailor-made to the specific question. However, the limited resources available would suggest the development of consistent and generic scenarios of different types that could be used by different practitioners. Maybe, since the necessary expertise is not always available when scenarios are developed, the availability of pre-defined scenarios, with a defined resolution at different level, would increase their use in analytical tools such as LCA and the final results would benefit from it in terms of consistency and transparency.

Further elements on how to analyse scenarios in LCA will be brought into the debate by the upcoming "guidance document for future scenario LCA studies and data", within the ILCD Handbook of the European Platform on LCA (2^{nd} half of 2008).

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5.3. Inventory

Life Cycle Inventory (LCI) is generally the most time and resource-consuming phase in Life Cycle Assessment (LCA). For this reason, in the last years, it has been the object of many and vivid debates in the LCA community, with a great proliferation of new thoughts, like time and space modelling, rebound effects and new efforts towards social and economic evaluations, just to mention some. This state of ferment is a clear symptom of an LCA issue: the inventory methodological framework, as defined in ISO standard, is often judged too narrow - or maybe would be more appropriate to say “not detailed enough” from the procedural point of view - for applications more coherent with the real life.

The tentative to go beyond the basic features of Life Cycle Inventory, i.e. the use of a steady state and equilibrium model, has been the main push for new developments. Developments with a different degree of “hardness” can be identified, starting from the consequential school, which has given rise to a new mode to conceive LCA, while other approaches see an increasing use of different methodologies combined with LCA, like Input Output Analysis (IOA). The combination/integration with other tools put on the table the question of how far we should go in “improving” LCA without resulting in something that is not anymore LCA.[4] A related question is when a further complexity of the models used for LCI with indirect effects of consequences etc. adds more in uncertainty than it helps to reduce it.

The analysis of literature on the inventory step has been organized following the two main trends singled out:

- the consequential school
- Input Output Analysis and hybrid approaches

The first one is coherent with the ISO standard structure and exploits the freedom of action available by the standard; the latter acts at the computation level, with a less apparent closeness with the standard but potentially able at the same time to overcome the present limitations.

A third trend should be introduced but not as stand-alone: the inclusion of the time dimension.

To complete the review, the issue of allocation, despite its strong interrelations with the previous themes, will be discussed in a separate section, due to its specificities that would be lost if treated together with the previous ones.

[4] With respect to the nature of modelling, a clear overview has been done also in the Scope Paper of CALCAS (Heijungs et al., 2007), that reflects the analysis made in the UNEP-SETAC WG Inventory Analysis on LCI modelling: in that document several research needs have been already underlined, with an insight on the future possible direction of the LCA.

5.3.1. General structure of the inventory

ISO 14040 par. 4.3 clearly states that "*there is no single method for conducting LCA. Organisations have the flexibility to implement LCA as established in this International Standard, in accordance with the intended application and the requirements of the organisation*".

This openness at a first glance could be seen as a strength point, because it does not limit a priori any possibility. However, at a deeper look, it represents an open door towards the subjectivity, and a potential threat for one of the basic stated purposes of LCA, i.e. its use in comparative assertion. Indeed, the standard defines the steps to be performed for conducting an inventory, by detailing the concepts and the content behind each element, i.e. Collecting data, Calculating data and Allocation. However, it does not provide any procedural guidance on how to conduct each step.

5.3.2. Consequential LCI

The present focus of the debate is on the question when attributional or consequential LCI method is the most appropriate. The starting point of the consequential approach is its focus on describing the consequences of changes caused by decisions or actions. This affects the methodology far beyond the issues of allocation and marginal data, on which the consequential debate was firstly restricted. Indeed, it represents a new way of conceiving LCA that has consequences for many methodological problems, like the definition of system boundaries, allocation approaches (they are treated separately in Sections 5.2.1 and 5.3.5, respectively), and the selection of data (Tillman, 2000). It can also affect the choice of functional unit (Ekvall & Weidema, 2004) and involve the use of partial equilibrium models experience curves, etc. (see below in this section).

The discussion also focused on what knowledge an LCA generates (Ekvall et al., 2005b) and thus, in which contexts one type of LCA is more useful and appropriate than the other. The opinions are diverging, but in general consider relevant both attributional and consequential LCA, depending on the goal and scope of the assessment. Some researchers consider attributional LCA to be more appropriate for hot spot identification and for developing market claims, such as environmental product declarations (Tillman, 2000). Consequential LCA is considered by others to generate the most relevant information, independently of the application, since an LCA is interesting only if it influences decisions and rational decision-making requires information about the consequences of the decision (Wenzel, 1998). Still other researchers claim that Consequential and attributional LCA are both relevant for hot-spot identification as well as decision-making. Consequential LCA is valid for assessing and reducing environmental impacts, while attributional LCA is valid for assessing and improving the environmental profile of a product (Ekvall et al. 2005b).

Not only the final state resulting from a decision or action can be important in a consequential LCA, but information on the "intermediate" consequences (those related to the process of changes) can also be valuable for the decision-making context.

Hondo et al. (2006) have presented an approach highlighting the importance of the transition phase, so far ignored by the LCA community. The methodology developed makes use of an inter-temporal linear programming model in order to identify the optimal technology configuration to minimize environmental burden over time on a social scale under various socio-economic conditions. The methodology developed deals with the process of changes by explicitly introducing the notion of time: the approach is described as a mathematical model, combining optimization (inter linear programming model) with economic input-output table and scenario development. First, casual relationships between different times were modelled. An inter-temporal model has been used to analyze the step-by-step consequences of a decision, with regard to technology/product selection in the future. Second, a basic scenario in the future was adopted to simulate the state of technology selection using the model. The drawback of this approach is that it requires expertise in the fields of economics as well as optimising models, not always available within the LCA community. This approach has been outlined in a case study, but still need to be improved and further tested.

A consequential LCA often includes **marginal data** that, by definition, represent the effects of a small change in the output of products and/or services from a system on the environmental burdens of the system. The marginal data are relevant to model environmental burdens from all production systems that are marginally affected by possible changes in the life cycle. This is typically true for all large production system, i.e. system with large production volume: the production of bulk materials (e.g. steel, aluminium and polyethylene), energy carriers (e.g. electricity, fuel oil and petrol), etc, systems that commonly appear in the background system and in system expansion. The issue of marginal data is connected to the one of system expansion: system expansion should include the marginal technology, since this is the technology actually affected by the life cycle or decision at stake (Weidema et al., 1998). However, a consequential LCA should include marginal data not only in the system expansions. Normally, marginal data are relevant to model all of the background system, since this is only marginally affected by the life cycle or decision at hand. There are still open questions on:

- *what type of marginal effects (short term or long term) should be included in the consequential LCA.*
  Short-term effects are changes in the utilisation of the existing production capacity in existing production plants. Long-term effects involve changes in the production capacity (Weidema et al., 1998). Energy systems analyses typically focus on the short-term effects. Weidema et al. (1998) suggest that the choice should be decided case by case, but state that the long-term effects are relevant in most cases. Other authors argue that the most complete description of the consequences is obtained if short-term and long-term effects are both accounted for (Eriksson et al., 2007).

- *How to identify the marginal technology.*
  Different approaches have been presented, corresponding to the different opinions on what marginal effect is the most relevant. The short-term marginal technology is identified as the technology with the lowest running cost, where the capacity is not fully utilised. Weidema et al. (1998) present a five-step procedure to identify the long-term marginal technology. Applying this procedure, which is publicly available, can still be difficult for an LCA practitioner, because it requires an analysis of the market and of the economy of different technologies. Such analyses also typically involve large uncertainties and subjectivity. The long-term marginal technology has been identified for several products (e.g., Weidema, 2000), and results from such studies can be used by LCA practitioners, if caution is taken to account for the uncertainties involved.
  The combination of short-term and long-term marginal effects is called complex marginal effects. Complex marginal effects on the Nordic electricity system have been investigated using a cost-optimising dynamic model of this system (Eriksson et al., 2007). But without having available a cost-optimising dynamic model, the identification of complex marginal effects in an LCA is impractical. In principle, the complex marginal effects could be investigated in separate studies and compiled in a database for use in LCA case studies. A drawback is that the complex marginal effects identified in a separate study might not be fully consistent with the LCA where they are applied. Furthermore, the only complex marginal effects that have been published, so far, concern electricity production in Scandinavia.

Despite the vivid debate arisen in the last years, our conclusion is that there is still no consensus neither at the methodological nor practical level. Indeed, no agreement has been reached on when the knowledge generated by attributional and consequential LCA is relevant, and regarding if the consequential LCA methodology is practicable: the identification of the marginal technology is one of the main debated aspects, and a further investigation should be devoted to the analysis of the related uncertainties.

It is clear from the widely differing views on these issues, that there is a need to further investigate the feasibility and relevance of consequential LCA, together with an accumulated experience from successful LCAs. Furthermore, in order to accelerate the development of the methodology, the consequential approach should not be developed only within the international LCA community: indeed, due to the strong implication of the consequential thinking at market level (see following paragraphs), a broader scientific community has to be involved, with focus on economic disciplines.

Most important, as LCA is meant to support real world decisions in business and public policy making, is to involve the private and public sector to ensure meaningfulness and acceptance of LCA in a decision context.

In this context of LCA applications it can be argued to be more important to have a level playing field with fixed recommendations that are applied for a number of years than to have LCA methodology as a constantly moving target that would render LCA unsuitable for real world uses in a stakeholder and market context. Upon that background the harmonisation work of the ILCD Handbook, as coordinated by the European Platform on LCA, should help to fix the presently best-available-methodology recommendation for LCA use in business and public policy context. In the meanwhile R&D should proceed to prepare the next step of further developed recommendations and test them.

Thinking in consequential way means thinking to the consequences of the actions, to the interrelations and thus means to project the problem at market level, with all its dynamics. In this context partial equilibrium modelling, experience curves and rebound effects become relevant and they will be discussed in the following subsections.

5.3.2.1. Partial equilibrium modelling

Partial equilibrium modelling is an important component of the consequential approach because it allows the introduction of market mechanisms in LCA models by describing the balance between supply for and demand of specific products, i.e. the effects of changes in the life cycle through the concept of own-price elasticity of demand and supply. Indeed, an increase in the use of goods in the life cycle typically can contribute to increasing the market price of these goods, and this not only stimulates the production of the goods, but also reduces the use of the goods in other life cycle (Ekvall & Weidema, 2004): partial equilibrium modelling describes and quantifies both of these effects.

Outside LCA, partial equilibrium modelling is an established tool in the area of environmental economics: it has been used to assess the environmental consequences of policy decisions and other strategic decisions. Nevertheless, in the framework of LCA its introduction poses several problems, both at conceptual and practical level, and still an answer to these fundamental questions is missing:

- *Should partial equilibrium models be integrated into LCA, or used in parallel to LCA?*
  It is evident that, ideally, the integration of partial equilibrium models into LCA "would result in a new tool with specific advantages with regard to modelling the consequences of changes" (Ekvall & Weidema, 2004). Furthermore, partial equilibrium modelling adds to the complexity and cost of the LCA, and when the complexity increases, the risk for calculation errors, the degree of subjectivity, and other mistakes also increases. The study also becomes more difficult to interpret and report in a transparent manner, adding to the uncertainty of the LCA.
  From the practical viewpoint, the modelling could be facilitated by calculating the elasticities of supply and demand for the most important markets of

recovered material (Ekvall, 2000). However, price elasticity estimates currently seem to be available in the literature for few goods only (Ekvall & Andrae, 2006) and using literature data increases the uncertainty, because own-price elasticities depend on case-specific factors such as time and place, and on whether the study focuses on short-term or long-term effects.

The marginal alternative use of materials outside the life cycle investigated is perhaps environmentally not very important in many cases, but more case studies are required to test this hypothesis (Ekvall & Andrae, 2006).

- *For what types of goods does a change in demand in a life cycle affect the demand in other life cycles?*

  Regarding the application fields of partial equilibrium models in LCA, the opinions are differing. Weidema (2003) argues that they are not applicable for goods that are traded on competitive, unconstrained markets (i.e. where there are no market imperfections and no absolute shortages or obligations with respect to supply of production factors, so that production factors are fully elastic in the long term). Ekvall (2000) suggests that partial equilibrium models of scrap markets can be applied to handle cases of open-loop recycling. Ekvall & Andrae (2006) use a semi-quantitative partial equilibrium model of the lead market to investigate the effects of a global shift to lead-free solders. Lesage et al. (2007) use a partial equilibrium model of the housing market to investigate the land-use effects of a case of brownfield rehabilitation.

- *When partial equilibrium modelling is relevant?*

  It is also an open question when partial equilibrium modelling is relevant. Weidema (2003) argues that long-term prices are in many cases not affected by demand, but determined by the long-term marginal production cost. This holds on competitive, unconstrained markets. When the price is not affected by the demand, an increase in the use of a good in a life cycle does not affect the long-term use of the good in other life cycles. In such cases, the use of a partial equilibrium model is irrelevant. The question is how common these cases are, and when they occur.

Despite the interest they deserve in the LCA community, due to their implication in terms of deepening and broadening the present modelling capability in LCA, still a lot of efforts are necessary to combine LCA and partial equilibrium modelling. If consensus is reached at the conceptual level, making the combination of LCA and partial equilibrium models attractive, the price elasticity of demand and supply has to be estimated for many more products; these data should be compiled in databases that are posted in connection to ordinary LCI databases.

5.3.2.2. Experience curves

A decision to invest in a new technology can have a large impact on its future. As the accumulated experience on the technology increases, subsequent investments in the technology become less expensive and, hence, more likely. Over time, a snowball effect can occur, giving the technology a much higher market share than it would have had without the initial investment (Ekvall et al., 2005a). If this effect is taken into account, it can largely affect the LCA results.

The fact that technologies tend to be more efficient and perform better environmentally can be relevant in both consequential and attributional LCAs. It is particularly relevant for case studies with a long time horizon and case studies with new and immature technologies. In this context, learning and experience curves play an important role, because they make it possible to estimate the possibly huge environmental effect of investments in new technologies.

Learning curves describe how an individual or production process becomes more efficient as a function of the accumulated experience (Wright, 1936). The curve is a log-log function with a constant progress ratio: for each doubling of the accumulated experience, the time and/or cost required for a process is reduced by a specific percentage (typically 5-35%). The initial time and cost and the progress ratio are empirically measured. The learning curve can then be extrapolated into the future to estimate further expected improvements. Experience curves is a similar concept, but on a higher system level: they describe how a technology becomes less expensive as the production of the technology grows more efficient with accumulated experience (Claeson, 2000). The term "learning curve" is sometimes used to denote both of these concepts.

It has been demonstrated that not only the cost but also the efficiency (Claeson, 2000) and environmental performance (Pento & Karvonen, 2000) of a technology improves with accumulated experience. This development can also be described using experience curves.

The accumulated experience is not a perfect basis for estimating future improvements. Improvements occur not only as a result of investments in the specific technology but also through learning from other, related technologies, and from research. It might be possible to define a better basis for estimating future improvements, but no such suggestion has been found.

There are several open questions related to the use of experience curves in LCA, mainly related to the following aspects:

- Use of experience curves for making *forecasts of future emissions*. In this case, it would be necessary to make assumptions or forecasts about the future investments in the technology, but how to make these forecasts is still debated.

- Use of experience curves *for estimating the effect of investments in new technologies*.
  Two approaches have been presented so far. Ekvall et al. (2005a) suggest using a dynamic model of the market where the technology has a share. This model has market-share functions that describe how the market-share of the technology depends on the difference in cost compared to other technologies. Such a model gives an estimate of the consequence of an individual investment in any of the technologies on the market. The drawback of this approach is that it requires expertise not only in experience curves but also in dynamic modelling. This approach has been outlined in a pilot study only, thus real case studies are required to test and illustrate this approach. Sandén & Karlström (2007) use a simplified approach that does not require advanced modelling but, on the other hand, it is not accurate in estimating the consequence of an individual investment. Instead, it allocates the impacts of the total learning investment in proportion to the size of the individual investment. They estimate:
    - the total learning investment, i.e., the investment that is required to make the new technology economically competitive with the traditional technology on the market,
    - the environmental impact that occurs if the new technology takes over the market, and
    - the probability that this will occur.

  Then they divide the estimated impact by the total learning investment, and multiply by the probability that the new technology will take over the market. The result is presented as the estimated consequence of a unit investment.

- *If the concepts of experience curves and learning investments should be integrated into the LCA and if so, what approach should be used in what circumstances*. Experience and learning curves make a consequential LCA more comprehensive and accurate in the aim of describing how the environmentally relevant flows from the technological system change, but on the other hand, they increase the complexity and cost of the study, while reducing the precision of the results.

- *Feasibility*
  The use of experience curves is feasible and adds value to LCA in several circumstances:
    - It adds value to estimate the future consequences of investments where effects on technological development are expected to be very important for the results and conclusions. This is typically the case of long time horizon and with new, immature technologies in important parts of the life cycle.
    - The use of experience curves is more easily defended for technologies where the progress ratio has already been estimated. Defining the

progress ratio as part of an LCA project will often be considered too difficult and expensive.

The issue of experience curves and their relation with LCA still need to be debated, both at conceptual and practical level. Further research is needed on how transfer of knowledge and experience among technologies and among geographical regions should be accounted for when applying experience curves (Claeson, 2000), and on how knowledge generated through research, as opposed to investments, should be accounted for.

From the practicability viewpoint, to make the combination of LCA and experience curves feasible, experience curves need to be established for more technologies. For each technology, the experience curves can be uniquely described through:

- the cost, efficiency, or environmental performance at a specified starting time,
- the accumulated experience at this starting time, and
- the progress ratio.

