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# SOLUTIONS TO IMPROVE THE RELIABILITY AND APPLICABILITY OF LIFE CYCLE ASSESSMENT IN INDUSTRIES

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# Summary

Since its first applications in the late 1970, Life Cycle Assessment (LCA) methodology has increasingly spread as an effective environmental management tool and has nowadays become a well-established analytical method to quantify the environmental impacts of products and processes at industrial level.
Despite its popularity and codification by organizations such as the International Organization for Standardization, the Society of Environmental Toxicology and Chemistry and harmonization initiatives promoted at European level (i.e. by the European Platform on LCA) through the publication of the ILCD Handbook, LCA methodology is still under development and there are a lot of open research issues which are widely discussed in literature.
Considering the increasing number of applications at industrial level, LCA is recognized not only as a field of scientific research but also a business of growing importance, whose successful development requires both influx of new ideas and harmonized methodologies and reliable and credible applications.
The primary needs of industries on LCA rely on the definition of tools for increasing the reliability of LCA results and solutions for enhancing the applicability of the methodology.
The present research mainly focuses on how to translate into practice these two aspects, with the aim of providing practical solutions for improving the reliability of the methodology and facilitate its application at industrial level.
The methodology applied is based on the conduction of multiple case studies, analyzing different sectors, both at product (tissue paper, milk packaging, wooden pallet) and process level (agricultural processes, wastewater treatment). Furthermore, the main applications of LCA methodology within the private sector are considered: comparison between different solutions, improvement of environmental performances and design of new products.
The aspect of how to increase the reliability of LCA results is addressed in the first part of the work, including the first three chapters. The starting point is that both the credibility and transparency of an LCA study can be enhanced by giving more attention to quantifying uncertainties. Uncertainties considered in the present work are connected with parameter uncertainty (connected with data variability) and scenario uncertainty (connected with subjective choices). Therefore the role of the two techniques identified by the ISO 14040 standards to analyze uncertainties during the interpretation phase of LCA, i.e. uncertainty analysis and sensitivity analysis, is discussed.
The second part of the work deals with the definition of simplified tools tailored to users' requirements in order to increase the applicability of the LCA methodology.

Streamlined tools are mainly connected with the definition of approaches to simplify the Life Cycle Inventory phase, which is the most time and resource demanding phase of an LCA study. Especially for small- and medium-size enterprises that rarely have the knowledge and resources necessary to implement LCA, a priority is indeed to facilitate the access to reliable, accurate, and relevant life cycle information, reducing efforts connected with data acquisition.

The research activities were mainly carried out at the Dipartimento di Processi Chimici dell'Ingegneria (now Dipartimento di Ingegneria Industriale), Università degli Studi di Padova, Padova (Italy) and Department of Environmental Science, Aahrus University, Roskilde (Denmark).

The results of research activities are here summarized in seven chapters:

**Chapter 1** includes an introduction to Life Cycle Assessment methodology as defined by ISO 14040 standards, as well as a discussion of its main methodological limits, focusing on the issue of uncertainty quantification and the use of streamlined techniques.

**Chapter 2** presents a case study in the agri-food sector, namely a comparative LCA between conventional and organic farming in a three year cycle including soybean in the first and third year and barley in the second year.

**Chapter 3** introduces a case study in the tissue paper sector, where LCA methodology was applied in order to compare three different type of wipers for professional use made from different raw materials: virgin pulp, waste paper and fibres recovered from the recycling of beverage cartons.

In **Chapter 4** the results of the application of LCA to evaluate and compare the environmental performances of four different technological solutions representative of Danish Waste Water Treatment Plants is presented, with a focus on the final sludge treatment. This study was performed during the internship at the Department of Environmental Science, Aarhus University, Roskilde, Denmark .

In **Chapter 5** starting from a case study of two milk containers, a laminated carton container and an High Density Polyethylene bottle, the significance of the use of the LCI indicator non renewable fossil CED as proxy indicator in the beverage packaging sector is discussed, in order to detect those situations in which companies can benefit from the use of proxy indicators before a full LCA application.

**Chapter 6** discusses the use of parametric Life Cycle Inventory models as a support in the design phase of new products in the wooden pallet sector through the definition of correlation between the most influent parameters and the environmental impacts. A LCI parametric model is set to define the life cycle of a series of wooden pallets with similar characteristics and tested with one reference product.

Finally, conclusions and proposals for future work can be found in **Chapter 7**.

# Sommario

A partire dalle prime applicazioni alla fine degli anni Settanta, la metodologia Life Cycle Assessment (LCA, valutazione del ciclo di vita) si è sempre più diffusa come un efficace strumento di gestione ambientale ed è diventata al giorno d'oggi un metodo consolidato per quantificare gli impatti ambientali di prodotti e processi a livello industriale.

Nonostante la popolarità raggiunta e le iniziative volte a codificarla ad opera di organizzazioni quali l'Organizzazione Internazionale per la Standardizzazione (ISO) e la Società di Tossicologia Ambientale e Chimica (SETAC) e le iniziative di armonizzazione promosse a livello europeo (come ad esempio dalla Piattaforma europea sull'LCA) attraverso la pubblicazione dell'ILCD Handbook, la metodologia LCA è ancora in fase di sviluppo e ci sono numerosi aspetti metodologici che sono ampiamente discussi in letteratura.

Tenendo conto del considerevole numero di applicazioni in ambito industriale, è ormai assodato che l'LCA costituisce non solo un ambito di ricerca scientifica, ma anche uno strumento gestionale di crescente importanza, il cui sviluppo richiede da un lato l'influsso di nuove idee e di una metodologia armonizzata, dall'altro applicazioni credibili e affidabili.

Le principali esigenze delle industrie che si apprestano ad utilizzare la metodologia LCA si identificano in due principali aspetti: da un lato è richiesto lo sviluppo di tecniche che consentano di intervenire sull'affidabilità dei risultati dell'LCA e dall'altro risulta determinante la definizione di soluzioni che mirino ad aumentare l'applicabilità della metodologia.

La presente ricerca si focalizza proprio su queste due esigenze, con l'obiettivo di fornire soluzioni pratiche per aumentare l'affidabilità della metodologia e facilitare la sua applicazione in ambito industriale.