These data can be compiled in databases that are posted in connection to ordinary LCI databases, to make the experience curves available to LCA practitioners.

5.3.2.3. Rebound effects

Assuming that the functional output from the life cycle is unaffected by changes in the life cycle is a simplification, because a change that affects price, quality, functionality, need for maintenance, etc. of a product, can influence the demand for this and for other products. All of these effects are often called "back fire", "take-back", "offsetting behaviour" or, as we denote them, rebound effects (RE). Some authors suggest that it might be more adequate to talk about *ripple effects* (Hertwich, 2005) because it is not always easy to know in advance if these indirect effects increase or reduce the environmental impacts. If, for example, a product becomes cheaper, allowing consumers to spend money on other products, this is likely to increase environmental impacts (Eyerer & Wolf, 2000; Thiensen et al., 2007). However, if this money is spent on, e.g., increased insulation of a house, the environmental impacts are reduced (Hertwich, 2005).

These effects have been much discussed and analysed in the context of energy system analysis (Greening et al., 2000; Weidema, 2003), but their applicability in LCA is still an open question, because of the complexity, uncertainty and costs involved. The aspects on which consensus still has to be agreed on are the followings:

- *The definition of rebound effects*.

  In the literature analysed, all these effects have been identified as rebound effects:

  - **RE1**. The demand for a specific product, and hence its production, can increase when a change in the life cycle reduces the price of the product (Greening et al., 2000) or when it enhances the quality or function of the product (Weidema, 2003).

- **RE2**. A reduction in the price of a specific product X (e.g., a car) can also affect the demand for other products. The demand for competing products (e.g., a car of a different brand) decreases, because they become less competitive compared to product X (Sanden & Karlström, 2007; Greening et al., 2000). This offsets part of RE1. On the other hand, the demand for complementary products (e.g., the specific fuel used by the car) increases when the demand for product X increases.
- **RE3**. When the price of a product X is reduced for a consumer or industry, more money is available for this consumer or industry to spend on other products. Hence, the demand for other products in general can increase (Greening et al., 2000; Thiesen et al., 2007). This adds to RE1.
- **RE4**. When the price of a specific product used in an industry is reduced, this industry can reduce the price of its products. This can result in an increase in the use of these products, which means the industry grows. Consumers and industries where these products are used benefit from the secondary price reduction, and so on. Through this chain of cause and effects, the initial price reduction is likely to contribute to economic growth (Sanden & Karlström, 2007; Greening et al., 2000). This also adds to RE1.
- **RE5**. A change in the price of a product can affect the technology development, consumption preferences and societal institutions (Greening et al., 2000).
- **RE6**. The demand for a specific product can increase when a change in the life cycle makes it less time-consuming to use and maintain. In addition, this will give consumers more time to spend on other kinds of consumption and other activities (Hofstetter et al., 2006).
- **RE7**. The demand for a specific product, and other products, can also increase when a change in the life cycle makes the product smaller, easier to handle, etc. (Hofstetter et al., 2006).
- **RE8**. The demand for products can increase to compensate when a change in the life cycle reduces the quality or function of the product (Weidema, 2003). When this happens to an industry, it is likely to reduce its output of products, reducing economic growth, and offsetting part of the increase in purchase to the industry.
- **RE9**. The demand for products might increase to compensate when a change reduces the fulfilment of needs and/or reduces the happiness of consumers (Hofstetter et al., 2006).

▪ *Methods to quantify them.*

Different approaches can be appropriate for quantifying the various ways in which the functional output from the technological system can be affected, however, in literature they are available only for few of the above listed rebound effects, in particular:

- The own-price elasticity of demand. It is a useful concept to analyse how a change in price for a product affects the demand for this specific product. (RE1).
- The cross-price elasticity of demand. It can be used in analyzing how a change in price for a product affects the demand for a small number of other (competing and complementary) products (RE2).
- General equilibrium models, whenever we wish to estimate the effect of the price of a product on the general demand for other products (RE3) (Thiesen et al., 2007) or the effect of the price of a product on the economic growth (RE4) (Greening et al., 2000).
- The concept of marginal consumption, to analyse the consequences of a change in the money available for consumers to spend on other products (RE3).
- The relation between the fulfilment of needs and the enhancement of happiness of consumers can be analysed using a step-wise procedure presented by Hofstetter et al. (2006) (RE9).

For the other rebound effects, approaches to quantify the effect have not been found.

- *The uncertainties associated to their evaluation and the increased complexity*. Even when established tools such as price elasticity and CGE models can be applied, the quantification of rebound effects depends on important subjective methodological choices that add considerable uncertainties to the evaluation of their magnitude (Greening et al., 2000; Hertwich, 2005). Furthermore, accounting for rebounds effects adds to the complexity, with all the consequences already mentioned.

It is evident that many questions on rebounds effects are still open, especially from the practical viewpoint. They make a consequential LCA more comprehensive and accurate in the aim of describing how the environmentally relevant flows from the technological system change, but, on the other hands, they add complexity and cost to the study, as well as reducing the precision.

The question of rebound effects is still a large field of research in the sense that it includes various types of cause-and-effect relations that need to be further investigated. It is also methodologically immature: for several rebound effects, no method has been found to quantify the effect. For other effects, only a single, first approach has been presented. As the rebound effect can sometimes offset part of the direct effect and at other times reinforce the direct effect, it might be questionable whether the term “rebound” is appropriate or whether a new terminology should be introduced. Substantial further research and development is also required even when established tools such as price elasticity and CGE modelling can be applied.

A common approach can be adopted for reducing complexity and cost related to the application of partial equilibrium models, experience curves and rebound effects: to focus on the parts of the technological and economic system where the most important consequences are expected to occur. This means that the life cycle model is reduced to include only the flows and processes where changes are expected to be important for the results and conclusions. If that is the case, the approaches are applied only when effects on the functional output are expected to be very important for the results and conclusions.

5.3.3. Time introduction in LCI modelling

The treatment of time turns out to be an important task of modelling in the inventory phase.

Historically, time has been ignored or assumed to be infinite. LCI is based on a steady state linear equilibrium model, i.e. a model that indicates a hypothetical equilibrium situation with *ceteris paribus* assumptions. This means that no technology will change; no adaptations other that supply-demand matching will take place; and equilibrium will be reached (Huppes et al., 2006 draft). Changes in time are ignored because time is outside the model: this could be an acceptable simplification if the situation develops gradually without interruptions and quick changes are not foreseen.

Looking at the future is the main task of policy makers, thus the analysis should allow considering the future boundary conditions like legal regulations, changes in technology, i.e. it should allow considering the time dimension in a proper way.

In the last years, several attempts to introduce time dimension in the LCA modelling have been proposed as a reply to the apparent limitation of ISO LCA structure, but the applications are still controversial.

This review analyzed the approaches related to time introduction in LCI both in terms of dynamic evolution of time and scenario analysis (the latter is treated in subsection.5.3.1.1), considering the time as a dimension linked to the future states.

5.3.3.1. Dynamic LCA

The term "dynamic LCA" in this review refers to the inclusion of time dimension in LCI, both in terms of continuous mathematical function ("true" dynamic) and discrete intervals ("quasi" dynamic). Indeed, the terminology used in LCA literature is sometime confusing and misleading as some authors use the word "dynamic" as synonym of dependence on chosen parameters rather than of "changes over time".

This is, for example, the approach presented by Kendall (2004), in which the proposed LCA model is considered dynamic due to the dependence on the chosen design parameters. The model allows comparing the sustainability of alternative concrete bridge deck designs, and broadens the scope of bridge LCA studies by accounting for the dynamic nature of the interlinked bridge and traffic system.

The timeline can be changed to reflect changes in assumptions regarding the life of the bridge deck, the durability of repairs and for sensitivity analysis. The idea is that an optimal formulation of materials can be developed through iterations in material formulations and their associated LCA. The LCA computer model developed is tailored to a specific bridge deck application, but its dynamic nature makes it applicable also to other infrastructure systems such as roadways and concrete pipe.

The other approaches selected for the review share the definition of "dynamic LCA" chosen for this review. In particular Pehnt (2006) developed a dynamic LCA in which the future state of the system is modelled extrapolating into the future those parameters that are environmentally relevant and at the same time show a significant time-dependency.

Yokota et al. (2003) presented an approach in which LCA and Population Balance Model (PBM) were integrated to quantitatively assess the total environmental impacts induced by the product population in a society over time. They introduced time as a critical parameter and employed all the products in a society as a proper unit of analysis to assess the environmental impacts of a technology/product, while not explicitly dealing with socio-economic aspects.

Fischer & Pflieger (2007) have presented the inclusion of time as a continuous mathematical function by developing a time-dynamic parameterized LCI-model, i.e. the modelling of time-related aspects by considering dynamic changes in physical/technical system. The model comprises three steps:

i) the parameterized modelling, i.e. the identification of relevant parameters in LCI: these are parameters with a high environmental relevance and important technological changes within short time periods.
ii) The development of time series for parameters processes and systems to be used in parameterized models.
iii) The development of prognosis functions for the parameters. This means that for each point of time, the inventory quantities can be calculated as a function of time dependent parameter: thus, each parameter is described by a continuous mathematical function.

Various statistics and existing databases provide a good basis to implement key categories like energy production, material production or transportation for a broad period, nevertheless the major limitation is the lack of data. LCI(t) is possible but challenging and existing tools have to be adjusted. Further research should be directed towards the identification of industry sectors where time-related issues are relevant to LCA results and the realization of data preparation for relevant processes there.

"True" (i.e. in terms of continuous mathematical function) dynamic approaches are still pioneering, and several efforts are still necessary both at methodological and practical level: indeed, the available software tools do not reflect advances in modelling because they are based on static relations, and are not supported by databases that could be representative of the future situation.

Research lines should consider what knowledge is added to LCA from dynamic models: maybe spending resources on developing models in which time is modelled as a continuous mathematical function could be unfruitful, since the use of recursive analysis in different time frames with data representing future technological relations could give a valid answer as well. On the other hand, the issue of distribution of impact over time would represent a more interesting field of analysis, at present not covered by the literature analysed in this review.

Then, an accurate balance should be found between the need of having an optimum representation of the reality and the complexity/feasibility of the modelling itself. For decisions related to the long period, the use of scenarios could be more relevant and feasible: on this aspect, efforts should be spent developing technological scenarios related to the main processes.

5.3.4. IOA and hybrid approaches

With the term “hybrid approaches”, we indicate both the use of physical and monetary units, and the integration of sector and process level data. In LCA domain, the term refers to the combination of process-based LCA and environmentally extended input-output analysis (IOA), in order to overcome the limitations and combine the advantages of both methods.

The benefit of IOA[5] is represented by its top-down nature: by allowing the entire economy to be considered as a system under study, it can provide the study with a potentially higher degree of completeness and details in system boundaries. Indeed, process analysis based LCI is always truncated to a certain degree (while in theory this may only be 1% if cut-off rules are properly applied, but see also section 5.2.1 b), since it is practically not viable to collect process-specific data for the whole economy. At the same time IOA limits strongly the effort for LCI data collection and this has led to the use of IOA in LCI.

IOA hybrid approaches in principle are not in contrast with ISO standards:

i) ISO series defines the framework without specifying which computation method has to be used, therefore, both process flow diagram and matrix representation are considered to be compatible and thus the use of an input-output model to describe (part of) a product system is not precluded;

ii) Hybrid techniques using input-output analysis seem to be almost the only practice compatible with ISO practices, due to the high requirements of the standard for the system boundary selection (Suh & Huppes, 2005).

[5] Economic input-output analysis is a top-down technique that uses sector monetary transaction matrixes describing complex interdependencies of industries within a national economy and is a suitable approach for LCI (Suh & Huppes, 2005).

On the other hand it can be argued that the inherent methodological approach of EIO to price-allocate across whole industry sectors and hence across not physically connected production plants and process chains may not be in line with the Allocation requirements of ISO 14044.

In literature, under the headline "IOA and hybrid analysis", several approaches are classified, but they are not always consistent among them. In this review, under this term we considered the following approaches:

- Input-output based LCI
- Physical input-output analysis
- IOA hybrid LCI
  - Tiered hybrid analysis
  - IO-based hybrid analysis
  - Integrated hybrid analysis
- Waste input-output analysis.

5.3.4.1. Input-output based LCI

Input-output based LCI is the approach in which process-based LCA can be made more complete by adding EIO data for more remote parts of the system.
IOA and LCA have a lot in common, but there are also important differences: in LCA there are no annual 'transaction' records available, quantities are in physical units, there are physical flows instead of money flows, use and end-of-life stages are considered, LCA primarily concerns the function of a system etc. (Suh, 2004a).
At the conceptual level, the advantages of the combination of LCA with IOA are evident and refer to:

- a higher completeness regarding upstream system boundaries (while process-based analyses are generally superior for other system boundaries),
- the inclusion of information on the environmental aspects of a commodity, using less resources and time, and
- their smaller data requirements, assuming that IO-based LCIs are available (Suh & Huppes, 2005), while this applies only to a limited number of countries.

Several case studies exist that apply the IOA framework to LCA, and new approaches are moving towards the setting out of input-output analysis combined with LCA for analyses with a regional resolution. The main feature of region-based LCA is that it can consider the structural and environmental characteristics of regions that are directly and indirectly affected by regional activity, but it requires a greater amount of time, cost, and work for the inventory analysis.
Starting from the assumption that both process-LCA and EIO-LCA are important decision making tools, but neither of them can perform regional- and state-level analyses efficiently, Cicas et al. (2007) and Yi et al. (2007) proposed two different approaches to this problem.

In Cicas et al. (2007) a regional US LCA model, Regional Economic Input-Output Analysis-based Life-Cycle Assessment (REIO-LCA) has been developed: it enables regional (multi-state in the US) and state-level analyses and allows decision-makers to estimate both the economic and the environmental implications of changes in a regional economy. The national model is based on the US 491 by 491 economic input-output model, and uses sector energy consumption and emission factors to approximate the environmental effects of production and services.

The model proposed in (Yi et al., 2007) is a region-based LCA method called LCRAM[6], which can reflect the differences of regional characteristics[7] for direct and indirect effects of regional activities by using IO table. First, the authors constructed a regional database that includes an Interregional Trade Matrix (ITM), Regional Environmental Burden Coefficients (REBC), and Regional Damage Factors (RDF). Second, they developed the Expanded Interregional Input Output Method (EIOM), which can identify the Emitting Regions[8] for indirect effects

The proposed LCRAM has several advantages, like:

- the ability to reflect the characteristics of each region for the direct and indirect effects, through all stages of activity;
- the ability to quantify the interdependent effects and transportation effects due to interaction among the regions which have not been reflected in conventional region-based LCA;
- it enables users to apply a regional evaluation for many more regions, and
- It makes available details of industry classification that has been impossible to reflect with an existing Multi-Regional IO method.

Still limitations exist in its applicability, due to the limited number of environmental burdens considered, and – mostly – due to the lack of regional databases.

Despite the large number of case studies performed on Input-output based LCI, there are still several limitations, not overcome by the present approaches:

- Approximations: the product of interest is approximated by its commodity sector, that is a broad aggregate including a large number of products; and the environmental burden coefficients of imports are assumed to be well-approximated by the corresponding domestic industry sector. This may introduce errors in analyses of products with high import content and very different foreign production technologies. For a commodity whose product system heavily relies on imports and newly developed technologies, however, applicability of IO based LCI methods is rather limited (Suh & Huppes, 2005).

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[6] Life Cycle Region-specific Assessment Method (LCRAM).

[7] Including structural (regional production and consumption, interregional trade, and the structure of energy consumption) and environmental features (geographical location, climate, natural conditions, and population density).

[8] The term „Emitting Regions“ is used to describe regions where environmental burdens are emitted (Yi et al., 2007).

- IOA method itself can provide LCIs only for pre-consumer stages of the product life cycle, while the rest of the product life cycle stages are outside the system boundary of IOA (Suh & Huppes, 2005).
- High uncertainties, rising from methodological assumptions and data (Nielsen & Weidema, 2001).

In terms of practicability, data availability and lack of consistency among the different input-output tables available (due to the different resolution) hamper the consistency of the methods. Data of IO-based LCI is normally older than process-based one, since it takes 1 to 5 years to publish IO tables based on industry surveys (Suh & Huppes, 2005), while the integrated environmental data is typically even older, as was argued more above.

The most mentioned tools for IO-based LCIs are EIOLCA and MIET. EIOLCA (Economic Input-Output Life Cycle Assessment) is a web-based IO-based inventory calculator that provides the amount of water usage, conventional pollutants emission, global warming gas releases and toxic pollutants emissions per sector output in monetary unit (Suh & Huppes, 2005). MIET (Missing Inventory Estimation Tool) is a software tool designed to assist IO-LCA and tiered hybrid analysis, which combine the strengths of process-specific LCA and IOA. It has been developed using the US input-output tables and various environmental statistics based on a consistent commodity-by-commodity framework, and it includes also capital goods (or investments) in the monetary transactions among the input-output sectors. Entering the estimated price of a missing flow either in the producer's or the consumer's price, MIET supplies inventory results for missing flows as well as characterised results, using around 100 different impact assessment methods that are in common use (last version of MIET: 2004) (Suh & Huppes, 2002).