La ricerca è stata strutturata attraverso la conduzione di casi studio multipli, che applicano la metodologia LCA a diversi settori produttivi, sia a livello di prodotto (settore della carta tessuto, imballaggi per bevande, pallet in legno), sia a livello di processo (a livello agricolo e nel settore della depurazione delle acque). I diversi casi studio analizzano le principali applicazioni della metodologia LCA in ambito industriale, ovvero il confronto tra diverse soluzioni, il miglioramento delle prestazioni ambientali di prodotti e processi e le fasi iniziali di progettazione di nuovi prodotti.

La tematica relativa all'aumento dell'affidabilità dei risultati degli studi LCA viene affrontata nei primi tre capitoli del presente lavoro. Il modo per aumentare sia la credibilità che la trasparenza di un studio di LCA consiste nel prestare maggior attenzione alla quantificazione dell'incertezza. Le tipologie di incertezza che vengono considerate nel presente lavoro riguardano l'incertezza di parametro, che è relativa alla variabilità intrinseca

dei dati utilizzati in uno studio di LCA e l'incertezza di scenario, che si relaziona alle scelte soggettive effettuate a livello metodologico. Nella trattazione dei primi tre casi studio viene discusso il ruolo delle due principali tecniche identificate dalle norme della serie ISO 14040 per trattare l'incertezza durante la fase di interpretazione, ovvero analisi di incertezza e analisi di sensibilità.
La seconda parte del lavoro di tesi riguarda la definizione di approcci che consentano di semplificare la conduzione della fase di analisi di inventario, che costituisce la fase più impegnativa in termini di tempo e risorse. Soprattutto per le piccole e medie imprese che raramente hanno le conoscenze per implementare studi di LCA completi, diventa infatti prioritario definire procedure e metodologie che consentano di ottimizzare gli sforzi durante la fase di acquisizione dei dati.

Le attività di ricerca sono state condotte presso il Dipartimento di Processi Chimici dell'Ingegneria (ora Dipartimento di Ingegneria Industriale), Università degli Studi di Padova, Padova (Italia) e il Department of Environmental Science, Aahrus University, Roskilde (Danimarca).

I risultati delle principali attività di ricerca sono riportati in sette capitoli:

Il **Capitolo 1** introduce la metodologia Life Cycle Assessment così come definita dalle norme della serie ISO 14040 e presenta una sintesi dei suoi principali limiti metodologici, focalizzando l'attenzione sulla quantificazione dell'incertezza sull'uso di tecniche semplificate.

Il **Capitolo 2** presenta un caso studio nel settore agroalimentare, ovvero un LCA comparativo tra le tecniche convenzionale e biologica in un ciclo triennale che prevede la produzione di soia nel primo terzo anno e di orzo nel secondo anno.

Il **Capitolo 3** riporta un caso studio nel settore della carta tessuto, nel quale la metodologia LCA è stata applicata per confrontare le prestazioni ambientali di tre diverse tipologie di strofinacci per uso industriale ottenuti a partire da diverse materie prime: cellulosa vergine, carta riciclata e fibre ottenute dal riciclaggio dei contenitori in poliaccoppiato.

Nel **Capitolo 4** sono riportati i risultati dell'applicazione della metodologia LCA per valutare e confrontare le prestazioni ambientali di quattro diverse soluzioni tecnologiche rappresentative degli impianti per il trattamento delle acque danesi. Una approfondimento particolare viene riservato alla fase di trattamento finale dei fanghi. Questo studio è stato sviluppato durante il periodo svolto presso il Department of Environmental Science, Aarhus University, Roskilde, Danimarca.

Nel **Capitolo 5** partendo da un caso studio di LCA comparativo tra due contenitori per il latte, ovvero un contenitore in poliaccoppiato e una bottiglia in polietilene ad alta densità, vengono discusse le potenzialità di utilizzo dell'indicatore di inventario *non renewable fossil Cumulative Energy Demand* come indicatore *proxy* allo scopo di individuare le situazioni in cui le aziende possono beneficiare dell'utilizzo di tale indicatore prima di avviare uno studio completo di LCA.

Il **Capitolo 6** discute l'uso di modelli parametrici di inventario a supporto della fase di progettazione di nuovi prodotti nel settore degli imballaggi terziari in legno attraverso la definizione di correlazioni tra i principali parametri costitutivi e gli impatti ambientali. Viene sviluppato un modello di inventario che consente di descrivere il ciclo di vita di una serie di pallet in legno con caratteristiche simili e l'efficacia del modello viene testata su un prodotto di riferimento.

Infine, le conclusioni e le prospettive future sono esposte nel **Capitolo 7**.

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# Chapter 1

# Introduction

In this Chapter an introduction to Life Cycle Assessment methodology as defined by ISO 14040 standards is provided. The main methodological limits discussed in the recent scientific literature are presented and finally a focus is given to the aspects discussed in the present work: the issue of uncertainty quantification and the use of streamlined techniques.

## 1.1 Introduction to Life Cycle Assessment

Life Cycle Assessment (LCA) consists in the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle (ISO, 2006a). The product's life cycle includes all consecutive and interlinked stages of a product system, from raw material extraction, production, distribution and use phase to final disposal, as shown in Figure 1.1. For each life cycle stage input and output in terms of material, energy, emissions into air, water and soil are collected in the life cycle inventory and serve as a basis to assess the potential environmental impacts.

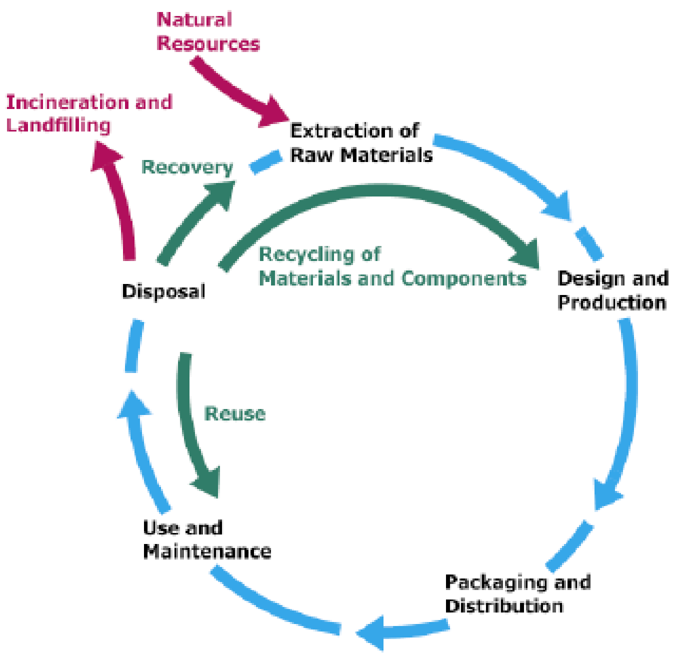


**Figure 1.1** *Schematic representation of a product's life cycle.*

The ISO (International Organization for Standardization) standardized the technical framework for the life-cycle assessment methodology in the 1990s. On this basis, according to ISO 14040 (2006a,b), LCA consists of the following steps (Figure 1.2):

- Goal and scope definition;
- Life Cycle Inventory (LCI);
- Life Cycle Impact Assessment (LCIA);
- Life Cycle Interpretation.

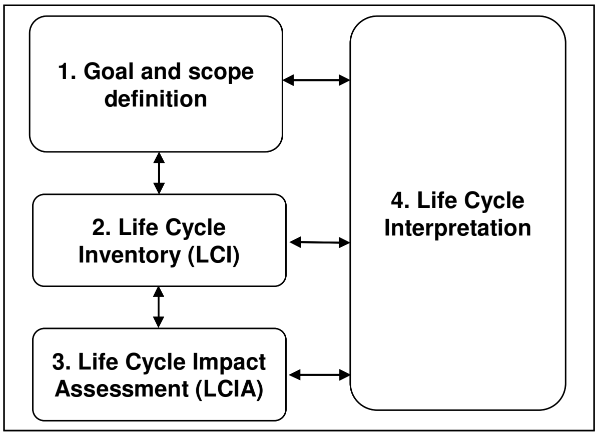


**Figure 1.2** *Definition of the four phases of an LCA study according to ISO 14040:2006.*

LCA is not necessarily carried out in a single sequence. It is an iterative process in which subsequent rounds can achieve increasing levels of detail (from screening LCA to full LCA) or lead to changes in the first phase prompted by the results of the last phase.
The Goal and scope definition includes the reasons for carrying out the study, the intended application, and the intended audience (ISO, 2006a). During this step, the strategic aspects concerning questions to be answered and identifying the intended audience are defined. To carry out the goal and scope of an LCA study, the practitioner must follow some procedures:

1. Define the purpose of the LCA study, ending with the definition of the functional unit, which is the a quantitative measure of the functions that the goods (or service) provide and therefore a quantitative reference for the study.
2. Define the scope of the study, which embraces two main tasks: establish the spatial limits between the product system under study and its neighbourhood that will be generally called "environment" and detail the system through drawing up its unit processes flowchart, taking into account a first estimation of inputs from and outputs to the environment (the elementary flows or burdens to the environment).
3. Define the data required, which includes a specification of the data necessary for the inventory analysis and for the subsequent impact assessment phase. The scope, including product system, functional unit, system boundary and level of detail, of an

LCA depends on the subject and the intended use of the study, therefore they are not defined unambiguously.

The LCI phase is an inventory of input/output data with regard to the system being studied, which involves the collection of the data necessary to meet the goals of the defined study.

All the data of the unit processes within a product system are related to the functional unit of the study, according to the following steps:

1. Data collection, which includes the specification of all input and output flows of the processes within the product system (product flows, i.e., flows to other unit processes, and elementary flows from and to the environment).
2. Normalization to the functional unit, which means that all data collected are quantitatively related to one quantitative output of the product system under study; usually, 1 kg of material is chosen, but the choice of the functional unit is case study-specific.
3. Allocation, which means the distribution of emissions and resource extractions within a given process throughout its different products.

The result from the LCI is a compilation of the inputs (resources) and the outputs (emissions) from the product over its life-cycle in relation to the functional unit.

The LCIA is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of the studied system (ISO, 2006a). The purpose of LCIA is indeed to provide additional information to help assess a product system's LCI results in order to better understand their environmental significance.

The LCIA phase aims at making the results from the inventory analysis more understandable and more manageable in relation to potential environmental impacts. A schematic overview of the impact pathway at the basis of the life cycle impact assessment methodology is presented in Figure 1.3.

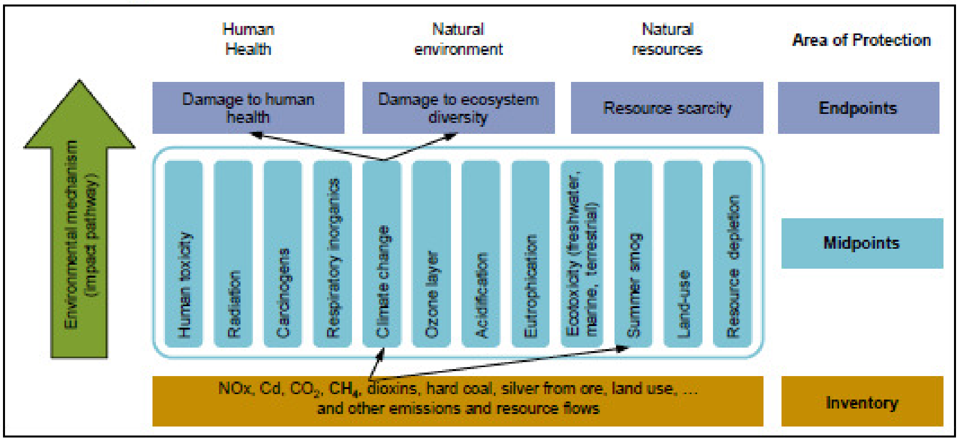


**Figure 1.3** *Schematic representation of the path from Inventory data to midpoint and endpoint impact categories (EC-JRC 2010a).*