Considering the present state of the art and the inherent limits of the approach analysed, we could consider the IO-based LCI a way to estimate missing data in process-based LCA, based on EIO data; other approaches discussed are to fill such gaps with process-based data estimates. From the methodological point of view, the natural evolution of this approach is represented by the hybrid LCI, intended as a framework in which detailed information at unit processes level in physical quantities is fully incorporated into the input-output model, which in turn represents the surrounding economy that embeds the process-based system (more information can be found in par. 5.4.4.3.). In the case of hybrid LCI, the problem of complexity and time-consumption of analysis still exists: efforts should be concentrated towards the simplification of the analysis in order to make it more practicable. Therefore, despite the present shortcomings, this line is relatively straightforward and it is readily available.
On this side, further efforts should be spent on three main aspects:

- Data reliability
- Uncertainty evaluation

- Find approaches to overcome the inherent methodological shortcomings identified above

If the methodological issues can be overcome, the building of reliable and publicly available environmental intervention databases for the input–output tables would be a very important next step. Although a number of national and international projects are in progress to incorporate environmental variables in national accounts, the number of pollutants covered and the resolution of the commodity classification are rather limited for LCA purposes. While this method would be particularly useful for the analysis at regional level, this is hampered by the lack of regional data on production and environmental emissions.

Regarding the uncertainty evaluation, no research lines seem to exist on uncertainty factors for IO data but this will be of great interest, due to the repercussions of the methodology on large-scale application.

5.3.4.2. Physical Input-Output Analysis

If the use of monetary input-output analysis integrated into LCA or in combination with LCA framework has several advantages, mostly in terms of system boundaries and data availability, nevertheless the use of monetary values implies that the analytical results obtained may be vulnerable to price fluctuations and unhomogeneity (Suh & Nakamura, 2007). Thus, an important contribution may come from the physical input-output (PIOT[9]) approach.

The PIOT is parallel to and fully consistent with traditional MIOT but all flows are reported in physical units. They describe changes in the natural environment caused by human activities that deserve less attention in traditional IOT because of the insufficient possibilities of a monetary valuation, like using of natural assets as source of raw materials and as sink for residuals (Suh, 2004b). A PIOT is not simply a unit conversion of a MIOT and cannot be derived only by multiplying the MIOT with a vector of prices per tons for each sector. This is mainly due to aggregation of non-homogeneous products/sectors into one category: the higher the aggregation level, the more notable become the differences between MIOT and PIOT.

The debate surrounding PIOTs has so far mainly focussed on methodological issues surrounding the treatment of wastes in physical input-output models: in particular, PIOTs have been applied to calculate the amount of land appropriation due to exports (so far all land related studies presented used MIOTs for attributing land to the different category of final demand) (Hubacek et Giljum, 2003).

[9] Physical input-output tables (PIOTs) describe the flows of material and energy within the economic system and between the economic system and the natural environment (Wiedmann et al., 2006). While the Monetary Input-Output Tables (MIOT) describes only internal flows of the economy, however, the PIOT also reports the exchanges taking place with the natural environment. Unlike the whole economy, individual industries have as sources of matter not only extraction and imports, but also other industries of the same economic system. Similarly, the outlets of each industry's production are not only the environment, the stocks and other economies but again other industries, and also consumers (Femia & Moll, 2005).

In particular, the authors computed the direct and indirect land requirements for the production of exports from EU-15 to the rest of the world and compared them to input-output models using coefficient matrices derived from monetary tables (MIOTs) and a vector of physical factor inputs per unit of output (Weisz & Duchin, 2006). They arrived at substantially different numerical results concerning overall and sectoral land appropriation for exports.

The choice of using PIOT instead of MIOT is due to the fact that PIOT illustrates land appropriation in relation to the material flows of each sector, which is more appropriate from the point of view of environmental pressures than the land appropriation in relation to monetary flows of a MIOT; furthermore, the most material intensive sectors are also the sectors with the highest land appropriation.

The debate arose on this issue, in particular on:

i) the aspects of mass balance inconsistency,
ii) the violation of the fundamental assumption of input-output economics that each sector produces a homogeneous characteristic output (Suh, 2004b).

In any case, the differences of results cannot be attributable to the superiority of the physical model, but to the (methodological) assumptions made to reflect real world conditions.

We should consider that the debate surrounding PIOTs is still very young: only a few PIOTs have been compiled to date and no standard methods for the compilation of PIOTs have yet been developed due to the relatively young history of physical input-output accounting (Suh, 2007).

Three are the main obstacles to the application of this approach:

i) the very limited and restrictive data availability (Hubacek & Giljum, 2003). PIOTs have so far been compiled only for a very small number of countries;
ii) the lack of a standardised methodology. Concepts, definitions and classifications are a theoretical reference point only. Existing PIOTs differ with regard to the number of sector reported, the disaggregation into product groups, as well as the inclusion or exclusion of specific materials: flows from/to the natural environment do not have a monetary counterpart so that there is no reference rule in National Accounting for their identification/classification. Moreover, a discussion about which materials should be separately balanced is currently missing at the theoretical level (in particular, the inclusion or exclusion of water and air dramatically changes the structure of the interindustry tables and as a consequence also the results of input-output analysis based on these tables) (Hubacek & Giljum, 2003).
iii) Finally some of the methodological problems of MIOTs also apply to PIOTs.

At first sight, the use of PIOT could be considered a good approach: indeed, using physical quantities, the accounting framework can be free from the price's non-homogeneity and fluctuation and from different taxation schemes and subsidies, which may distort the actual physical flows between industries when a monetary unit alone is considered (Suh, 2004b).
Nevertheless, the methodological limitations are still numerous and it is a case specific question whether the benefits of using a PIOT outweigh its disadvantages (Suh, 2004b). Indeed, PIOT data are very detailed and as such, they do not seem to be very useful for policy makers. Therefore, before investing resources in their development, a deeper evaluation should be done on the effective use of the PIOTs data, according to the different potential users.

5.3.4.3. Integrated Hybrid Analysis

As we have already explained in the previous paragraphs, so far hybrid analyses have been the simple sum of process analysis and input-output based analyses. Indeed, the term *hybrid analysis* has been used with different meanings, referable to not only IO-LCA but also to tiered hybrid analysis, IO-based hybrid analysis and integrated hybrid analysis (Suh & Huppes, 2005). While these overcome some of the problems named above, the inherent methodological issues of the EIO data still remain to be solved, as the data is used in the hybrid calculations.

Tiered hybrid analysis uses process-based analysis for the use and disposal phase as well as for several important upstream processes, and then the remaining higher order input requirements (e.g., materials extraction and manufacturing of raw materials) are imported from an IO-based LCI. Tiered hybrid analysis can be performed simply by adding IO-based LCIs to the process-based LCI result; the data traffic is one way: EIO data into the process-based framework.
Tiered hybrid analysis is meant to provide reasonably complete and relatively fast inventory results. However, the following aspects should be considered:

- the border between process-based system and IO-based system should be carefully selected, since significant errors can be introduced if important processes are modelled using the aggregated IO information;
- there are some double-counting problems in tiered hybrid analysis. In principle, the commodity flows of the process-based system are already included in the IO table, so that those portions should be subtracted from the IO part;
- the tiered hybrid model deals with the process-based system and the IO-based system separately, so that the interaction between them cannot be assessed in systematic way. For example, the effects of different options at the end of the product life cycle, which can change the industry-interdependence by supplying materials or energy to the IO-based system, cannot be properly modelled using the tiered hybrid method.

The **IO-based hybrid approach** is carried out by disaggregating industry sectors in the IO table when more detailed sectoral monetary data are available (in this way, detailed process-specific data can be fully utilized without double counting). Inventory results for the remaining stages of the product life cycle, including use and disposal, should be added manually. Since this approach partly utilises the tiered hybrid method, the interactive relationship between pre-consumer stages and the rest of the product life cycle is difficult to model.
The disaggregation procedure is the most essential part of IO-based hybrid approach. Joshi (2003) suggested using existing LCIs for information sources of detailed input requirements, sales structure and environmental intervention.

These approaches have shown their limits, and lead to the conclusion that relying only on IO data for detailed LCA is not desirable at all.
The integration between EIO and LCA should be a “harder” approach, relaying on stronger bases, able to efficiently overcome the limitations of the previous attempts: a promising example is represented by the **integrated hybrid analysis**, since it allows combining the two systems with different resolution in a consistent framework (Suh, 2007). It is a hybrid model that integrates the computational structure of an LCA with an input-output analysis within a consistent mathematical framework throughout the whole life-cycle of a product. Integrated hybrids analysis relies on full process analysis, and then utilises IO-based LCI only for cut-offs. (Suh et al., 2004): detailed information at unit process level in physical quantities is fully incorporated into the input-output model, which in turn represents the surrounding economy that embeds the process-based system (Suh & Huppes, 2005). This approach enables a consistent allocation method throughout the hybrid system and avoids double counting by subtracting the commodity flows in a process-based system from the input-output system.
Integrated hybrid analysis has clear advantage in terms of the quality of the results, especially as regards the system completeness. With information on the monetary value only for cut-off flows and with improved availability of environmentally extended IO data, preferably regionalised, the additional data requirements and the added complexity both may become quite limited. This seems the best choice for the future, if not for now yet; however, it adds to the cost of already expensive and time-consuming full process LCA. Nevertheless it has to be reiterated that also this approach does not overcome the methodological weaknesses of the EIO data and that other approaches for achieving the data completeness may be found e.g. by using data estimates using process-based LCI.
Indeed the practical implementation is not yet publicly available, and hybrid LCA is still considered a complex tool. Efforts should be focussed on two different levels:

- *methodology (see also other issues listed for Input-output based LCA in chapter #5.3.4.1)*

  IO techniques present still unresolved issues for which efforts should be made:

  - as international trade has grown to be an integral part of the global economy, the applicability of a national input output table and analysis is becoming increasingly limited for addressing global challenges;

- lack of comparability between IOTs of different countries;
- quantitative uncertainty analysis has rarely been attached to IO-LCA, as basic uncertainty information for individual elements of an IOT is generally unavailable.

Research efforts should focus on addressing the underlying methodological issues and subsequently developing well-structured input-output tables, with increased environmental data: this means the development of better statistics on environmental emissions and resources use that can be used in input-output LCA. This could imply also that environmentally important industry sectors are not classified within another aggregate, but separately, in order to reduce the allocation error. If the inherent methodological issues of IOA can be overcome, in the long term the development of a multinational environmental input-output model with complete trade links would be very desirable. This would be especially in connection with regionalized LCIA methods that will result in a complete system with regional specification[10] (Suh et al., 2004). The EXIOPOL (A New Environmental Accounting Framework Using Externality Data and Input Tools for Policy Analysis) project is working on this issue: among the several goals of the project, one is related to setting up an environmental extended (EE) Input Output (IO) framework for the EU-25 in a global context, in order to take pollution and externalities embedded in imports to the EU25 into account, and also to analyse the effects of sustainability measures taken in Europe on the economic competitiveness of the EU25.

- *practicability*

  Beside the drafting of more case studies, and the dissemination of useful findings, since most commercially available LCA software is not able to handle matrix inversion for LCI computation, a software tool development that enables hybrid analysis would also required. (Suh & Huppes, 2005).

If the methodological issues of EIO can be overcome, with the hybrid analysis, we may in future be able to combine process-based LCA and IOA in order to overcome the limitations (mainly incompleteness due to cut offs and lack of process specificity) and combine the advantages (high level of detail and completeness in system boundaries) of both methods, but when waste treatment methods have to be evaluated in the analysis, a more tailored method is needed[11]. Within the hybrid LCI, the **Waste Input-Output (WIO)** analysis deserves particular attention. The WIO is a hybrid methodology that takes into account the interdependencies between the flow of goods and waste, where the technology matrix of a product system in LCA is fully integrated with technical coefficients matrix of an economy in IOA.

[10] A multi-region input–output framework should be set up including national data on resource use and pollution in order to reduce the error associated with the imports assumption.

[11] Indeed, the conventional IOA was originally developed to represent the intersectoral flow of goods and hence is not designed to take into account the flow of waste associated with it.

WIO is capable of taking into account all the phases of life cycle, production, use, and end of life (EoL) and allows taking into account the interdependencies between the flow of goods and wastes[12], by establishing a correspondence between waste generation and its treatment at the level of each sector (Nakamura & Kondo, 2002). The model has two main features:

i) it expands the Leontief environmental input-output (EIO) model with respect to waste flows. It turns out that the EIO model is a special case of the WIO model in which there is a strict one-to-one correspondence between waste types and treatment methods, providing a general framework for LCA of waste management:

ii) it takes into account the dynamics of waste management treatment, by incorporating an engineering process model of waste treatment (a model of individual processes describing the quantitative relationships between characteristics of waste feedstock, inputs of utilities and of chemicals, specification of the equipment and the generation of treatment residues[13] and by including an allocation matrix (a matrix that provides a means to represent a change in the allocation of waste to different treatment methods) (Nakamura & Kondo, 2002).

The model shows a high flexibility (Kondo & Nakamura, 2004):

i) it can treat an arbitrary number of waste type and treatment methods;

ii) it can handle the case in which several treatment methods are jointly applied to a single type of waste (and vice versa), provided that the combinations are technically feasible;

iii) it can take account of waste generated from virtually any waste source in the economy (including: municipal solid waste (MSW) from final demand sectors; industrial and commercial waste from the goods and service producing sectors; treatment residues from waste treatment sectors);

iv) it provides a means to evaluate some aspects of life cycle costing of waste management scheme through the use of the indicator of total employment:

v) it can be considered as a first order measure of the economic efficiency of a particular waste management scenario.

The authors presented also an analytic extension of the waste input output model, based on the method of linear programming (Kondo & Nakamura, 2005). The resulting model, the Waste Input Output Linear Programming model (WIO-LP), allows one to automatically obtain an optimal waste management and recycling strategy starting from a given set of alternative feasible strategies.

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[12] Wastes are considered in a broad sense, i.e. containing both by-products and waste in a narrow sense that refers to waste for treatment.

[13] The level and composition (calorific value, water and ash content and other aspects) of waste feedstock entering into the system are exogenously given. Integrated into the WIO model these variables become endogenous and can be determined by the interaction between goods production and waste treatment for a given level and composition of the final demand.

Furthermore, the model can explore the extent to which a given measure of eco-efficiency can be maximised by an appropriate combination of existing (technological and resources) potentials, and it provides a flexible framework to incorporate not only the average technologies given by the public IO tables but also any technology that is feasible in the sense of engineering.
The WIO model has been further developed in order to analyse households' sustainable consumption patterns (Munksgaard et al., 2005). Indeed, the WIO model is much more suitable for the analysis of sustainable consumption than the conventional input-output model because it can deal with the disposal stage of consumed goods as well as the purchase and use stages.

The WIO approach is quite new, but its relevance has been highlighted also in the work funded by DEFRA "Sustainable Consumption and Production – Development of an Evidence Base" (Wiedmann et al., 2006). At present, the lack of appropriate data represents one of the main shortcomings of the method: indeed, the WIO model and the applications analysed do not consider a large number of environmental loads, despite its number is not limited in principle, and the factors concerned with human toxicity and ecotoxicity as well. Together with data, future research lines should be related to further deepen the model, in the direction of including dynamics. Indeed, the analysis was static, and no aspect of the dynamic process, where goods are transformed into waste was considered. Proper consideration of these dynamic aspects is of great importance for analyzing issues of durable goods such as building, structures, automobiles and appliances.

5.3.5. Allocation

According to the ISO 14044, allocation means "partitioning the input or output flows of a process or other product system to the product system under study". The international standard ISO 14044 requires the following stepwise procedure for dealing with allocation problems:

- Step 1. Allocation should be avoided, wherever possible, either a) through division of the multifunction process into sub-processes, and collection of separate data for each sub-process, or b) through expansion of the system investigated to include the additional functions related to the co-products.
- Step 2. Where allocation cannot be avoided, the allocation should reflect the physical relationships between the environmental burdens and the functions, i.e., how the burdens are changed by quantitative changes in the products delivered by the system.
- Step 3. Where such physical causal relationships alone cannot be used as the basis for allocation, the allocation should reflect other relationships between the environmental burdens and the functions, for example in proportion to the economic value of the products.

This procedure has been criticized because it fails to account for the fact that different approaches to allocation might be adequate in different LCAs, depending on the goal and decision context of the study. The ISO procedure has also been criticized because it fails to account for the feasibility of the methods, the amount of work required, and the type of information resulting from the different methods (Ekvall & Finnveden, 2001). It also does not account for fairness or equity (Frischknecht, 2000).

The procedure has also been subjected to conflicting interpretation: researchers disagree on what approaches are allowed according to the ISO procedure and on what approaches are possible. In particular, the conflicting issues are related to:

- The identification of the most useful step in the ISO procedure
- The system expansion: whether it is always feasible and solves allocation problems
- Step 3 of the procedure: its relevance and what approaches are included.

The debate on how the procedure should be interpreted indicates that the current ISO text is too vague to give adequate guidance. At the same time, the criticism against the procedure indicates that it is not flexible enough.

In the debate on how to approach the allocation problem, in terms of what allocation approach is the most appropriate in different cases and how to identify the most appropriate approach to allocation, we distinguish between different types of allocation problems: multi-output processes, multi-input processes, and open-loop recycling.

A **multi-output process** is an activity that generates more than one product. A useful distinction is made between combined production, where the volume of products from the process can be independently varied, and joint production, where the ratio of products is fixed (Huppes, 1992; Weidema, 2000; Frischknecht, 2000).

Most authors would probably agree that allocation based on physical measure such as mass, energy, volume, etc. is the easiest approach to apply in most cases, but, so far, no single method stands out as a general solution to the allocation problem, because the choice of the method is strongly dependent on the specific application context. However, Weidema (2000) argues that system expansion (Step 1) at joint production and allocation based on physical relationships at combined production (Step 2) are the most adequate approaches for consequential LCA, because they results in the most accurate model of the consequences of decisions. Furthermore, system expansion is always possible and can solve the allocation problems. Other researchers argue that system expansion in practice is often based on inaccurate data (Ekvall & Finnveden, 2001) and typically adds new allocation problems to the system (Heijungs & Guinée, 2007). Data availability and the increased uncertainty associated to the larger and more complicated system, i.e. what processes should be included in the expanded system, are the subject of the ongoing debate.