The quantification of potential environmental impacts is performed considering human health, the natural environment, and issues related to natural resource use.
Impacts considered in a Life Cycle Impact Assessment include climate change, ozone depletion, eutrophication, acidification, human toxicity (cancer and non-cancer related) respiratory inorganics, ionizing radiation, ecotoxicity, photochemical ozone formation, land use, and resource depletion. The emissions and resources are assigned to each of these impact categories. They are then converted into indicators using impact assessment models with the use of characterization factors, according to Equation 1.1:

$$S_j = \sum_i \left(Q_{ji} \cdot m_i\right) \qquad (1.1)$$

$S_j$ = *score for impact category j (kg reference unit)*
$Q_{ji}$ = *characterization factor that links intervention i to impact category j*
$m_i$ = *intervention of type i (expressed in kg)*
A full list of the impact categories usually addressed in LCA studies can be found in the work provided by the ILCD Handbook (EC-JRC, 2010a).
LCIA step includes mandatory and optional steps. The mandatory steps are:

1. Selection and definition of impact categories, which are classes of a selected number of environmental impacts as reported in Figure 1.3.
2. Classification by assigning the results from the inventory analysis to the relevant impact categories.
3. Characterization by aggregating the inventory results in terms of adequate factors (so-called characterization factors) of different types of substances within the impact categories; therefore a common unit is defined for each category. The results of the characterization step are known as the environmental profile of the product system.

Life cycle interpretation is the final phase of the LCA procedure, in which the results of both LCI and LCIA are summarized and discussed as a basis for conclusions, recommendations and decision-making in accordance with the goal and scope definition (ISO, 2006a).
The following steps can be distinguished within this phase:

1. Identification of the most important results of the LCI and LCIA;
2. Evaluation of the study's outcomes, consisting of a number of the following routines: completeness check, sensitivity analysis, uncertainty analysis and consistency check;
3. Conclusions, recommendations and reports, including a definition of the final outcome, a comparison with the original goal of the study, drawing up recommendations, procedures for a critical review, and the final reporting of the results.

The results of the interpretation may lead to a new iteration round of the study, including a possible adjustment of the original goal.

### *1.1.1 Applications of LCA in the industrial sector*

Among the direct applications of the LCA methodology we can distinguish (ISO 2006a):

- product development and improvement;
- strategic planning;
- public policy making;
- marketing, as LCA is the basis for Environmental Product Declaration.

At the end of the twentieth century, the adoption by industry of the LCA approach was recognized as relatively slow, but the methodology was progressively gaining acceptance.
Some sectors such as plastics, detergents, personal care products and automobiles were identified as pioneers investing in LCA. They were closely followed by agriculture, mining and oil and gas extraction, the construction/building material sector, manufacturing industries and retailing, and more recently by infrastructure industries (electricity, gas and water supply, transport, storage and communication) (Jacquemin et al., 2012).
The increasing importance of LCA methodology within the industrial sector is confirmed by the growing number of publications in the last 10 years; as it can be seen from a search in ISI Web of knowledge with the keywords "LCA" and "Industry", reported in Figure 1.4.

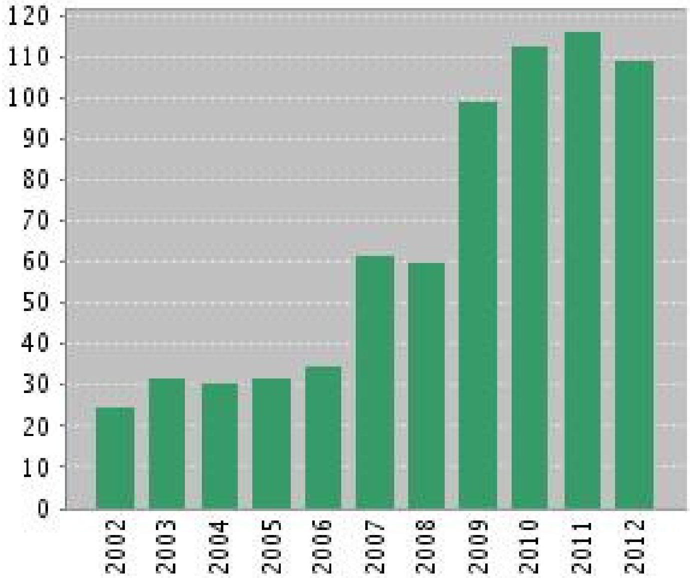


**Figure 1.4** *Search for "LCA" and "industry" in ISI Web of knowledge.*

The strength of LCA is to support many different decisions concerning environmental sustainability in practice in a reliable unambiguous way.
LCA can assist in many applications in the industrial sector which can be grouped into the following categories:

- Product/process improvement: when LCA is used to identify opportunities to improve the environmental performance of products at various points in their life cycle (hot spot analysis);

- Product/process comparison: when the aim is to compare the environmental performance of a product or process with a competing products (benchmarking);
- Product/process design: for the purpose of strategic planning, priority setting, inclusion of environmental aspect in the design process (ecodesign).

From a survey conducted in 2008 within the CALCAS project by the means of interviews with different stakeholder groups it emerged that business actors (industry, retailers) mainly use LCA as a decision support tool in product development (77%), selection of raw materials (66%), and choices of technologies (55%) (Zamagni et al., 2012).

## 1.2 Overview of main methodological limits of LCA

The harmonization process conducted by ISO has increased the maturity and methodological robustness of LCA, but the method is still under development and there are also several ongoing international initiatives to help build consensus and provide recommendations, including:

- the Life Cycle Initiative of the United Nations Environment Program (UNEP) and the Society of Environmental Toxicology and Chemistry;
- the European Platform for LCA of the European Commission (2008b), and the recently published International Reference Life Cycle Data System (ILCD).

LCA is a field of scientific research but is also a business of growing importance, showing different aims. The primary aim of LCA in business is to improve products and processes; this relies upon a foundation of sound and applicable (scientific) methods. Meanwhile, the primary aim of LCA in academia is to improve methods; this relies upon application in case studies as validation for practical implementation (Baitz et a., 2012).

It is recognized that LCA application in practice must fulfill three basic characteristics (Baitz et al., 2012):

(1) reliable, in order to ensure the credibility of information and results generated,
(2) applicable, as it must fit into existing information routines and practices in business, and
(3) quantitative, in order to provide relevant information to help decision makers.