Regarding subdivision (Step 1a), researchers disagree on whether this is an appropriate approach for consequential LCA. Ekvall & Finnveden (2001) argue that, in most cases, it does not result in accurate information on the consequences of decisions, since the sub-processes are typically not independent from each other (Ekvall & Finnveden, 2001).

Regarding Step 2, the application of linear programming model of the specific multi-output process has been recommended, and the comparison of data from different existing production plants with slight differences in product mixes. But it is evident that the latter approach requires a huge effort, since it is necessary to collect data from a large number of production plants in the sector of the multi-output process: nevertheless, the effort done will result in the availability of allocated data useful not only to model a specific plant but all production plants in the sector.

On Step 3, the ISO standard is not clear in describing what it should comprise. (Guinèe et al., 2004) recommend economic allocation as the baseline approach for most allocation problems in a detailed LCA, and this is also the approach chosen by Ecoinvent team. Frischknecht (2000) proposes a further Step 3 approach for joint production where allocation is based on the capacity to carry environmental burdens and still compete with alternative products, arguing that not only economic costs but also environmental burdens affects the competitiveness of products. When the sum of economic costs and environmental burdens of separate production is higher than the sum of costs and burdens for joint production, the costs and burdens are allocated between the joint products in a way that makes them all competitive.

A **multi-input process** is e.g. a waste-management process that simultaneously deals with more than one waste stream. A pure multi-input process is typically not a joint but a combined process, because the input of different waste streams can be independently varied. However, waste-management processes often also has outflows of products, such as electricity, heat, biogas, fertilisers, recycled materials, etc that are at least partly joint processes, because the maximum volume of output products depends on the quantity of waste treated.

The allocation problem here is to decide what share of the environmental burdens of the activity should be allocated to the waste stream of the life cycle investigated, i.e., included in the LCA of the product investigated. The international standard ISO 14044 presents no specific guide for allocation at multi-input processes. Essentially, the same procedure as for multi-output processes applies.

From the literature review, there is an evident general agreement that allocation based on physical relationships (Step 2) is the most appropriate for pure multi-input processes, when the input of different waste streams can be independently varied, and it is the most accurate model of the consequences of decisions that affects the waste flows. But this substantial agreement does not imply that the allocation problem can be considered solved; indeed the discussion is related to the feasibility of Step 1b, i.e. system expansion, and on how to address allocation when the waste-management process has outflows of products, such as electricity, heat, biogas, fertilizers, etc.

Seyler et al. (2005) allocate the emissions of the waste-management process based on physical relationships between the emissions and the waste flows (Step 2), and apply system expansion (Step 1b) to account for the consequences of the products from the waste-management process. Heijungs & Guinée (2007) denote this kind of process “open loop recycling“ and recommend to allocate the emissions of the waste-management process between the inputs and outputs using a Step 3 approach, despite acknowledging that the choice between different Step 3 approaches is essentially arbitrary. They argue against system expansion (Step 1b) because of the uncertainties it involves, because it might introduce new allocation problems, and because it requires more data; furthermore, the uncertainties can be large concerning what processes should be included in the expanded system.

Open-loop recycling occurs when the material from one product is recycled for use in other products. An allocation problem arises in the LCA because the recycling process provides one function for the product being recycled (waste management) and one function for the product containing recycled material (materials production). The problem is to decide what share of the environmental burdens of the recycling process, primary-materials production, and final waste management should be allocated to the life cycle investigated, i.e., included in the LCA of the product investigated.

Here, it can be noted that allocation of primary materials production and final waste management cannot be avoided through subdivision (Step 1A) in cases of recycling. On the other hand, ISO 14041 allows a few additional options for allocation at recycling. If the recycling does not cause a change in the inherent properties of the material, the allocation may be avoided through calculating the environmental burdens as if the material was recycled back into the same product. Otherwise, the allocation can be based on physical properties, economic value, or the number of subsequent uses of the recycled material.

The debate in LCA community is related to several issues:

- *What the allocation problem is.*

  Guinée et al. (2004) and Heijungs & Guinée (2007) propose that the allocation problem arises because the recycling process provides one function for the product being recycled (waste management) and one function for the product containing recycled material (materials production). The problem is then reduced to deciding what share of the environmental burdens of the recycling process should be allocated to the life cycle investigated, i.e., included in the LCA of the product investigated.

  Azapagic & Clift (1999) and Matsuno et al. (2007) propose that the allocation problem arises because a specific quantity of material is used in a cascade of products. The allocation problem then involves the environmental burdens not only of the recycling process but also of the original, primary production of the recycled material. The rationale is that primary production of the material is required to provide material also to the other products in the cascade.

Ekvall & Finnveden (2001) add that final waste management of the recycled material is also necessary for all products in the cascade. The allocation problem then involves the environmental burdens of the primary production, recycling, and final disposal of the recycled material.
Weidema (2000) has a similar perception of the problem, but recommends accounting for the marginal primary production, recycling, and waste disposal of the specific type of material, because these can all be affected by the recycling.

- *How the environmental burdens should be allocated.*
  - When the allocation problem only involves the environmental burdens of the recycling process, a cut-off approach is applied. The question is to decide if none, all, or parts of the recycling process belong to the life cycle of the investigated product:
    - Vogtländer et al. (2001) recommend that all the environmental burdens of the recycling process are allocated to the product in which the recycled material is used.
    - Guinée et al. (2004) propose to allocate the environmental burdens of the recycling process in proportion to the economic revenues from accepting the waste and selling the material. But the approach is criticized (Vogtländer et al., 2001) because it depends on the gate fees and price of recycled materials. These are unstable and highly influenced by governmental policies. For product with a long service life, they are unknown when the product is designed.
  - Azapagic & Clift (1999) suggest allocating the burdens by using a linear-programming model. The application of this model gives the same results as the cut-off method of (Vogtländer et al., 2001). All environmental burdens of primary production will be allocated to the first product in which the material is used, and the environmental burdens of the recycling process will be allocated to the product in which the recycled material is used.
  - An approach to allocate the environmental burdens equally among all the products in the cascade has been presented by, e.g., Matsuno et al. (2007), based on Markov chain model, using matrix-based numerical analysis in which it provides evidences on how to allocate environmental burdens of virgin materials by calculating the average number of times the elements of iron is used and its residence time in society.
  - Ekvall (2000) proposed also the so-called 50/50 allocation: it means that the environmental burdens of primary production and final waste management are equally divided between the first and last of the products in the cascade. The environmental burdens of recycling are equally divided between the product that delivers scrap to recycling and the product where the recycled material is used.
  - Methods to allocate the environmental burdens in proportion to the loss in quality or economic value of the material among the products in the cascade

have been presented by, e.g., Werner et al. (2007). With this approach most of the environmental burdens of primary production will typically be allocated to the last product in which the material is used.

- Weidema (2000) and Ekvall (2000) agree that allocation should be avoided through system expansion, and that the expanded system should include the processes that can be expected to actually be affected by the recycling. However, they disagree on what processes are expected to be affected by the recycling:
  - Weidema (2000) argues that recycled material from the investigated life cycle typically replaces virgin material. If this is true, the environmental burdens of primary production should be allocated to the last product in which the material is used.
  - Ekvall (2000) argues that recycled material from the investigated life cycle typically replaces a mix of virgin material and recycled material from other life cycles. If this is true, the environmental burdens of primary production should be divided between the first and the last product in which the material is used. The ratio depends on how sensitive the collection for recycling and the demand for such material are sensitive to changes in the price of the collected material. Thus, an important research task is to investigate, for the most important materials and markets, how sensitive the collection for recycling and the demand for such materials are sensitive to changes in the price of the collected materials.

The diversity of views and perspectives in the LCA community regarding allocation is evident: the procedure has also been subjected to conflicting interpretations, and this implies that researchers disagree on what approaches are allowed according to the ISO procedure, and on what approaches are possible.

The system expansion approach stands out like the most suitable approach to allocation. A common position still have to be agreed on it, because important drawbacks still have to be faced, like the increased data need and the more complicated system to be modelled. System expansion can introduce new allocation problems in LCA, although the new allocation problems are often less important than the original ones. Hence, it can be a fair approximation to neglect the new allocation problems or to solve them with a more simplistic approach.

The large diversity in views and the importance of the methodological problem imply that a significant effort is required to reach a general agreement on what allocation approach is the most appropriate in different cases, or on how to identify the most appropriate approach to allocation. The challenge in this area is to develop and to agree upon a text that is clear and flexible enough to give adequate guidance on allocation in LCAs with different purposes (Curran, 2007).

Allocation issues are extensively addressed in the Main Guidance document of the ILCD Handbook coordinated by the European Platform on LCA, currently under stakeholder consultation. Goal-dependent recommendations will be given, reflecting what the Platform and its active stakeholders perceive to be the state of the art. However, as clearly indicated above, LCA researchers are deeply divided, and further research is required to obtain a scientific consensus on the structure and possible solutions to the allocation problems.

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5.4. Impact Assessment

Because the ISO framework leaves room for a variety of approaches and interpretations, the LCIA is rather difficult for non-experts to apply and especially to interpret. Furthermore, more recent developments like the concept of Midpoint and Endpoint or Damage approaches, which have been included in the 2006 standards, need to be better explained and integrated. The state of the art of the research concerning the development of the impact assessment methods, including mandatory and optional elements in agreement with the ISO definition, is summarised here below.

Before going into the detail of the progresses in LCIA methodology, in the following paragraphs, the international ongoing activities of R&D concerning improvement of existing methods and elaboration of new methods and impact categories are summarised.

5.4.1. International activities and projects

The survey of international activities in the field of life cycle impact assessments sees the involvement of three major initiatives: the SETAC Europe working group, the UNEP SETAC Life Cycle Initiative and the European Commission's project "European Platform on LCA.

The SETAC Europe Working Group on Life Cycle Impact Assessment started its activities in 1994 with two groups active in this field: one in Europe, aiming to define a scientific basis for LCI, and one in North America, aiming to identify critical issues in this area. After the first phase, concluded in 1996, a second one was launched, active in the period 1998 to 2000, with the objective of making the first step towards the identification of best available practices in the field of LCIA. These activities, whose main outcome was the publication "Life Cycle Impact Assessment: Striving towards best practice" became the basis of a proposal for a structural cooperation between UNEP and SETAC, called Life Cycle Initiative, which deals with the identification of best available practice in LCA, including LCI, LCIA and life cycle management.

Under the UNEP SETAC Life Cycle Initiative, the Life Cycle Impact Assessment programme was launched aimed at increasing the quality and global reach of the life cycle indicators by promoting the exchange of views among experts whose work results in a set of widely accepted recommendations. The first phase of activities was concluded in 2006, during which important enhancements have been reached, like the "Final draft of the LCIA definition study", which proposed to structure both midpoint and damage approaches of LCIA in a consistent way. Now the second phase has been launched (2007-2010) with two main projects just started: one related to the *Improvement of Characterisation factors in life cycle impact assessment of ecotoxicity* and the other on *Indoor exposure assessment within LCA*.

Building on recommendations of the SETAC working group and the UNEP-SETAC Life Cycle Initiative, the European Commission's project "European Platform on LCA"[1], is working on the definition of one comprehensive framework for recommended methods and factors for LCIA, addressing environmental impacts at both midpoint and endpoint level in the impact pathway and covering different impact categories:

- Climate change
- Ozone depletion
- Acidification
- Eutrophication
- Ecotoxicity
- Human toxicity
- Respiratory inorganics
- Photochemical ozone formation
- Ionizing radiation
- Land use
- Resource Depletion.

Final recommendations will be published as part of the International Reference Life Cycle Data System (ILCD) Handbook and are expected to be provided by October 2008.

5.4.2. Mandatory elements of the impact assessment according to ISO

In agreement with the ISO definition, the mandatory elements of the LCIA phase shall include:

- selection of impact categories, category indicators and characterization models;
- assignment of LCI results to the selected impact categories (classification);
- calculation of category indicator results (characterization).

The main emerging issues from literature have been organised along three research lines: improvement of existing methods of characterisation; Development of mid-point and damage oriented methods in a common framework; Development of new characterization methods and new impact categories.

5.4.2.1. Improvement of existing methods of characterization

According to the results of the survey conducted on literature analysis and on-going research activities the following areas have been identified as the most relevant for improvement:

- Identification of spatially differentiated characterisation factors for some impact categories e.g. acidification, eutrophication, human toxicity etc.

[1] http://lca.jrc.ec.europa.eu/

- Improvement of existing approaches for some impact categories (through e.g. comparison of different models) in particular human and eco toxicity, resource depletion etc.
- Further development of existing models to assess the damage (endpoint category) for e.g. climate change or ozone depletion.

In LCA studies, emissions are not considered as continuous fluxes but as discrete pulses, since they are linked to single amounts of product function. To handle large numbers of such concentration pulses, it is advantageous to integrate them over both space and time because each pulse is thus characterised by a single value and pulses having different spatial and temporal characteristics can be compared. Temporal differentiation mainly deals with dynamic modelling; as regards LCIA main questions are which time horizon to take for time-integration of impacts and whether and how to align the different time horizons selected for different impact categories (Guinée et al., 2002).
The issue of temporal differentiation has been debated by some authors together with discounting, of which is considered a special case (Hellweg, 2003), but approaches are not suitable for LCIA yet. Indeed, it is necessary to take a decision about whether or not discounting should be applied in an LCA and if yes, in which cases and which methodological issues it should affect (Hellweg, 2003). The new approach to intergenerational discounting for computing net benefits from the use of environmental resources, proposed by Sumalia and Walters (2005), could give new insights but it has been developed outside the domain of LCA and has yet no relation with LCA. It would thus be necessary to apply their – and other discounting approaches – approach to LCIA, possibly by adapting GWPs and other characterisation factors to their discounting proposals. In relation to GWPs, Fearnside (2002) proposed a unified index that assigns explicit weights to the interest of different generations, but the new lists of GWPs is not available yet.

Spatial differentiation requires collecting location-specific data and calculating spatially specific characterisation factors (CF) (Guinée et al., 2002). Location-specific data are rarely available for all processes within a product life cycle, but at least for processes that appear to predominate in the overall impact of a product life cycle, additional effort to collect location-specific data is advisable. These data, when available, can be directly used or used as a basis for calculating the mean factor of a larger-region and the uncertainty values (e.g., standard deviation or range).

From the literature analysis, a growing interest towards introducing spatial differentiation in regional impact categories emerged. Different indicators and characterization models have been proposed to calculate the site-dependent CFs for a variable number of interventions and for the following impact categories: acidification, photo-oxidant formation, terrestrial eutrophication and toxicological impacts. For all of these categories the characterization factors have been calculated for Europe on a country basis (see Finnveden & Nilsson (2005) for bibliography).

For United States, region specific fate factors have been calculated for acidification, photo-oxidant formation and terrestrial eutrophication (Norris, 2003).

Region-specific effect factors have been developed only for the human health impacts of smog based on differences in population density, but not for acidification or eutrophication. The study of Hayashi et al. (2004), which was aimed at developing damage function of acidification for terrestrial ecosystems, provides acidification factors for emissions of SO2, NOx, NH3 and HCl into air for Japan as a whole, without any further spatial differentiation. An attempt is in progress to develop a global fate and exposure model called GLOBOX (Wegener, 2003), based on the combined approach of LCA and Risk assessment. This approach provides a separate assessment of above and below threshold pollution, offering the possibility to tackle above threshold impacts with priority. Spatial differentiation in fate, exposure, and effect modelling plays a central role in the implementation of the model. A methodological framework for the construction of characterization factors is provided, but a list of CFs for different countries has not been published yet.

Whether the country basis is the optimum level for site-dependent factors is an open question. It can be therefore of interest to develop methodologies for the definition of the optimal regions for site-dependent CFs (Nansai, 2005), or to propose CFs for different parts of a country (Finnveden & Nilsson, 2005; Nigge, 2001). The work of Nansai et al. (2005) develops a new concept for defining regions, called SAME (Spatial Area of iMpact Equivalency). A SAME is defined based on the main geographical characteristics that determine the magnitude of the environmental impact, rather than solely on the physical boundaries of a single physical region and can be mapped using a geographic information system. This approach, which till now has been only applied to the human health impact evaluation of benzene emissions, can be the basis for spatial differentiation of all impact categories. In (Finnveden & Nilsson, 2005) CFs for air emissions of NOx, SOx and particulates regarding ecosystem and human health impacts have been calculated for four different places in Sweden with two different stack heights. The results suggest that for ecosystems, site-dependent CFs for the considered atmospheric pollutants on a country level may be sufficient for most applications (the differences between different parts of Sweden are within a factor of two). However, for health impacts, where the differences are up to one order of magnitude, site-dependent factors on a country level may be inappropriate. Also the difference between low and high stack heights may be relevant, especially in densely populated areas. The method described in (Nigge, 2001) calculates the impact of primary airborne pollutants from transportation and energy production on human health taking into account the spatial differentiation due to emissions height and local population density distribution classified in five generic spatial classes defined for Germany. The strongest deviations from country average impacts apply to emissions at low heights in urban areas and decrease with an increasing atmospheric residence time.

As regards the results obtained by using site-dependent characterization factors, two case studies have been analysed, which concern the application of the EDIP2003 methodology.