The effective improvement and utilization of LCA in industrial sectors hinge upon identifying current problems that burden LCA (Reap et al. 2008a). Multiple problems occur in each of LCA’s four phases and reduce the accuracy of this tool. Many recent papers have attempted to summarize the main methodological issues connected with LCA methodology (Reap et al., 2008a,b, Finnveden et al., 2009).

A schematic overview of the main methodological problems connected with the four steps of an LCA study is presented in Table 1.1.

**Table 1.1** *LCA problems divided by phase.*

| Phase | Problem |
|---|---|
| Goal and scope definition | Functional unit definition<br>Boundary selection<br>Modelling approach (attributional vs consequential)<br>Social and economic impacts<br>Alternative scenario considerations |
| Life cycle inventory analysis | Allocation<br>Negligible contribution (‘cutoff’) criteria<br>Local technical uniqueness |
| Life cycle impact assessment | Impact category and methodology selection<br>Regionalization of impact assessment<br>Spatial variation<br>Local environmental uniqueness<br>Dynamics of the environment<br>Time horizons |
| Life cycle interpretation | Weighting and valuation<br>Uncertainty in the decision process |
| All | Data availability and quality |

Some of these problems can be seen as constraints for spreading LCA application into business practice, therefore application of LCA and its integration into decision-making process has not been as widespread as expected. Although in principle LCA can inform consumer and policy decisions on environmental grounds, often decision-makers need information on other sustainability dimensions as well. In order to provide such information, it has been argued that there is a need to expand the ISO LCA framework for sustainability assessment by taking into account broader externalities, broader interrelations and different application/user needs with often conflicting requirements (Jeswani et al., 2010).

During the last decades, the scientific community identified three main strategies to overcome these applicative and methodological limitations (De Haes et al., 2004):

1. improve the robustness of the methodology with interventions on its critical aspects;
2. combining LCA with other environmental assessment tools;
3. inclusion of economic and social evaluations towards a Life Cycle Sustainability analysis.

Concerning the first strategy, an important step forward was the publication of the ILCD Handbook (EC-JRC, 2010b) by the Joint Research Center, who launched the ILCD Handbook to develop technical guidance that complements the ISO Standards for LCA and provides the basis for greater consistency and quality of life cycle data, methods, and LCA studies.

As far as the combination with other tools (Jeswani et al., 2010), examples of tools which can be complemented with LCA are procedural methods (assessment frameworks), such as: Environmental Impact Assessment (EIA), Strategic Environmental Assessment (SEA), Multi-Criteria Decision Analysis (MCDA) or analytical methods, such as: Material Flow Analysis (MFA), Substance Flow Analysis (SFA), Energy/Exergy Analysis, Environmental Extended

Input Output Analysis (EIOA)/Hybrid LCA, Risk analysis (RA/ERA/HERA), Cost–Benefit Analysis (CBA), Eco-Efficiency (EE).

Finally, the last strategy is based on the inclusion of economic (Life Cycle Costing) and social aspects (Social Life Cycle Assessment) in order to broaden the scope of LCA towards a Life Cycle Sustainability Assessment (Klöppfer W., 2008).

When analyzing the main constraints to the application of LCA methodology within the industrial sectors, two main research thrusts can be identified: one devoted to increase model fidelity, the other to increase the practicability of LCA (Zamagni et al, 2012).

The former is aimed at increasing the reliability of LCA results, with interventions on exploiting the influence of data variability on the final results. This aspect affects mainly the Life Cycle Interpretation step, when conclusions and recommendations generate from the analysis of the outcomes of the study.

The second aspect focuses on making knowledge available in an easily usable way, and is connected with the need to increase the applicability of LCA methodology. One way of increasing applicability is working on data availability. Especially for small- and medium-size enterprises that rarely have the knowledge and resources necessary to implement LCA a priority is to facilitate the access to reliable, accurate, and relevant life cycle information, reducing efforts and time connected both with data acquisition and with the creation of the modelling.

## 1.3 Uncertainty quantification

When LCA is used for decision support, uncertainty is an important issue to be taken into consideration (Huijbregts et al. 2001; Geisler et al. 2005; Lloyd and Ries, 2007). Uncertainty is lack of knowledge about the true value of a quantity, true form of a model, appropriateness of a modelling, or methodological decision.

LCA practitioners generally assign single values to model parameters, build deterministic models to approximate environmental outcomes, and report results as point estimates. This approach fails to capture the variability and uncertainty inherent in LCA, reducing the effectiveness of LCA for supporting private and public sector decision making (Lloyd and Ries, 2007).

Whether the desired outcome of an LCA is a simple benchmark or a more involved recommendation of action, its reliability depends on appropriate consideration of uncertainty. The existence of uncertainty connected with data used for LCA studies can hamper a clear interpretation of LCA results. Providing results with uncertainty information allows assessing the stability of the result, and in some cases, a ranking order may be changed by considering the underlying uncertainty (Ciroth A., 2004). Even though it is widely recognized that there is a need to quantify uncertainty in LCA, such as in other decision support tools, the role of uncertainty analysis in LCA studies is still neglected. ISO standards (ISO 2006a, b) provide little guidance on how to practically deal with uncertainty quantification, however this issue is analyzed by some complementary initiative (EC-JRC, 2010b).

The ISO LCA series of standards briefly mentions two techniques:

1. uncertainty analysis, which models uncertainties in the inputs to a LCA and propagates them to results. For comparative LCAs, this can reveal whether there are significant differences between decision alternatives.
2. sensitivity analysis, which studies the effects of arbitrary changes in inputs on LCA outputs. This helps to identify the most influential LCA inputs when their uncertainty has yet to be or cannot be quantified.

In response, LCA researchers and practitioners have proposed or adopted different variations of these techniques (Björklund A.E., 2002; Lloyd and Ries 2007). Choosing one can be difficult, especially for predictive assessments, comparative assessments of complex systems, or assessments with broad scope. Even when an appropriate choice is apparent, in practice, there are still many hurdles to using them. An example of how to handle data quality with application to a specific case study at industrial level is presented by Guo and Murphy (2012).

Another technique, which can be identified as sensitivity analysis is contribution analysis. Contribution analysis consists in decomposing the LCA result (characterised, normalised or weighted impact) of a system into its individual process contributions, providing a quick overview of the important contributors. Processes that have both positive and negative impacts have to be subdivided into their sub-components, to avoid neglecting important processes.