The first paper (Bellekom et al., 2006) has checked the practicability of site dependent methods by quantifying the time needed to collect the required additional inventory data and by looking into the added value of site-dependent LCIA results. The acidifying impact for three existing LCA studies was re-calculated with the site-generic and site-dependent acidification factors provided by the EDIP2003 for the European emissions. The introduction of spatial aspects in the impact assessment resulted to be important for the interpretation: the overall results are unchanged, but relative contribution of basic processes changes with the use of site-generic factors. This conclusion needs to be validated by comparing and checking the methodology adopted with other spatial differentiation models. Currently, any generalization of this conclusion should be avoided.

In the second paper (Hauschild et al., 2006) regionalised factors for NMVOC, CO and NOx for 41 European states and Europe generic were applied to a case study. This method is close to endpoint modelling because it comprises a large part of the cause-effect chain including exposure assessment and exceeding of threshold values. Moreover, the authors found that the variation in site-dependent characterization factors is larger than the variation in POCP factors**.** If it were confirmed for other methods too, it would be more important to represent the spatially determined variation than the difference among the substances, making it unnecessary to derive CFs for individual VOCs.

In conclusion it emerged that one of the main research area is the development of spatial differentiation over the different impact categories in such a consistent way that clear and unambiguous guidelines can be developed about how and which geographic data should be collected in inventory analysis. One impact assessment method may need more detailed geographic data (at the level of locations or in terms of the value of a certain parameter, e.g. pH value) than another (e.g., country or continent) may. The data needed for each different characterisation method need to relate in such a way that the practitioner is requested to collect data at a certain geographic level and that all other geographic data needed for other characterisation methods can be derived from this entry. Maybe GIS could offer the data hierarchy framework, but the key issue is the availability of regional emissions and environmental data in order to calculate regional characterization factors and to combine these with the appropriate regional emission data. The SAME approach (may be also GLOBOX, but it needs further elaboration before evaluating) offers the greatest potential with respect to this challenge.

At the same time, the manageability and quality-control of the resulting inventories with tens or hundreds of thousands of location or country-specific elementary flows for LCI results has to be taken into account. Different approaches may be needed to keep the balance between precision and manageability/practicality.

In general the analysed literature concerning spatial differentiated CFs has highlighted some opens questions about the comparability of the different models proposed and/or the possibility of extending the use of a model to other countries/regions.

Seppälä et al. (2006), for example, propose impact category indicators for acidification and terrestrial eutrophication, but do not compare and discuss them with other indicators widely used; the Gaussian plume model-based approach of Nigge (2001) and the application of the Ecosense model by Finnveden & Nilsson (2005) have not been compared neither this last approach has been validated for other countries; the comparison of the different characterization models proposed for the photo-oxidant formation impact category by Labouze et al. (2004) and by Hauschild et al. (2006) has not been discussed.

What has been observed above about the comparability and the need for developing more suitable indicators can also be said more in general for all models that have been proposed in literature. In addition to this general aspect, further needs emerged from the review for the following impact categories:

- toxicity, including human toxicity and ecotoxicity,
- resources, abiotic and biotic.

Both impact categories were addressed in specific task forces of the Life Cycle Impact Assessment (LCIA) Programme, one of the programmes of the UNEP/SETAC Life Cycle Initiative (http://jp1.estis.net/sites/lcinit/). In that context a comprehensive comparison of the LCIA toxicity characterisation models developed was carried out, which led to the development of USEtox, a scientific consensus model both for human and ecotoxicity that contains only the most influential model elements (Rosenbaum et al., 2008). Based on a well-referenced database, the consensus model has been used to calculate CFs for more than thousand chemicals and will also be the basis for improving CFs for metals (Gandhi et al., 2007). The USEtox has the important advantage of being supported by most model developers in the LCIA field. The model, which exists as a research model in Excel, is now under peer review and will be further developed during the second phase of the UNEP/SETAC Life Cycle Initiative (Rosenbaum et al., 2008).

Moreover, as regards the ecotoxicity effects assessment, improvement and further development of existing methods (mainly PNEC-Predicted No Effect Concentration and PAF-Potentially Affected Fraction of species) are recommended (Larsen & Hauschild, 2007). In particular the PNEC based methods need to be less risk-assessment oriented and more suitable for LCIA, while the PAF related methods need to improve the chemical coverage and to include mixtures and damage modelling.

Abiotic resource depletion is one of the most frequently discussed impact categories and there is a wide variety of methods available for characterising contributions to this category. A recent work of Steen (2006) reviews existing LCIA methods in relation to depletion problem definitions.

He observes that resource depletion is not a well-defined concept due to the large subjectivity (the concept mainly depends on the user's definition of depletion and on the expectations about the future).
Moreover, the distinction between environmental aspects and economic aspects related to the exploitation of a resource is not always clear and different ideas of time perspective to apply exist. As regards this last point some authors define the abiotic depletion in terms of changes in the environmental impact of extraction processes at some point in the future (e.g. as a result of having to extract lower-grade ores or recover materials from scrap). This seems to be not consistent with the methods adopted for the other impact categories, where the concern is not with the impact of future changes in processes and interventions, but with those of current interventions. Moreover, future changes in processes and interventions constitute changes in the product system and should be accounted for in the Goal and scope and Inventory phases, not during Impact assessment for a particular impact category (Guinée et al., 2002). However, on these aspects, discussion is still needed. A consensus currently seems to be achieved about moving the discussion focus from resource extraction to the concept that exploited resources come back to the environment in a degraded form, which is no more able to deliver its original functionality. In other words, "it is not the extraction of materials which is of concern, but rather the dissipative use and disposal of materials" (Steward & Weidema, 2005). In Steward & Weidema (2005), Weidema et al. (2005) a method is proposed to quantify the effects of resources use, both biotic and abiotic. It is "based on a generic concept of the quality state of resource inputs and outputs to and from a production system" and uses two key variables for the modelling of impacts: the ultimate quality limit, which is related to the functionality of the material, and backup technology. This offers, however, merely a framework that needs further elaboration. In particular, it will be necessary to determine, for each resource, values for functionality/quality indicators, ultimate quality limits, backup technologies.

Finally, the analysis of some recent references concerning the improvement of characterization factors for ozone depletion (Kentaro et al., 2006) and climate change (Brakkee, ongoing) shows that the research is currently focused on the development of damage models. To obtain a complete framework of the damage function, expertise from other scientific disciplines is needed. Main problems are how to use the complex models, which were developed in different contexts, for the improvement of the LCIA and to evaluate the uncertainty of damage function. Interesting results are expected from the Japanese national LCA project, which developed LIME, a comprehensive methodology for LCIA also including endpoint approaches (see also 5.4.1.2).

The European Platform on LCA project has conducted a quite comprehensive review of existing methods for LCIA and comes up with recommendations. Besides, important research needs has been highlighted for the impact categories considered (Hauschild et al., 2008b).

5.4.2.2. Development of mid-point and damage oriented methods in a common framework

According to ISO 14044, the first step of impact assessment is the classification of LCI results into impact categories. The indicator of an impact category can be chosen anywhere along the impact pathway linking inventory data to the damage to human health, the natural environment and natural resources. Characterisation at midpoint level models the impact at an intermediate position along the impact pathways, while characterization at the endpoint level models the impacts to the damage. A relevant aspect of the research activities concerning the impact assessment is the development of damage-oriented methods, which aim to an easier interpretation of the LCA results. Even if there are differences between midpoint and endpoint methods, the current trend aims to harmonise the two approaches, according to the recommendations of the SETAC working group and the UNEP/SETAC Life Cycle Initiative (Jolliet et al., 2004). In fact, the design of a consistent method with both midpoints and endpoints has the advantage of offering the LCA practitioner more flexibility in the choice of the impact assessment method, depending on the goal of the study and the intended application. Methods of this type are already available (IMPACT 2002+ and LIME) or their development is in progress (Recipe project (Heijungs et al., 2003; Goedkoop et al., in prep.), LIME2 (Itsubo & Inaba, 2007)). The framework proposed by the task force of the UNEP/SETAC Life Cycle Initiative is expected to provide a basis for the analysis and the comparison of the existing methods and to facilitate the inclusion of new impact categories, also those particularly suitable for developing countries. The goal of defining quantitative impact pathways up to the damage categories cannot be achieved yet for all types of impacts because of the limited scientific knowledge of certain aspects. In fact the choice of stopping the models at midpoints was often due to uncertainty or lack of agreement of endpoints models. In this case, the proposal of the Life Cycle Initiative is to give at least a qualitative description of the expected influence of the midpoints indicators on their respective damages in order to facilitate the LCA interpretation phase.

Also the European Platform on LCA is working on the definition of one comprehensive framework for recommended methods and factors for LCIA, addressing environmental impacts at both midpoint and endpoint level in the impact pathway and covering different impact categories. Therefore, since the integration of midpoints and endpoints is a recent development, additional research is expected to be carried out in the next future in order to provide consistent and operational sets of methods and factors for LCIA. In this context, the development of adequate damage indicators is a desirable result of the research activities. Moreover uncertainty of damage function modelling is another important point to be elucidated.

It is worth to highlight that some of the endpoint/damage methods include a weighting scheme, which may be a deviation from the IISO standards when a study concerns a comparative assertion intended to be disclosed to the public.

5.4.2.3. Development of new characterization methods and new impact categories

Starting from the state-of-the-art described in Jolliet et al. (2004), the most recent developments concerning new characterization methods or approaches and new impact categories are here presented.

The first interesting topic of discussion focuses on approaches to risk assessment (RA) in conjunction with LCA. LCA is mostly based on the general prevention principle, whereas RA is based on the risk minimization principle. Discussions on the delimitation between the two methods have gone on since years. Methodological frameworks have been proposed following two different approaches: 1) combining LCA and RA, i.e. fitting some key characteristics from RA (threshold and sensitive areas) into LCA methodology itself (Nishioka et al., 2002a; 2002b; 2006; Wegener Sleeswijk, 2003); 2) linking the strategies of life cycle and risk analysis within the same toolbox (Sweet & Strohm, 2006). Unfortunately the methods proposed are not ready to be used yet. The advantages given by the combination of RA and LCA need to be clarified and further elaborations are required to identify which specific methods (or only elements of methods) are useful to combine and for which decision-situations.

Most LCIA methodologies do not have an impact category ‘noise’. This seems in contrast with the observed fact that most people consider noise to be a major environmental problem, but it is probably due to the unavailability of an appropriate and practically feasible impact assessment method for noise. Müller-Wenk (2004) has provided a method enabling practitioners to take into account noise impacts. A further work of Meijer et al. (2006) has developed different traffic scenarios with different car/truck speeds etc. and has added the indoor compartment to the work of Müller-Wenk (2004). Unfortunately the results can only be applied to the sector ‘road transport’. An adaptation to rail noise is planned by Müller-Wenk (2004), but elaboration to other sources of noise should be also developed. There are also some open questions on this subject, such as the aggregation in a life cycle perspective of the traffic noise to noise from other phases of the life cycle (occurring also in different regions/countries) and the uncertainty related to the calculations (variability of traffic scenarios and parameters used for the calculation). Despite the recent progresses in developing methods to take into account noise impacts and understanding the cause-effect mechanisms,, no generally applicable models have been developed yet.

The category ‘land use’ is a relatively new topic in LCIA and is still under development, despite the numerous existing proposals. A distinction is often made between use of land with impacts on the resource aspect and use of land with impacts on biodiversity, life support functions, etc. In this review the resource aspect is captured separately under the heading of “resource depletion”. On the intervention side, a distinction is often made between land occupation (i.e. occupancy and use) and land transformation (i.e. changing its quality).

The UNEP-SETAC Life Cycle Initiative Working Group on LCIA Task Force 2 (TF2) on Resources and Land Use has proposed a framework for the Life Cycle Impact Assessment (LCIA) of land use, but no practical methods have been recommended yet (Milà i Canals et al., 2007a; Milà i Canals et al., 2007b).
A common agreement on this proposal has not been reached up to now and the debate on this issue needs to continue, because LCA results will be incomplete and less credible as long as land use impacts are not being incorporated (Udo de Haes, 2006). The open questions for this impact category are still very basic: what are the main impact categories under the heading "land-use impact"? How can these impacts be best assessed in LCIA? Which indicators, which interventions, which characterisation models and factors? What should be done with impacts/aspects of land use that do not fit in the general LCIA structure?
Key differences between a variety of methods which are currently being practised have already been identified by European Platform document (Hauschild et al., 2008a). Now it would be necessary to build consensus on what needs to be assessed (impact categories) and then to elaborate indicators, characterisation methods and factors. It is particularly important to develop and elaborate methods with readily-available lists of intervention-characterisation factor combinations, so that it is clear to the practitioner which intervention data should be collected with respect to land use and how these can be practically and easily handled in the impact assessment phase. It could also be suggested to learn from the LCIA experiences with the toxicity categories, i. e. a step-by step process of definition of a simple model and further improvements towards more sophisticated but operative approaches (Guinée et al., 2006).

The subject "exergy" can be thought as "new characterisation methods – resources" when exergy is proposed as indicator for depletion of resources, and as "new impact category" when exergy is proposed as overall method for the impact assessment of resources and emissions. Anyway several proposals have been published on this subject. Gong & Wall (2001), Cornelissen & Hirs (2002) and Bösch et al. (2007) advocate applying exergy for assessing the depletion of natural resources. Physical resources are classified into natural exergy flows, exergy funds and exergy deposits. Natural exergy flows and sustainable use of exergy funds establish the renewable resources. Unsustainable use of exergy funds, e.g., careless clearing of forests, and exergy deposits make up for the non-renewable resources. The total exergy use over the life cycle is considered. Bösch et al. (2007) explain that exergy "can be utilised as an indicator of resource quality demand when considering the specific resources that contain the exergy. Such an exergy measure indicates the required resources and assesses the total exergy removal from nature in order to provide a product, process or service." Daniel & Rosen (2002) and Bakshi (2000) include emissions as well as resources. Daniel & Rosen (2002) discuss a case study examining emissions produced during 13 fuel life cycles for automobiles, on mass and exergy bases. Bakshi (2000) advocates applying the concept of emergy, being "the embodied energy or energy memory in any product or service".

However what is the precise added value of emergy over exergy and if exergy/emergy can also be an environmentally relevant indicator for emission related impacts remain open questions yet. Exergy has been applied to a number of different areas with different methods. The results from these methods are not immediately comparable, so general methodological guidelines should be developed in order to increase comparability. All approaches require more data than conventional LCA and many of these data still need to be collected and/or calculated.

Most LCIA approaches today neglect effects due to ionising radiation. However, a comprehensive tool such as LCA should not neglect potentially relevant effects to human health. Data for the assessment of the human health damages related to the releases of radioactive material to the environment has been published in (Frischknecht et al., 2000) and (Meijer et al., 2005). In (Frischknecht et al., 2000) the fate and exposure analyses are based on site-specific modelling of the French nuclear fuel cycle, from which generic exposure factors are derived. The effect analysis is based largely on epidemiological studies. The impact pathway and damage factors assigned correspond to the typical situation for Western European nuclear power supply. The assessment does not include human health damages due to ionising radiation released by severe accidents, nor by long-term underground waste storage facilities. In (Meijer et al., 2005) CFs for radon and 3 gamma-radiating elements that are released from building materials inside dwellings are given for the Dutch situation. Main R&D needs for this impact category are to expand the list of CFs to include more than the current 31 radionuclides (Frischknecht et al., 2000) and 3 gamma radiating elements and radon (Meijer et al., 2005) and to get the relevant emission data as far as these are not usually provided by databases.

As far as the inclusion of water use in LCIA is concerned, the indicator often used is the total input of water used (kg or m3), which, however, is not adequate to assess water resources from a sustainability perspective. Suggestions have been found in recent literature (Owens, 2002; Heuvelmans et al., 2005) about how to improve and elaborate the impact assessment of water use (water as a resource), but practical value of the methods proposed is currently very limited and significant efforts should be put into operationalisation of the characterisation models and related characterisation factors. Another project (Pfister, 2007) proposes a methodology to develop the assessment of water consumption in the context of LCA using GIS-based data evaluation, correlation analyses and case studies in cooperation with industrial partners. Being started in 2007, it has not provided results yet.
There is a specific aspect concerning the water quality which is of strategic concern in countries as South Africa and Australia and would require incorporation in the life cycle assessment studies: the salinisation of water resources and of agricultural plots. There are sufficiently clear cause-effect relationships between the sources and impacts of salinity, and impacts are claimed to be sufficiently different in nature from existing categories to warrant a separate salinity impact category.

Two specific methods have been included in this review which concern soil salinisation in Australia (Feitz & Lundie, 2002), only applicable for irrigation practices, and water and soil salinisation in South Africa (Leske & Buckley, 2004 Part I – Leske & Buckley Part III). The approach described in (Feitz & Lundie, 2002) does not use a readily available list of CFs, but for each relevant life cycle step a site specific CF has to be calculated and data on irrigation volume, pH, electrical conductivity (EC), alkalinity and the concentrations of Na, Ca, and Mg of the irrigation water have to be collected. In order to increase its applicability, it will be necessary to develop more easily accessible data sets needed for the calculations or even better, to develop a list of CFs for a set of environment types. Main limits of the approach described in (Leske & Buckley, 2004 Part I – Leske & Buckley Part III) are the applicability only to South African conditions and the needs of some specific knowledge for the collection of the correct inventory data. A further elaboration should include the development for other regions than South Africa and associated inventory data collection guidelines.