Summarizing, in LCA three main types of uncertainty can be distinguished (Bjorklund A.E., 2002):

- parameter uncertainty, connected with data uncertainty regarding process inputs, environmental discharges, and technology characteristic, i.e. quality of available data;
- scenario uncertainty, with regard to methodological choices (e.g. functional unit, allocation rules, system boundaries, cut-off rules, impact assessment methods);
- model uncertainty, connected with the adoption of linear models to describe the relationships among environmental phenomena and models for deriving emissions and characterization factors.

Uncertainty can be dealt with in several ways, both in LCI and LCIA phase, namely the ‘‘scientific’’ way, the ‘‘social way’’, and the ‘‘statistical way’’ (Finnveden et al. 2009).

The ‘‘statistical’’ way, in contrast to the previous two ways, does not try to remove or reduce the uncertainty, but to incorporate it.

Statistical theory comprises a large body of methods:

- parameter variation and scenario analysis: these involve calculating a result with a number of different data values and/or choices, e.g., using the maximum and the minimum fuel efficiency, and seeing if the results are stable;
- classical statistical theory on the basis of probability distributions, tests of hypothesis, etc.;
- Monte Carlo simulations, bootstrapping, and other sampling approaches;
- using analytical methods, based on first-order error propagation;
- using less conventional methods, such as non-parametric statistics, Bayesian analysis, and fuzzy set theory;
- using qualitative uncertainty methods, for instance, based on data quality indicators.

From a survey on quantitative approaches for characterizing, propagating, and analyzing uncertainty in LCA (Lloyd and Ries 2007) some highlights emerged, namely:

- parameter uncertainty is the most analyzed (100%);
- uncertainty is mainly reported at LCI level (71%);
- stochastic modeling is the main method used for estimating uncertainty propagation (67%);
- qualitative assessment of data variability is important to characterize uncertainty (42%).

## 1.4 Streamlined techniques

One of the main applicative constraints to LCA application into industrial practice is given by data collection costs, which can be prohibitively large, e.g., when data must be gathered from the field or when data must be frequently collected to remain relevant (Maurice et al. 2000). In other cases, data exists outside of the LCA practitioner's organization, e.g., when withheld upstream or downstream by suppliers or other partners who have concerns (potentially valid) that sharing inventory data might reveal confidential information related to their competitive advantage (Ayres R.U., 1995).

Moreover, for practical decision-making in early phases, there is a demand for less complicated, thus more widely utilizable, tools in situations in which preliminary analyses need to be made or in which less-than-perfect results can still be considered better than no results at all (Pesonen and Horn, 2012).

There is indeed a need for streamlined (or "simplified") approaches to quicken the lengthy and resource-consuming assessment process, as a full-scale LCA can be both time and resource intensive, which leads to the outcome that they are not always the primary or best action of a company trying to develop its processes or products towards a more sustainable direction. In fact, the inherent complexity of carrying out a full LCA can be hypothesized as standing in the way of a widespread application in the industry and policy-making sectors (Bala et al. 2010). Furthermore, the results of a full LCA can be very complex and difficult to

understand for decision-makers either in the industry or in the public sector.
A streamlined approach is given by screening LCA, which uses mainly quantitative data; however, it is available from readymade databases so that no new inventory calculations are made. In general, streamlined life cycle approaches can be qualitative, quantitative, or semiquantitative. A large number of simplified LCA methods have been developed (Baumann and Tillman 2004, Pesonen and Horn, 2012).
Another way for streamlining data collection is given by the use of parameterization, which refers to the practice of presenting LCA data using raw data and formulas instead of computed numbers in unit process datasets within databases. Parameterization is a powerful way to ensure transparency, usability, and transferability of LCI data. (Cooper et al., 2012).
As LCA is in most cases too complex to be integrated as a regular constituent into product development. This especially holds true in the early phases of product development, when the “rough” estimations of the product's impact on the environment are needed in order to take advantage of the significant (and relatively cheap) improvement opportunities characteristic to early design. Parametric estimation techniques - known from production and/or Life Cycle Cost estimation practice - have been proposed as a simplified LCA technique for estimating the environmental impacts of similar technical products based on a limited number of LCA studies. The aim of these environmental parametric estimation techniques is to establish a coupling between functional requirements (FR) or design parameters (DP) that product developers have at hand in early design phases and the environmental impact (EI) of the product (Dick et al., 2004).

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# Chapter 2

# Comparative LCA of barley and soybean: organic vs conventional farming

This chapter[1] presents a case study of comparative LCA in the agrifood sector, namely a comparison between conventional and organic farming in a three year cycle including soybean in the first and third year and barley in the second year. The influence of the geoclimatic parameter rainfall index on the LCIA results is discussed by the means of a sensitivity analysis. A four step procedure for parameter uncertainty is developed and applied to the case study. The results of both sensitivity and uncertainty analysis allowed to broaden the validity and reliability of the LCA study.

## 2.1 Introduction

The number of applications of LCA to agriculture has increased recently, pursuing comparative assessment of agricultural production systems and identification of environmental hot spots and the improvement potential of agricultural practices (Hayashi K., 2012).

Agricultural systems are sufficiently different from industrial systems that this area of application introduces new methodological issues for all phases of LCA (Milà i Canals et al, 2006). The main difference between agricultural and industrial studies is the inclusion of output data deriving from natural processes from farming practices (in particular fertilisation and pesticide treatments). Many studies have already demonstrated the potential of LCA as a decision-supporting tool to evaluate different industrial food products, dairy and meat production and agricultural products and highlight environmental hot spots (Roy et al., 2009). The applicability of LCA for agricultural and food products is however restricted by certain limitations, some of which are characteristics inherent to LCA methodology as such, and