Indoor and occupational exposure, which includes also injuries (casualties) related to working environment accidents, is another aspect that should be addressed in the LCA studies because of its not negligible relevance in comparison to the total human toxicity effects. The literature review highlights that data is still a problem on both impact assessment and inventory levels. Emissions to the workplace are often unknown. Thus, in order to consider workplace emissions within LCA, emission factors need to be made available. Furthermore, exposure determinants vary among workplaces. Therefore, it is unlikely that a single standard model for indoor exposure with fixed parameter values can be used. Except for the method of Meijer et al. (Meijer et al., 2005; Meijer et al., 2006) – who have determined emission rates for 38 volatile organic compounds, radon and gamma radiation emitted by 17 building material categories – other practical lists of emissions and associated characterisation factors are lacking, hindering full application of these methods/models. Actually, the trend foreseen for the inclusion of these aspects in LCIA is that indoor and occupational exposure will not be a separate impact category anymore but will become a compartment in the human toxicity impact category, such as in the current improvement of the USE-tox model for life cycle impact assessment of toxic releases.

At the end of this part of the report it is also worthwhile to bring some examples of impact categories developed for specific production sectors. Here we mention the applications involving GMOs and seafood production systems. For most of them the impact categories have been identified but elaborated methods fitting in the general structure of LCIA are still missing and should be developed. The work of Jank et al. (1999) tried to integrate the concept of risk assessment of micro organisms used in biotechnology into the impact assessment of LCA, but significant gaps exist yet (unambiguous definition of the interventions, the indicator, the characterisation models, its data needs and related characterisation factors).

Since all of these need further elaboration, the practical use of this approach is very limited and requires significant efforts from the practitioner.
A review of published LCA research in fisheries and aquaculture (Pelletier et al., 2007) indicates that traditional environmental impact categories are often used, but modest efforts have been produced to develop a range of non-traditional life cycle impact categories specific for seafood LCAs. Notable examples include the modelling of benthic impacts, by-catch, emissions from anti-fouling paints, and appropriation of Net Primary Productivity, which is the net flux of carbon from the atmosphere into green plants per unit time, as a proxy for biotic resource use impacts. Even if several authors have discussed the desirability of new impact categories or have described possible additional categories, much more work seems to be necessary in this production sector to develop methods and CFs.
Besides, the European Platform on LCA project has also identified research needs for some (relatively) new impact categories (Hauschild et al., 2008b).

5.4.3. Optional elements of the impact assessment according to ISO

In agreement with the ISO definition optional elements of LCIA may include: normalization, grouping, weighting and data quality analysis of LCIA profile. Normalization, grouping and weighting methods are used depending on the goal and scope of the LCA and methods and calculations must be documented to provide transparency. In the following two paragraphs normalization and weighting will be treated. Data quality analyses will be included in Interpretation, according to the review scheme proposed in this document.

5.4.3.1. Normalization

ISO 14044 defines normalization as ‘the calculation of the magnitude of the category indicator results relative to some reference information’. The aim is ‘to understand better the relative magnitude for each indicator result of the product system under study’.
Other definitions of ‘normalisation’ exist, as is the case in multicriteria analysis where it is often understood as the ratio of the various values in a data set by a single reference value from that set. In contrast to the ISO normalisation, this can be called “internal normalisation”. Internal normalisation is not usually applied in LCA and it is also not in line with the ISO definition of normalisation given above, which could be called ‘external normalization’. Moreover it does not allow assessing the relative significance of the impact categories.
In this framework, Norris (2001) discussed a problem concerning the need for congruence between normalization and valuation. He gives examples, which he says very common in North American LCAs, of combined use of internal normalization, whose results reflect the relative performance of the alternatives with respect to each other, plus valuation with case-independent weights, which reflect the importance of each impact category in general.

In this case, congruence will always be absent, but congruence is not guaranteed with external normalization. In fact, to obtain congruence it is necessary that spatial and temporal scope of the emissions inventory used to calculate the normalization factors and the emissions inventory of the case study to be assessed are coherent.

Another problem is highlighted in Heijungs et al. (2007): incompleteness due to a lack of emission data and/or characterisation factors in product and/or reference systems leads to biased normalization. This may affect the three types of usage of normalized results: error checking, weighting and standalone presentation. The application of contribution analyses is proposed in order to alert LCA practitioners of the possibility of biased results. The following hypotheses can be put forward, but more evidence on their correctness would be necessary:

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

Congruence and bias issues may play an important role in normalization and should therefore receive further attention resulting into clear guidelines how to deal with these issues.

Consistency should be ensured for methodological and data choices made in drafting normalization and valuation/weighting data and in performing LCA case-studies. It would be useful to draft a list of these issues as checklist for practitioners and normalization data/method developers.

Recently different works have been published concerning normalization factors on different spatial levels: global and European (Wegener et al., 2008), US (Bare et al., 2006) and Australian (Lundie et al., 2007). The possibility of merging them into one encompassing global normalization data set with regional differentiations should be explored.

5.4.3.2. Weighting

Weighting consists in assigning numerical factors to each impact category according to their relative importance, multiplying these factors by the indicators and possibly aggregating the results in one indicator. Before weighting, the various indicator results must first be converted into the same units, one possible method for which is normalisation. Weighting is based on value-choices (e.g. monetary values, standards, experts' panel). According to ISO, weighting shall not be used for comparative assertions disclosed to the public.

As observed in Bengtsson & Steen (2000), the contribution of weighting to the relevance and acceptability of LCA results is a matter of discussion. They propose a different viewpoint defining the weighting as a check of the compatibility between the environmental impact associated with a certain technical system and different sets of societal preferences. The challenge is to choose a set relevant in the context of the particular study or decision situation, i.e. in agreement with the purpose of the study and the intended audience of the results. In addition, Schmidt & Sullivan (2002) observed that no universal weighting set for the world is likely to be derived, especially for global organizations and companies. Hence, the authors recommend that LCIA quantitative weighting, especially those provided in pre-packaged software instruments, should not be employed. So the first point under discussion is the following: is there a need for a universal weighting set for the world, and if so, are there other ways to obtain such a set? Itsubo et al. (2004) propose to apply the conjoint analysis, which has been widely used in market research, to the step of weighting, as they have done in LIME, the Japanese Life Cycle Impact Assessment Method based on Endpoint modelling. Two types of weighting factors, an economic valuation and a dimensionless index were obtained, but how the conjoint analysis results compare to the other economic valuation methods (a review of them can be found in Eshet et al. (2006)) remains an open question. Also the Ecotax method proposed by Finnveden et al. (2006) raises questions about its suitability. The approach for monetization of environmental impacts is based on the consistent use of ecotaxes and fees in Sweden as a basis for the economic values. "An underlying assumption for this, is that the decisions taken by policy-makers are reflecting societal values thus reflecting a positive view of representative democracy" (Finnveden et al., 2006). But whether monetary measures which are decided for one purpose (taxation) are valid for use as a measure of the environmental importance is not so evident. Moreover, the assumption that taxes correspond to the external costs or at least reasonable approximations of them cannot be proved because the real external costs are not known. Another problem raised by the Ecotax method is that taxes usually change over time and differ per countries.

Generally monetization is used for several purposes: as a form of weighting different environmental impacts, to incorporate environmental costs in an economic cost-benefit analysis, for determining environmental (Pigouvian) taxes, etc. In particular, monetization is at the basis of specific decision situations where environmental impacts need to be compared with other costs or benefits that by their nature are expressed in terms of money. Two main approaches are the use of damage costs and the use of prevention costs. With damage costs, impacts such as climate change, acidification and toxicity are converted into an estimate of the economic damage incurred, e.g. related to damage to property (buildings, crops), costs related to curing diseases, loss in income due to illness, or even a measure of the loss due to a decrease of the quality of life. With prevention costs, the basis is the costs that would have to be made to prevent the chemical from being emitted, or to prevent the damage from occurring.

In this framework particular attention has been given to the problem of the transferability between health impacts measured in disability adjusted life years (DALYs) and monetized health impacts. In general, monetary values are not available for all relevant health impacts and values from different studies can often not be combined and/or aggregated. As observed in Hofstetter & Müller-Wenk (2005), there is no general conversion factor between DALYs and monetary values. Moreover, adding up results from different monetization methods (e.g. damage costs and prevention costs) is not valid as the common unit of measure of money is not a justification in itself. To support the use of monetization for assessing health damages, data availability on monetary values of health impacts and on DALYs should be increased, especially for those health impacts that are likely to dominate most LCA case studies. Moreover, Hofstetter & Müller-Wenk (2005) strongly endorse studies about mild illnesses, which tend to be very relevant within LCA but are poorly studied in the DALYs systems as well as in health economics.

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5.5. Interpretation

The review showed that no major new insights or progresses in developments exist about the Interpretation phase. It seems to be a "free zone", in which the lack of clear procedural guidance in ISO framework, together with the inherent features of the Interpretation itself, has legitimated a scarce development. Indeed, as choices based on values become important when conclusions and recommendations have to be drawn up, this would require transparency and guidance on several possible approaches, both in terms of procedures and competencies.

For the purpose of the present review, even if uncertainty affects all the phases of LCA procedure, we have considered its analysis as the main topic of interpretation. Indeed, ISO 14044 does not explicitly include it in interpretation: "*[...] interpretation shall include an assessment and a sensitivity check of the significant inputs, outputs and methodological choices in order to understand the uncertainty of the results.*".

5.5.1. Uncertainty

ISO 14044 defines the uncertainty analysis as "*a systematic procedure to quantify the uncertainty introduced in the results of a life cycle inventory analysis due to the cumulative effects of model imprecision, input uncertainty and data variability*". To be precise, uncertainty is different from variability, which can be attributable to the natural heterogeneity of numerical values and cannot be reduced with further measurement (Bjorklund, 2002); however, with the term uncertainty many authors referred to both and in the present review we do the same.

Uncertainty analysis is recommended by ISO standard but guidance is not given for a systematic approach and best practices are missing. There are many ways of classifying uncertainty, but a proper framework to distinguish types of uncertainty in LCA has not been agreed yet. Among those available, we considered the following classification[14]:

- **Parameter uncertainty**, which depends on the reliability of data and it is due to inaccuracies of approximations, instruments used, operator, etc.;
- **Model uncertainty**, which is due to limitations in the modelling process (e.g. ignoring non-linear processes in inventory and impact assessment, no spatial and temporal details on emissions, no interaction with other pollutants, etc.);
- **Scenario uncertainty**, which is due to normative choices (e.g. allocation procedure, functional unit, etc.).

Several techniques for the quantification of the uncertainties are available, and all of them seem to be necessary in order to cover the full range of needs.

In the following paragraphs, the state-of-the-art related to the three types of uncertainty as resulted from the review is described.

[14] We could add the "Computational uncertainty" as proposed by R. Heijungs.

5.5.1.1. Parameter Uncertainty

Parameter uncertainty comes out from our incomplete knowledge about the true value of a parameter and it is generally due to measurement errors in input data.
Several applications of parameter uncertainty exist in literature and different techniques are employed, such as: Monte Carlo Analysis, Bayesian statistics, analytical uncertainty propagation methods, calculation with intervals and fuzzy logic. Recently, also an analytical approach based on Taylor series expansion for lognormal distribution has been presented, as a tool to support the analysis of Uncertainty Propagation in Life Cycle Inventory and Impact Assessment[15].

Most of the authors analysed in this survey make use of statistical methods, like Monte Carlo analysis. Lo et al. (2005) used the combination of Monte Carlo technique with the Bayesian method, making it available a framework to combine judgement information and observational data. Huijbregts (1998, Part I) used the Crystal Ball® software (DECISIONEERING, 1996) for Monte Carlo Analysis, in order to calculate the uncertainty importance of each parameter. It was used to spot the correlation between each parameter and the model chosen, in this way it has been possible to identify parameters whose uncertainty has the biggest influence on model outcomes. Sonnemann et al. (2003) and LaPuma et al. (2002) used a similar method.
Similar methods, like the Latin Hypercube, are less used but it seems, from the available literature, that good results can be obtained as well.

The fuzzy logic[16] has been used by Tan et al. (2007) and Cellura et al., (2004): the latter developed also the F.A.L.C.A.D.E. (Fuzzy Approach to Life Cycle Analysis and Decision Environment) software that helps practitioners to use the fuzzy set theory in an LCA study. Indeed, even thought the fuzzy logic is a very useful method that allows reducing costs and processing time, few examples of application in LCA studies exist and thus no general conclusions can be drawn.

Furthermore, it is not possible to compare the processes analyzed since the authors did not explain the reasons behind the choice of each technique and did not make comparison among them. At a first sight, it seems that the choice of a specific technique is not determined by an evaluation of its pros and cons but simply by the knowledge of the method and, in the case of Monte Carlo, by its availability in the majority of LCA software.
In the next future it would be useful to provide guidance on which specific technique for parameter uncertainty analysis is better to use in which context; furthermore, methods to assess the appropriateness of distributions are necessary, together with a more detailed investigation of the understanding of parameters' interdependencies (Lloyd & Ries, 2007).

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[15] The approach has been presented by Jinglan H. et al. during the 18th SETAC Annual Meeting in Warsaw.
[16] Fuzzy logic allows the practitioner defining the uncertainty using both numerical values and logical expressions, and works assessing a true or false judgement.

5.5.1.2. Model uncertainty

Model uncertainty has been defined as the uncertainty related to the modelling and it is caused by simplifications of aspects that cannot be modelled within the present LCA structure, such as (Huijbregts, 1998 Part I):

- temporal and spatial characteristics lost by aggregation;
- linear instead non-linear models: in Impact Assessment it is assumed that ecological processes respond in a linear manner to environmental interventions and thresholds of interventions are disregarded;
- calculation of characterisation factors. They are computed with the help of simplified environmental models which also suffer from model uncertainties;
- lack of characterisation factors for toxicological important substances or important sum emissions, such as metals.

The papers selected for the review focus more on the model uncertainty in multi-media fate and exposure models for toxicity potentials, since they represent an important source of uncertainty.

It is unclear from ISO standards how the uncertainty (and variability) in toxicity potentials should be assessed and how these uncertainty estimates should be implemented in LCA case studies.

In the approaches analyzed in this review, model uncertainty is often evaluated in combination with parameter uncertainty: indeed, the correlation between these two types of uncertainty is very high. According to Huijbregts (1998, Part I), when a model suffers from large model uncertainties, the results of a parameter uncertainty analysis may be misleading: for this reason is important to operationalise parameter uncertainty in the model. Indeed the approaches analysed deal with both parameter and model uncertainties, but they mostly focus on identifying the source of model uncertainty rather than in developing a defined framework for the analysis.

In Hertwich et al. (2000) an uncertainty analysis framework for multimedia risk assessment is proposed, in which parameter uncertainty/variability as well as model uncertainty and decision rule uncertainty are addressed. The authors put great efforts in identifying the different and several sources of model uncertainty, focussing on two model components that have been found to be important among the range of model assumptions in CalTOX: the steady state assumption for the pollutant transfer by rain from air to soil, and the modelling of the pollutant concentration in plants. But this analysis has been only exploratory since these uncertainties are difficult to analyse quantitatively. Efforts towards quantification have been made by (Huijbregts et al., 2001 and 2000) who quantified the toxicity potentials variance resulting from choices in the modelling procedures in USES-LCA by means of scenario analysis. The choices evaluated in the modelling procedure are the following: i) time horizon; ii) the decision whether or not to include potential impacts exported from the continental scale to the global scale. It has been demonstrated that the value choice of the time and spatial horizon in the impact assessment of toxic substances is important.

Other quantitative attempts to evaluate the influence of different model choices has been done by Huijbregts et al. (2000) by comparing the outcome of the models USES-LCA and USES 1.0. The comparison revealed that the dominant source of uncertainty depends on the nature of the substance under study and on the initial emission compartment chosen.

An interesting approach for estimating uncertainties for toxicological impact characterisation, implemented in the LCIA method IMPACT 2002, has been presented by Rosenbaum et al.(2004): it is very transparent, quick to use and it can be easily applied to combine the uncertainty of the emissions inventory with those of the impact assessment phase in a LCA study. The uncertainty is estimated for intermediate results from the chemical fate, human intake fraction and two toxicological effect modules. Then, the overall uncertainty estimates are arithmetically calculated.

Several authors highlight the importance of a quantified analysis of model uncertainty, because it is recognized that these uncertainties can alter the results by several order of magnitude. However, very few studies have an analysis on model uncertainty and discuss strategies for identifying model uncertainty that are important contributors to the overall uncertainty (Lloyd & Ries, 2007). Efforts are still necessary, but mainly at procedural level: there is a need of a framework helpful in organizing the analysis and identifying significant sources of uncertainties (Hertwich et al., 2000). Furthermore, co-operation with specialists of other scientific disciplines will facilitate the implementation of these improvement options.

5.5.1.3. Scenario uncertainty

"Scenario uncertainty", or "Uncertainty due to choice" or "Decision rule uncertainty", reflects that LCA outcomes inherently depend on normative choices (Huijbregts, 1998). It can affect every phase of LCA methodology and possible sources are represented by:

- Goal and scope
  - Functional unit
  - System boundaries
- Inventory analysis
  - Allocation, i.e. choice of the procedure to allocate environmental impacts for multi-output processes, multi-waste processes and open-loop recycling
  - Waste handling of long-life products, i.e. choice how to asses future situations
- Impact assessment
  - Number of impact categories
  - Impact definition
  - Time horizon of impacts
  - Spatial horizon of impacts
  - Expected technology trends.

These uncertainties resulting from methodological choices cannot be eliminated but could be made operational with the help of scenario analysis, probabilistic simulation and cultural theory perspectives.
Scenario analysis can show the effect on LCA outcomes of several combinations of choices by identifying the relevant alternatives and performing sensitivity analysis (Huijbregts, 1998), but it does not allow evaluating belief-related uncertainties of each methodological choice. Cultural theory perspectives are fixed and it is sometimes difficult to relate them to the practitioner's belief in specific choices. Probabilistic simulation is a detailed approach and is considered a possible way to evaluate belief-related uncertainties affecting LCA results. Probability theory is well suited to represent precise results of mutually exclusive (independent) events.