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[1] *Portions of this Chapter have been published in Niero et al. (2012) and submitted for publication in Fedele et al. (2012).*

some of which are relevant specifically in the field of agri-food, i.e. the calculation of emissions from field (Audsley et al., 1997).
Given the increasing importance of reaching a sustainable agriculture management, in the last years there has been a growing interest to understand the relative environmental impacts of organic and conventional farming methods. At European level the rules for organic farming are defined by the Council Regulation (EC) No 834/2007 of 28 June 2007 (EC, 2007) which establishes the essential elements of the organic plant production, namely soil fertility management, choice of species and varieties, multiannual crop rotation, recycling organic materials and cultivation.
Some common elements emerged from LCA studies comparing conventional and organic farming: while organic farming improves soil quality and biodiversity (Mader et al. 2002) and reduces the use of pesticides (Roy et al., 2009), conventional farming has yields typically higher than organic farming (Meisterling et al., 2009, Salomone and Ioppolo, 2012) and consequentially it requires lower land occupation than organic production (van der Werf et al., 2009, Williams et al., 2010). While the lower yields push the energy use and emissions higher per product unit in organic production, the avoided production and use of synthetic inputs act as a countervailing force. The final outcome often depends on the characteristics of the specific production systems being compared (Seufert et al., 2012). Particular attention for environmental impact assessment studies in agricultural sector should indeed be paid to the definition of specific farming location features such as climate, soil properties, cropping management, as they are all important factors which may have significant influence on the environmental performance of products (Kim et al., 2009). Moreover, when comparing different production systems, the focus is usually on a particular crop, but management practices include crop rotations, therefore there is an interest in the assessment of a multiannual cycle.
Within this sector, it is particular relevant to discuss the results from comparative studies including the effect of local conditions, as well as including the effect of data variability on the final outcomes. Uncertainty is rarely considered in LCA studies and even less in LCA of agricultural products (Basset et al., 2006). Some examples of applications refer to citrus-based products (Beccali et al, 2010), beverage sectors (Mattila et al., 2012) and table potatoes (Röös et al., 2010). The greatest level of uncertainty is widely acknowledged as on-field emissions and, in particular, the $N_2O$ emissions, emission factors and the resultant contribution to climate change, but it is important to understand which is the contribution of uncertainty over the different impact categories considered in a study.
The main goals of this case study are two: (i) to compare the potential environmental impacts of the production of soybean and barley according with conventional and organic farming systems within a three-year cycle, and (ii) to verify the reliability of LCA results through the application of a sensitivity analysis and combined qualitative and stochastic quantitative

method for uncertainty analysis.

## 2.2 Materials and methods

A comparative LCA of soybean and barley production with conventional or organic farming was performed using data from a three-year cycle provided by a farm in the Po Valley area, which can be considered (both for barley and soybean seeds production) representative of a generic cultivation of North-eastern Italy. A three-year cycle was considered, as multiannual crop rotation in accordance with Art. 12 of the Council Regulation dealing with plant production rules (EC, 2007) can increase soil quality and reduce nitrogen fertilizer requirements in crop production (Meisterling et al., 2009). Therefore, multiannual crop rotation is mandatory when dealing with organic farming, but it is a common practice also within conventional farming. Moreover, crop yields are very sensitive to geo-climatic parameters, both for organic farming and conventional farming.

It is recognized that in agricultural LCA studies, site-specific data are necessary for the description of operations on the field (Kim et al., 2009; Meisterling et al., 2009; Mourad et al., 2007). For example, differences in fertiliser use can generally depend on locations and these differences are largely due to soil condition and, also, the number of pesticide applications is affect by grower-, region-, and variety-dependent factors (Milà i Canals et al., 2006). Therefore, the results provided by this single case study are valid within the scope of the context from which they are derived.

The methodology for the environmental impacts quantification used is in accordance with the international standards of series ISO 14040 (ISO, 2006a,b).

### *2.2.1 Goal and scope definition*

The aim of the LCA study was to conduct a three-year comparative LCA of agricultural production systems (conventional versus organic) and to identify the variability in environmental impacts according to the geo-climatic factor spring rainfall index. The focus of the LCA study was on the agricultural stage and the product system includes all agricultural processes required for the production of soybean and barley and by auxiliary processes such as transport of seeds and fertilizer and the maintenance of farm vehicles, as suggested by Milà i Canals et al. (2006).

A mass-based functional unit is adequate when analysing only the agricultural stages of the life cycle of the agricultural product (Audsley et al., 1997; Mila i Canals et al., 2006). In accordance with the scope of the study the functional unit was 1 kg of seeds, composed respectively by two parts of soybean from first and third year of the three-year cycle and one third of barley from second production year.

As the focus of the study was crops production rather than product transformation, storage or consumption, the system boundaries included all the life cycle stages from cradle to the farm gate, excluding the distribution, processing and consumption of products. Similarly to other studies (Milà i Canals et al., 2006; Zimmermann et al., 2010), only the preliminary agricultural step of the whole life cycle of an agricultural product is considered. Therefore the system boundary includes soybean and barley production processes from ploughing to harvesting and the manufacturing processes of fertilizers, herbicides, pesticides, compost and fuels. The system boundaries of the analyzed system are reported in Figures 2.1 and 2.2.