Like for model uncertainty, also in this case scenario uncertainty is analysed together with parameters and sometimes with model uncertainty, as proposed by Huijbregts et al., (2003). The authors developed a methodology in which the procedure to identify scenario uncertainty foresees the identification of the potential sources of scenario uncertainty, and then a non parametric bootstrapping[17] procedure is proposed to quantify the resulting output uncertainty. With this approach, the different types of uncertainty are treated simultaneously and the results indicate the great influence of scenario and model uncertainty, although they were not quantified comprehensively and although the results cannot be extended to other problems and applications.

There are approaches that make use also of different techniques, like the one proposed by Basson & Petrie (2007). Here both technical and valuation uncertainties (to be read as parameter and scenario uncertainties) are treated, by means of "principal component analysis (PCA)", which facilitates the rapid analysis of large numbers of parallel sets of results; moreover, PCA *"allows for the graphical representation of the performance of a set of alternatives in such a way that it is possible to determine whether the alternatives are distinguishable from one another, and in which performance criteria the key differences and similarities among the alternatives occur* "(Basson & Petrie, 2007).

A different approach to uncertainty analysis has been proposed by Benetto et al. (2006) and relates to the application of Possibility Theory in LCA for uncertainty analysis in complement to classical approaches, i.e. the probabilistic approaches. The use of possibility theory is considered more appropriate for evaluating belief in LCA results, i.e. uncertainties due to methodological choices, because of the inability of probability calculations to take into account the independency of events. Indeed, it is also difficult to identify all the possible events, as the probabilistic framework requires.

[17] Bootstrapping is the practice of estimating properties of an estimator (such as its variance) by measuring those properties when sampling from an approximating distribution. The advantage of bootstrapping over analytical methods is its great simplicity - it is straightforward to apply the bootstrap to derive estimates of standard errors and confidence intervals for complex estimators of complex parameters of the distribution, such as percentile points, proportions, odds ratio, and correlation coefficients.

The approach is time and resource consuming, as Monte Carlo simulation is, but could be worth applying in specific situations when it is difficult to make methodological choices. In any case, the two formalisms (probability and possibility) are not in contrast, but complement each other and could be considered simultaneously.

It is apparent that major efforts should be spent on scenario uncertainty: it should become routine practice, due to the influence of choice to the final LCA results.
Due to the complexity and variety of choices and sources of uncertainties, scenario uncertainty is easier treated at qualitative level but, depending on the contexts, also a rule of thumb method could be useful to evaluate case studies where quantitative uncertainty analysis is infeasible. Some authors (Hellweg et al., 2005) suggested modelling quantitatively only choices of specific interest for goal and scope.

Parameter uncertainty was the most frequently addressed type of uncertainty, but it is not possible to determine whether it is the most commonly recognised form of uncertainty, because it is considered the most important or simply because data for characterising this type of uncertainty are easily available (Lloyd & Ries, 2007).
At a general level, for all the types of uncertainty identified, more guidance is needed in terms of guidelines on the definition of uncertainty in LCI and LCIA, together with an increased number of good practices. Indeed, neither guidelines, that could help practitioners to set the analysis and to identify significant sources of uncertainties, nor guidance on the following aspects, are available:

- to clearly describe the types of uncertainty analyzed;
- how to quantify uncertainties and to choice related method,
- to suggest qualitative approaches for the uncertainty evaluation when quantitative methods are not applicable;
- to involve stakeholders in the modelling of the process to reduce the uncertainty due to model and scenario;
- to explain how to clearly report results of uncertainty obtained in size, importance, and influence, to help decision makers in the interpretation;
- to identify a series of method to explain how to present the results of the uncertainty analysis.

5.5.2. Data quality assessment

Two different approaches exist to make data quality assessment operational (Van den Berg et al., 1999): the probability distribution function method and the qualitative indicator method. The first approach allows a quantitative evaluation of specific groups of parameters, the second one allows dealing with missing data or data from sources unknown or poorly described. The use of both of them is recommended to evaluate uncertainty in LCI. The review (Van den Berg et al., 1999) includes all relevant arguments of the entire topic and bibliography up to 1998.
This paragraph treats only the qualitative indicator method, while the quantitative ones have been discussed in the previous paragraph on parameter uncertainty.

A conceptual distinction must be made between quality goals and quality indicators: the quality goals are defined in the goal and scope definition phase of LCA and define the requirements of data quality; the quality indicators are a measure of the fitness for use. A qualitative indicator method consists of defining the attributes of the data in question and assigning a score to them. To do this, a “Pedigree Matrix” can be defined, in which each row corresponds to an attribute of the data and each column to a score, ranging from good to poor. A basic approach of data quality management and use of data quality indicators (DQIs) is the work of Weidema & Wesnaes (1996). They proposed a formal procedure with the use of a pedigree matrix and 5 data quality indicators for the measure of the following attributes:

- reliability of the data (assessment of the sampling methods and verification procedures),
- completeness of the data (statistical representativeness),
- temporal, geographical and technological correlations (in comparison with the data quality goals).

The DQIs are semi-quantitative numbers, which represent the quality of the data and may be used to evaluate the consistency of the quality of the collected data in relation to the data quality goals. Moreover, they may also be used to identify main sources of data uncertainty. In Maurice et al. (2000), for example, a procedure is proposed which combines qualitative and quantitative approaches in uncertainty assessment and where the selection of the probability functions concentrates on data with a significant contribution to the cumulative results and/or with a high uncertainty, qualitatively evaluated. In this case the authors decided to use an aggregated indicator, but not to transform it into a probability distribution, as proposed in other case studies (May & Brennan, 2003; Lewandoska et al., 2004).
Different sets of data quality indicators have been proposed through case studies and different uses of the scores can be recorded (May & Brennan, 2003; Lewandoska et al., 2004; Rousseaux et al., 2001). An LCA data documentation format according to ISO 14048 makes it practicable to classify data in a Pedigree Matrix, even if with a certain amount of subjectivity (scoring cannot be seen as objective). However, the subjectivity does not compromise the utility of the method for data quality management and communication of data quality in a simple way.

The validity of aggregating DQIs and translating them into distributions for propagating uncertainty is under discussion (Lloyd et al., 2007). Weidema (1998) observed that the score does not represent an ‘amount’ of data quality (e.g. a score of 4 for an indicator is not necessarily twice as problematic as a score of 2) and then he disagrees with the aggregation of the indicators suggested for example by Wrisberg et al. (1997) and Lindeijer et al. (1997) (discussed in Van den Berg et al. (1999)).

As regard the use of DQIs to evaluate the so called ‘additional uncertainty’, expert judgment is required and in this case it may be easier for experts to identify ranges of foreseeable values rather than more abstract DQIs (Lloyd et al., 2007).

In May & Brennan (2003) two different methods of combining numerical and qualitative uncertainty have been compared in a case study concerning an LCA of Electricity Generation. The conclusions highlight the following points: the methods produced very different results; there was no evidence that either approach produced results more accurate, or more representative of qualitative uncertainty; it could not be demonstrated that either method produced a measure of uncertainty more relevant than that of the numerical uncertainty method alone. Other authors (Lewandowska et al., 2004) are more positive on the use of combined methods.

On the basis of the different approaches/experiences presented in literature, a generalisation of the uncertainty assessment, including the use of DQIs, can be recommended. Moreover, a simplified method for a reliable uncertainty assessment should be developed in order to use LCA in early phases of product development (Maurice et al., 2000). The need of simplifications to reduce the need for detailed analysis of each datum and the time required for data quality analysis is also stressed in May & Brennan (2003).

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5.6. Cross issues

There are topics that cannot be classified as part of a specific step of the LCA standard procedure, because they are horizontal to the methodology, aiming at broadening its scope and/or improving its applicability. In this review they include (Environmental) Life Cycle Costing (LCC), Social Life Cycle Assessment (SLCA), and Simplified LCA.

5.6.1. (Environmental) Life Cycle Costing

The object of this review is environmental Life Cycle Costing (LCC), which differs from conventional Life Cycle Costing because of the following main elements: product system modelled, system boundaries, actors involved, reference unit, cost categories and cost model.

A sound survey on environmental LCC has been performed by Ciroth et al (2008), as a result of the activity performed by the SETAC Working Group on Life Cycle Costing. The authors define LCC as a technique that considers "[…] all costs associated with the life cycle of a product that are directly covered by one or more actors in that life cycle […]. Externalities that are expected to be internalised in the decision-relevant future comprise real money flows as well, and they must also be included". One of the main features of LCC is that it shares the same LCA structure, i.e. they have equivalent system boundaries and functional units, because they are built upon the same product system providing the same function, and have a steady-state nature.
Indeed, the framework developed can be easily linked together with an LCA, without generating overlaps, because impact assessment indicators deriving from LCA are not translated into monetary terms but are kept separate.
The work performed by Ciroth et al. (2008), whose publication is forthcoming, represents a fundamental step in the LCC development, since it addresses the question of how costs and environmental aspects can be combined and provides a clear guidance for performing LCC studies. The approach, which represents the evolution and the completion of those presented by the same authors like Rebitzer & Seuring (2003) and Hunkeler & Rebitzer (2003; 2005), just to mention some, contributes to the development of a code of practice for LCC and leads to a potential standardisation in analogy to ISO 14040 series.

Not all previous works are aligned to this statement, starting from a different use of the terminology. For example, Reich (2005) proposed a terminology and methodology for the economic assessment of municipal waste management systems, which include financial LCC, life cycle costing (which is used in parallel with an LCA) and environmental LCC. The last is intended as the weighting of environmental impacts of an LCA system in monetary terms; in particular, it makes use of three different weighting methods to monetarise environmental effects such as emissions and resources.

The results from financial LCC and environmental LCC are thus combined in order to provide an economic methodology useful to make environmental aspects directly

comparable to the economy of the studied options. This approach has been tested through a case study, by using the ORWARE model (Assefa et al., 2005). Some open questions emerged, in particular with reference to comprehensiveness and consistency in both theory (how to deal with timing of emissions and economic activities, and system boundaries) and data, together with a difficulty in communicating results to the actors of the waste management sector.
A different approach has been proposed by Nakamura & Kondo (2006): they developed a hybrid LCC methodology, called WIO price model, which builds upon the hybrid method of LCA based on WIO (Kondo & Nakamura, 2004; 2005), and illustrated it by a case study of electrical home appliance under alternative end-of-life scenarios.

Beyond conventional and environmental LCC, also societal LCC should be mentioned, introduced by Ciroth et al. (2008) as the third type of LCC. Societal LCC takes a society perspective, and includes all of the environmental LCC plus additional assessment of further external costs: it means that it should include the monetisation of externalities. The use of "should" is mandatory as the identification and quantification of externalities is strongly affected by high uncertainties, and thus their inclusion represents a great challenge for the methodological development. The approach is still under development, and it is suggested to deepen its relation with SLCA: indeed, as societal LCC is used to quantify environmental effects on society in monetary terms, it is considered an important ingredient for performing "sustainability" evaluation.
But social aspects are not simply a quantification of environmental effects on society in monetary terms: more complexities exist, more interrelations that need to be accounted for. Perhaps, the possibility of finding a point of contact between two approaches should be investigated, in order to avoid spending too much efforts in developing methodologies that could have a limited application.

5.6.2. Social Life Cycle Assessment

The development of a Social Life Cycle Assessment (SLCA) that includes the assessment of social aspects related to the life cycle of a product can give a contribution to the interpretation of its sustainability by stakeholders and decision makers.
Even if in its infancy, SLCA is object of an increasing number of published papers, thus demonstrating the existing interest about the methodology and its application. Recently a comprehensive review has been published (Jǿrgensen et al., 2008), where the existing methodology and proposals are presented and discussed. The methodological framework adopted, based on the ISO-LCA structure, was proposed by the taskforce "Integration of social aspects in LCA", nominated in the context of the UNEP-SETAC Life Cycle Initiative (Grießhammer et al., 2006).
In this context, several aspects have been discussed, both related to the methodology and to the **scope of the analysis**.
The first aspect under discussion is if social impacts are related to processes or to the conduct of companies carrying out those processes, as Dreyer et al. (2006) suggest. In this latter case, where the causal link is different than the one of an environmental LCA,

more problems of allocation arise, which introduce bias in the assessment. To face them under this approach, Dreyer et al. (2006) propose that the allocation principle should reflect the company's importance in the overall life cycle and could be based on:

i) value creation, which would require that monetary input and output for each company or for each life cycle stage be used;
ii) number of hours spent at the company for functional unit;
iii) material costs and product price for the company in the product chain.

The definition of the **system boundaries** in SLCA is also matter of discussion in the scientific community. Some authors point out that the choice depends on the goal of the study: if the focus is on product comparison, a full assessment is necessary; for supporting management decisions it could be enough to include only those part of the life cycle which can be directly influenced by the company. In Dreyer et al. (2006) some general criteria for setting system boundaries are presented and discussed. If compliance with ISO 14044 is recommended, as in Weidema (2005), exclusion of life cycle stages indeed should be accepted only if it does not significantly change the overall results of the study.

Another important aspect to consider is the **selection of indicators**. A good indicator, in agreement with Weidema (2006), allows 'quantification of the extent, the duration and the severity of the considered aspect'. Some SLCA approaches use inventory results, other midpoints or endpoint indicators: attempts of separating them are often confused and in some cases it is difficult to express the cause–effect relationship between midpoint and endpoint. Which type of indicators to use is under discussion: endpoint indicators have the advantage that no subjective weighting is needed, but require the impact pathway be known; midpoint indicators are closer to the activities and for this reason more understandable for decision-makers. The UNEP-SETAC task force recommends combining inventory, midpoint and endpoint indicators, starting with the first two types of indicators and thinking about adding the third ones later on. They also stress the need for well discussed indicators and indicator-sets for SLCA (Grießhammer et al., 2006).

Two types of questions have been raised concerning **formulation of indicators** (Jǿrgensen et al., 2008):

- Should indicators be formulated in quantitative, semi-quantitative (scoring systems) or qualitative terms?
- Should the indicators measure the impact directly or based on indirect indication/proxy measurements?

The latter aspect relates to the problem that sometimes the direct measurement does not reflect the actual problem (see for example Dreyer et al., 2006;) or to the problem of data availability.

For the first question, the UNEP-SETAC task force suggests a combination of quantitative, semi-quantitative and qualitative indicators in order to produce the most accurate and relevant assessment possible.

Some **example of social indicators** proposed for SLCA studies are here given.

- Hunkeler (2006) suggested the use of a methodology, which focuses on the work hours required to meet basic needs, transforming the life cycle inventory into labour units, a unique indicator that can be used to compare different solutions.
- Norris (2006) proposed the use of human health as indicator of socio-economic status: he suggested that they are directly linked as the grown in well-being is followed by a grown in the economy and used the Eco Indicator 99 methodology to evaluate health impacts in terms of life years lost, measured in disability-adjusted life-years (DALYs).
- Weidema (2006) proposed the QALY (Quality Adjusted Life Years) indicator, deriving from aspects linked to the human life intrinsic values like: life and longevity; health; autonomy; safety, security and tranquillity; equal opportunities; participation and influence. This last procedure measures the well-being in a single value that could be compared to the monetary index of other procedures like for example Cost Benefit Analysis or Life Cycle Costing.
- Dryer et al. (2006) observed that there are two layers of SLCA, a mandatory one, driven by normative, and an optional one, that respects specific interests. They focussed their work on the development of a methodology for the mandatory part of the SLCA, which includes the minimum expectations for a social responsible company.

Besides the selection of indicators, another challenging aspect of SLCA is **data collection**. Some authors claim that the use of generic process data, as in environmental LCA, is irrelevant because social impacts have to deal with the behaviour of each company (Dreyer et al., 2006). However, using generic data can give an estimate for a certain number of social impacts, while collecting site-specific data is a very demanding task and guidelines on monitoring approaches would be necessary. In general, a problem of data availability and reliability exists. It seems that further discussion and case studies are needed in order to reach a common position on this subject and develop agreed guidelines.

The impact assessment phase of SLCA can be analysed following the steps of ISO 14044 for environmental LCA. In the classification step the indicators are arranged in impact categories, but also another type of classification, the ‘stakeholder approach’, has been proposed. The UNEP-SETAC task force has agreed that the two approaches are not incompatible and that four main stakeholder categories can be identified: workforce, local community, consumers (for the use stage), society (national and/or global).

The review of Jørgensen et al. (2008) presents a certain number of approaches to the characterisation of social indicators, but concludes that the trend seems more oriented towards simplification of inventory results than towards a characterisation in line with the environmental LCA methodology.

Several authors suggest normalisation and valuation in SLCA, but very little work has been done on this subject (Jǿrgensen et al., 2008).

The UNEP-SETAC task force has also pointed out some aspects of the interpretation phase of SLCA. It should include checks of completeness and consistency, but also on the relevance of information provided and on the engagement of stakeholders. Moreover, they consider that the process of evaluation of SLCA is fundamentally subjective, so that a plurality of evaluation and of weighting methods can be accepted.
The conclusions of the feasibility study (Grießhammer et al., 2006) are topical issues up to now and can be summarised in the necessity of:

- conducting case studies with the key elements that have been discussed above in order to find a solution to the numerous open questions and
- composing a 'Code of Practice' for SLCA.

5.6.3. Simplified LCA

The need for developing simplified (or streamlined) LCA and life cycle thinking (LCT) simplified approaches stems from the consideration that detailed LCA can be time and resources consuming and this is an obstacle to a wider adoption of life cycle approaches, especially among SMEs. Moreover, some authors suggest that simplified methods can be useful in the early product design phases, when it is difficult to assess the potential environmental impacts because only a limited amount of information is available (Rydh & Sun, 2005), and in green procurement, to identify critical aspects of products and then criteria for procurement (Hoschorner & Finnveden, 2003). We can distinguish three different types of simplified approaches[18]: *qualitative* (e.g. MET- Materials Energy and Toxicity- matrix, checklists, ABC hot spot screening, expert panels); *semi-quantitative* (e.g. ERPA – Environmentally responsible product assessment- matrix, MECO - Materials, Energy, Chemicals, Other chart, ABC/XYZ assessment, Environment-Failure Mode Effect Analysis) and *quantitative* (simplified LCA).
According to Guinée et al. (2001) "a simplified LCA is as a simplified variety of detailed LCA conducted according to guidelines not in full compliance with the ISO 1404X standards and representative of studies typically requiring from 1 to 20 person-days of work". Efforts to develop streamlined LCA often focus on the life cycle inventory analysis (LCI), which is the most time consuming phase. In agreement with Rebitzer et al. (2004) main approaches to LCI simplification can be the following: the direct simplification of process-oriented modelling; LCA based on economic input/output analysis; the hybrid method, which combines elements of process LCA with input/output approaches.

[18] For the references concerning the methods refer to Park et al. (2006), Hoschorner et al. (2003) and Rebitzer at al. (2004)

I/O analysis and hybrid method are analysed in paragraphs 5.3.4. The simplification in process-oriented modelling can be obtained by reducing the scope of the study, excluding some phases of the system (horizontal cut-off) and/or by reducing data needs (vertical cut) through the use of surrogates, which can be already available, or generic databases for one or more steps of the life cycle (Hur et al., 2005). The identification of relevant stages and stressors, for which primary data have to be collected, requires a preliminary screening step, which can use the above mentioned qualitative and semi-quantitative methods or other quantitative approaches (e.g. I/O LCA, calculation of the cumulative energy demand, assessment of single key substances) (Rebitzer et al., 2004). The simplification of the model of the product system after this screening phase is the most critical, but the least developed, step of the simplification process. Unfortunately, it is not easy to find general methods because the choice depends on the intended application. When detailed LCAs exist, simplifying methods can be identified for those specific applications; when they are not available, research on developing general methods has been oriented to restricting system boundaries with cut-offs and to transferring cut-off procedures from a specific product system to similar systems. It has been observed (Rebitzer et al., 2004) that input-output based hybrid methods could give a good contribution to this development. Another critical step of the simplification process is the data collection, because of the lack of data availability, especially in early design phases, and the confidentiality of some information (Mueller et al., 2004). Exchange of relevant data along the supply chain and the possibility of having averaged data concerning a product range rather than specific products could represent a solution to this problem. Different studies have been produced aimed at developing methods for generating generalised data (ex. parameterised inventories (Mueller et al., 2004), LCIs for groups of materials (Rydh & Sun, 2005), LCI of chemicals (Geisler et al., 2004; Hischier et al., 2005).

Methods that help dealing with data gaps can also facilitate the impact assessment as well. A first example is the method developed by Fleischer et al. (2001), the semi-quantitative impact assessment ABC/XYZ method. It is an integral part of the Design for Environment (DfE) software tool euroMat and can deal with incomplete knowledge on LCI of non-energy related emissions with respect to their quality (what is emitted) and/or with respect to their quantity (how much is emitted). The cumulative energy demand indicator (CED) represents energy-related impacts. It is suggested that outside DfE, the method should be capable of facilitating simplified LCAs in general, but, in order to guarantee consistency, the method should be applied as a stand-alone method, without advocating the LCA for the impact assessment evaluations. Some open questions remain: what shall be done if quality and quantity of an emission are not known? Does the method require a reduced effort compared with closing the data gaps and applying "normal" LCIA methods?

Huijbregts et al. (2006) have published a paper about the appropriateness of the fossil Cumulative Energy Demand (CED) as an indicator for the environmental performance of products and processes.

They have carried out a regression analysis between the environmental life-cycle impacts and fossil CEDs of 1218 products. This correlation analysis was based on cradle-to-gate and waste treatment data, further research would be required for a correlation on a cradle-to-grave basis. However, within this scope, they have shown that the fossil CED correlates well with most impact categories, such as global warming, resource depletion, acidification, eutrophication, tropospheric ozone formation, ozone depletion, and human toxicity (explained variance between 46% and 100%) and that it may therefore serve as a screening indicator for environmental performance. They also observed that the usefulness of fossil CED as a stand-alone indicator for environmental impact is limited by the large uncertainty in the product-specific fossil CED-based impact scores especially due to non-fossil energy related emissions and land use.

Other proposed proxy indicators for the environmental performance of products are materials flow based indicators, such as MIPS (Material Input Per Service unit). MIPS is an elementary measure to estimate the environmental impacts caused by a product or service, which includes the whole life-cycle from cradle to cradle (extraction, production, use, waste/recycling). The method of calculation is described in (Ritthof et al., 2002). In (van der Voet et al., 2004) the role of Mass-Based indicators versus LCA is discussed. Van der Voet et al. (2004) published the results of a study performed to support the Dutch environmental policy of dematerialization by the use of a methodology that combines LCA and MFA (Material Flow Accounting). In their case study, they found that the mass flows of an individual material are not indicative of its environmental performance, but on a more aggregate level, mass-based and impact-based indicators seem to point in the same direction. They suggest that further case studies should be carried out in order to verify if mass-based indicators could be good indicators of the environmental performance of products.

The choice of the most suitable simplified method, or combination of simplified methods, depends on the type of results users are looking for. In Hoschorner & Finnveden (2003), for example, two simplified methods (ERPA matrix and MECO-method) were evaluated and compared with the results of a detailed LCA. They observed that a differentiation could be made between a study aimed at supporting a choice between several alternatives and a study aimed at identifying critical aspects and suggesting mitigation strategies. In the first case, it is much more important to have quantitative data than in the second case. In fact, in the first case the lack of a quantitative dimension would hinder the comparison and make it difficult to differentiate between products. On the other hand, problems could arise when in quantitative LCA, aspects that are difficult to quantify are handled qualitatively, because this qualitative information is often overlooked. Therefore, they propose the use of the ERPA method as a checklist to identify critical aspects, because the arbitrariness of the scoring system does not allow quantitative comparisons. They found the MECO-method more suitable for comparing alternatives because it allows adding a quantitative dimension to the qualitative evaluation.

Moreover, they suggest the use of this method as a complementing study to an LCA to overcome the problem of neglecting qualitative information in the interpretation phase. Hur et al. (2005) arrive at the same conclusion: the ERPA method can be used in eco-redesign to identify the potentials for improvement and alleviate harmful environmental impacts since it identify areas where environmental improvement are needed and can be made.

Another work deals with the use of combined simplified methods for ecodesign purpose. Park et al. (2006) propose an ecodesign method which combines a bottom-up approach (first identify environmental weak points of a system, second generate ecodesign ideas) and a top-down approach (use ecodesign strategies to generate ecodesign ideas). They have evaluated different bottom-up approaches and have proposed a method for consumer electronics, where screening LCA, based on literature information, is used to identify the key life cycle stage: when the key life cycle stage is not the manufacturing stage the environmental benchmarking method is employed to identify key environmental aspects; to identify environmental weak points of the manufacturing stage the checklist method was used instead. Panel of experts was used to determine relationships between benchmarking or checklists results and corresponding ecodesign strategies. The authors found that the use of this combined method minimizes time and money resources because it allows identifying weak points only within the key life cycle stage.

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5.7. Map of the main tools

The present analysis deals with operational aspects, by mapping the main tools (software and databases) available on the market, in support of LCA practitioners, in order to analyse if the main *tools* (software and databases) available on the market are in line with the methodological developments highlighted in the literature review.
42 software and 26 databases have been mapped and classified according to a set of predefined parameters. Inputs for the analysis derive mainly from information collected by JRC-IES starting from 2005, when a questionnaire was sent out to the tools providers in the worldwide LCA community. The results of that survey, with further updating up to June 2007, are currently available in the LCA Resource Directory on the website of the European Platform on LCA (http://lca.jrc.ec.europa.eu/lcainfohub/directory.vm). Further information has been collected in the specific websites of these tools.
Even if a whole coverage is not assured, the most widely used software and databases are included in the list.

5.7.1. Methodology

The software have been mapped according to the following parameters:

- being **sector specific**;
- usability for **screening LCA**[19];
- suitability to **eco-design** approach;
- possibility to perform **life cycle cost**;
- possibility to perform **social life cycle analysis**;
- applicability for **laws/regulatory compliance**;
- completeness of life cycle steps (**goal and scope definition, inventory, impact assessment, interpretation);**
- possibility to perform **hybrid LCA**[20].

For the databases, some different parameters have been considered:

- being sector specific;
- being country specific;
- possibility of Input/Output data integration;
- inclusion of data for Life Cycle Cost;
- inclusion of data for Social Life Cycle Analysis;
- suitability for consequential[21] LCA.

[19] Screening LCA can be defined as an initial simplified analysis that aims at identifying whether more in depth assessments are needed and the type and level of these assessments.
[20] Hybrid LCA combines process-based LCA with environmental input/output analysis.
[21] While attributional LCA aims at describing the environmental properties of a life cycle and its subsystem, consequential LCA aims at describing the effects of changes within the life cycle.

In case of no available data for the characterization of one or more parameters in the LCA Resource Directory, a further search has been made on the publicly available information on the providers/tools websites. However, neither specific requests nor direct contacts with the providers have been taken during the survey.

5.7.2. Main findings

The table below summarizes the results of the mapping and classification of the 42 software, according to the predefined parameters.

SOFTWARE										
Result	**Sector specific**	**Screening LCA tool**	**Eco-design approach**	**Application for laws or regulatory compliance**	**Life Cycle Cost**	**Social Life Cycle Analysis**	**LCA**			**Hybrid LCA**
							LCI	**LCIA**	**LC Int.**	
Yes	21	5	21	2	20	4	34	32	27	3
No	21	37	19	40	21	38	0	3	3	39
N/A	0	0	2	0	1	0	8	7	12	0

Table 2 - Summary of results for software

The following main considerations can be derived.

- All software enable the users to perform LCA studies based on a “traditional” approach; the missing information are more likely due to inaccuracy in filling in the questionnaires, rather than to a lack of functionality in the tool.
- A significant number of software is sector specific, addressing building materials, waste management and forest/agricultural products.
- A considerable number of tools can support eco design, but very few of them look suitable for screening LCA and regulatory compliance.
- Regarding innovative and broader LCA approaches, about half of the available tools may support Life Cycle Cost modelling, but only a few are suitable for hybrid approaches as well as for the integration of social aspects into the analysis

The Table 3 below provides the results of the analysis carried out on the 26 databases.

Result	Sector specific	Country specific	I/O data integration	Data for LCC	Data for SLCA	Consequential
Yes	12	6	3	7	2	0
No	14	20	0	0	0	0
N/A	0	0	23	19	24	26

Table 3 - Summary of results for databases

Conclusions similar to those related to software can be drawn, but there is a far larger amount of unavailable and uncertain data.

- A considerable number of databases are sector specific, addressing building materials, waste management, electric and electronic products, iron and steel, food production chain, forest/agricultural products, polymers and other chemicals.
- Most of the databases include a full geographic coverage, while some of them are country specific, mainly about Japan and the United States.
- With reference to innovative and broader LCA approaches, the available data do not allow for a proper classification. Some databases include explicitly economic data for Life Cycle Cost modelling; few others include input/output tables for data integration and data for social LCA. In most cases, no information can be derived about the presence of data on social aspects, but since most of the software do not include the Social LCA, it can be assumed that databases do not contain such data. Even the information on the applicability to consequential LCA is not obtainable from the current available information.

The analysis performed showed that at present there is a wide choice of LCA software and databases for different sectors and applications within the boundaries of the traditional ISO-LCA. Some tools support also the integration of economic information associated to the environmental aspects of products and services, but the modelling of social aspects is feasible in very few cases.

The analysis showed also that LCA tools are generally not considered for the assessment of compliance with legislation. This is probably due to the so far limited number (at least for Europe) of laws based on a life cycle approach.

In general, the capabilities for broadening and deepening LCA are still missing in the main LCA software and database used. The development of such capabilities will require a considerable effort in the design of innovative tools as well as in the definition of new data quality requirements and in the collection of new data.

6. CLOSURE NOTES

The critical review of the scientific literature allowed exploring LCA in its depths, highlighting a multitude of approaches and new thoughts. LCA applications have been increasing in number in the last years, probably due to both pull and push actions: a pull from business and consumers with a greater awareness of the need of adopting more "sustainable" behaviours, and an increasing push from the regulatory process at national and supra-national level, incorporating a life cycle approach.

The developments identified are oriented towards two main directions:

1. improvement of the most debated issues (system boundaries and allocation) and
2. further development of the methodology, in order to understand if and to what extent the present LCA can give an answer to more comprehensive problems, like those related to sustainability.

LCA, as conceived now in ISO 14040 series, considers only environmental dimension, with several assumptions: it is more suited for applications at micro level as the model is steady state and linear, impacts are *potential* impacts. These limitations represent the main driver for new developments, in particular in the inventory and impact assessment phases. Developments show different degree of "hardness", directed towards deepening and broadening the methodology, respectively towards the inclusion of more mechanisms (cause-effect relations) with sophistication in the modelling and towards the expansion to economic and social aspects.

The consequential school has given rise to a new mode to conceive LCA, with consequences on the majority of the methodological issues, such as allocation, system boundaries, modelling changes over time, etc. For this reason, consequential LCA can be seen as a sort of starting point for a number of new approaches.

Other approaches see an increasing use of other methodologies combined with LCA, like Input Output Analysis (IOA). The combination/integration with other tools puts on the table the question of how far we should go in "improving" LCA.

On the Impact Assessment side, developments include:

- new impact categories and methods
- improvement of existing methods,
- inclusion of more aspects like spatial and/or temporal differentiation, and
- the definition of a common framework for the development of mid-point and damage oriented methods.

Generally, it is apparent that developments of such importance would require the involvement of different disciplines and expertise at several levels; hence the contamination with other disciplines could play a more and more important role. At present, most of the authors advocate the importance of that involvement in different phases of the methodology, in order to:

- better model the system,
- reduce uncertainty,
- collect more representative data,
- define scenarios,
- include mechanism belonging to the sphere of micro economics, etc.

Indeed, this need so far is only worded and a real involvement does not seem to be very common.

This attitude could be understood as a sort of closure of the LCA community, caused by a concern that the contribution of other disciplines to the further development of LCA methodology could violate its inherent principles as defined in ISO standards. Indeed, this concern could be justified by the fact that LCA was very criticized in the past and, only with the standardisation, it gained again a reputation. Therefore, any major change of ISO-LCA could endanger again its credibility. Thus, some questions arise:

- To what extend could such expanded LCA be considered still LCA?
- To what extent the inclusion of economic and social element would allow a more comprehensive evaluation of sustainability concept?
- Would these contaminations with other disciplines be desirable?
- Does the increased complexity make available reliable information, with a tolerable uncertainty?

Maybe all these questions could be faced with a debate on scientific journals. Indeed, on several issues we did not find any real debate: if one author proposes an approach for a specific issue, comments/critics on it (besides those from reviewers, but they are not available to the public) are not very common. It is not (always) true that a lack of debate on a specific issue is a symptom of consensus on the approach and vice versa. In this way, it is not always possible to properly evaluate from a literature review, the consensus on and the importance (relative and absolute) of a new approach. In any case, despite the fact whether a broader and deepened LCA would be feasible and practicable, it is clear that needs for an improved LCA cannot be ignored. Overcoming the present limits of ISO-LCA and further improve it is a clear need. The review showed that first suggestions and solutions are already on the table, starting from the consequential approach, which would potentially have consequences at different level in methodological development. Indeed thinking in consequential way means thinking about the consequences of the actions, about the interrelations and thus it means to project the problem at market level, with all its dynamics. In this context, partial equilibrium modelling, experience curves and rebound effects becomes relevant, since they introduce micro-economic mechanisms in LCA models. On this side, fruits are still unripe: many efforts are necessary, both at conceptual and practical level. Starting out by involving expertise outside LCA community could help easier overcoming many difficulties. Efforts should be spent with this regard, because the inclusion of micro-economic relations stands out as one of the most important elements in order to further develop the methodology.

Some of the other issues can be already considered as ripe fruits, in particular those related to:

- system boundary,
- allocation and
- data quality assessment.

Indeed, the debate could never end but some elements might be already made available to experts, working for reaching consensus on the procedural side more than on the analytical one.

Efforts in improving LCA should be strongly encouraged, and their developments have to go hand in hand with two main aspects: availability of tools and user needs.
The review allowed analysing whether the main tools (software and databases) available on the market are in line with the methodological developments highlighted in the literature. The results show that at present, despite the wide choice of LCA tools for different sectors and applications, the capabilities for broadening and deepening LCA are still missing in the main LCA software and databases. The development of such capabilities will require a considerable effort in designing innovative tools as well as in defining new data quality requirements and in collecting new data.
Regarding users needs,[22] the importance of making available methods and tools simple to be used is raised by several authors, and many proposals have been published but no progresses in developments have been identified by the end of '90s: existing applications are based on the methodological developments of those years. Thus, further efforts should be spent in developing simplified tools, and at the same time on reaching consensus on the procedural side, e.g. by introducing simplification to reduce the need for detailed analysis of each datum and the time required for data quality analysis.

[22] CALCAS faces the question of user needs with a dedicated work package, WP6. Its deliverables are available at www.calcasproject.net

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