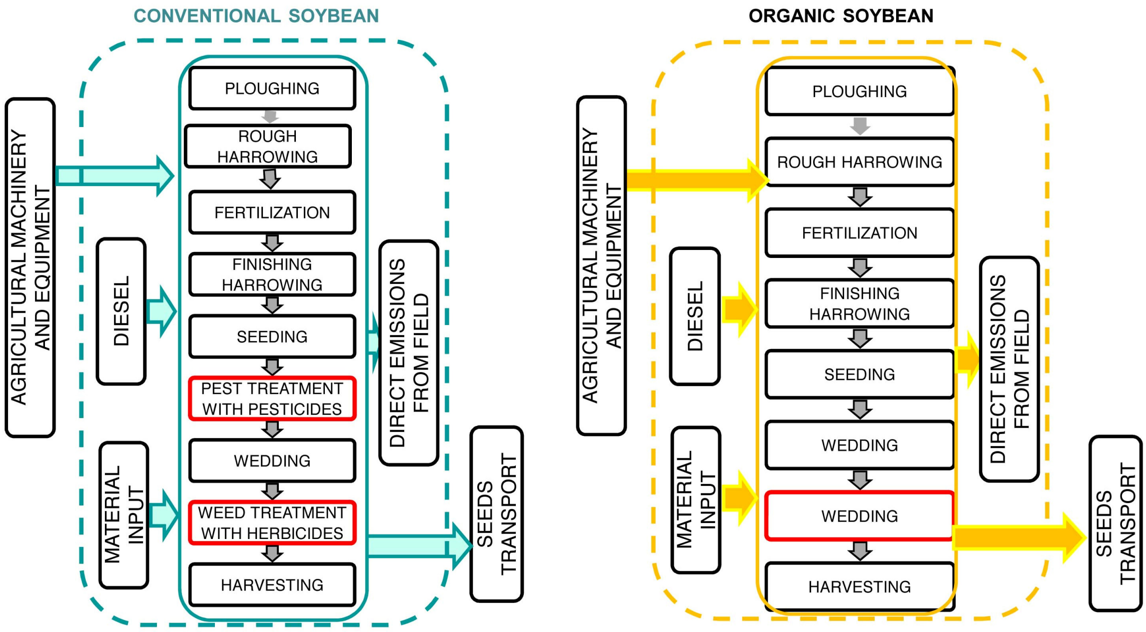


**Figure 2.1** *System boundaries of the conventional soybean farming (on the left) and organic soybean (on the right)*

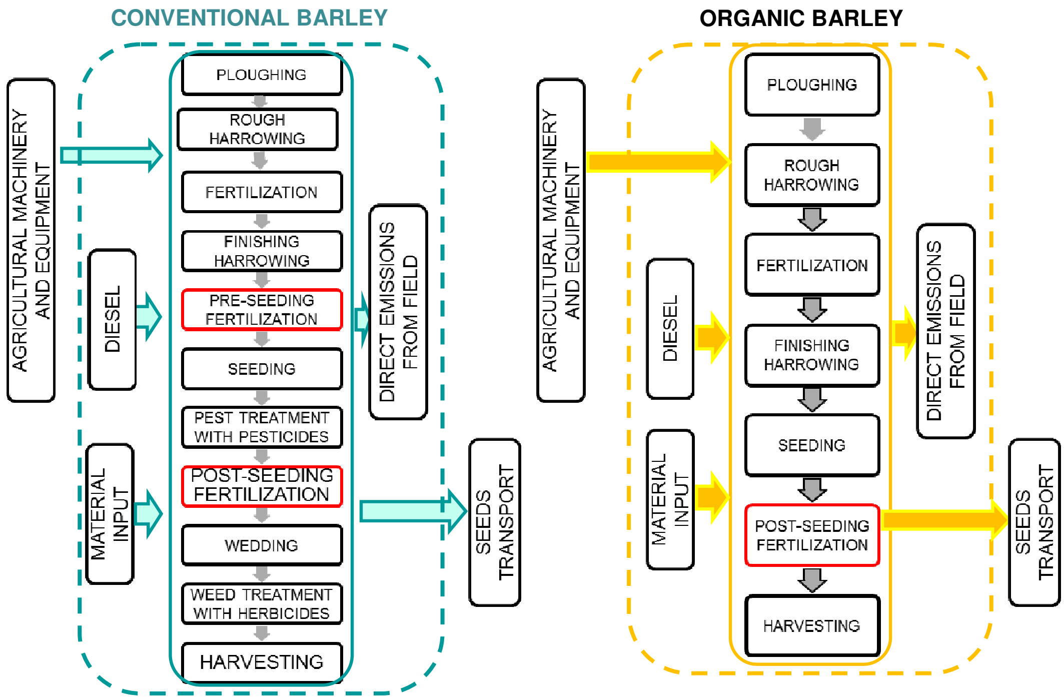


**Figure 2.2** *System boundaries of the conventional barley (on the left) and organic barley (on the right)*

Even if some studies suggest that machinery production is relevant in assessing the environmental impacts of agriculture (Milà i Canals, 2006), in this case agricultural machinery and equipments (i.e. tractors, plough, harvester), were not taken into account, as they are independent from the farming technique.
The cut-off criteria for initial inclusion of inputs and outputs was based on 1% mass basis, but this choice would have led to exclude weed killers and pesticide. However, due to their environmental significance they were included, even if their value was below the cut-off. Therefore a criteria base on environmental relevance was added. The exclusions based on mass cut-off were inherent to plastic and paper wastes and lubricating oil, grease and filters for the maintenance of tractors: a sensitivity analysis was performed to evaluate changes on the final results considering these exclusions and no significant differences emerged.

## *2.2.2 Life Cycle Inventory*

In order to define the agricultural steps that are part of each crop cycle, a specific questionnaire was elaborated and fulfilled by personal interview with farmers. The questionnaire included questions about all material and energy inputs like fertilizers, pesticides, water usage, diesel, electric energy and was based on the model proposed by Mourad et al. (2007), considering a 12-month seasonal period of agriculture activities.
The agricultural steps evaluated in the analysis are:

- *Ploughing*: a mechanical step for the preliminary treatment of the soil; it is the same for all crops under study;
- *Harrowing:* operation of breaking clods and levelling the surface (known as roughing harrowing), which can be performed after fertilization (finishing harrowing) for burrowing fertilizers;
- *Fertilization*: with chemicals or organic fertilizers, according to the farming techniques;
- *Seeding:* with conventional or organic seeds;
- *Pest treatment* (with pesticides): operation made only in conventional agriculture (specific for each crop), which allows to eliminate pests from plant;
- *Weed treatment* (with chemical herbicides): operation made only in conventional agriculture, which permits to kill weeds;
- *Weeding*: mechanical operation carried out after the seeding, once the plant is germinated in order to aerate the soil, to kill weeds, to favour the penetration of solar heat and a reduction of water evaporation.
- *Harvesting:* final step which allows to obtain the final output, it is basically the same operation for every crop.

For the LCI step mainly primary data were collected from field investigation (reference year 2009). The main primary data for each crop are listed in Table 2.1.

**Table 2.1**. *Input and output primary data referred to 1 ha of cultivated area.*

| | **Conventional soybean** | **Conventional barley** | **Organic soybean** | **Organic barley** |
|---|---|---|---|---|
| | *Input* | | | |
| Diesel[1] (l) | 103 | 103 | 116 | 96 |
| Triple superphosphate (kg) | 300 | 450 | / | / |
| Urea (kg) | / | 200 | / | / |
| Compost (kg) | / | / | 150 | 150 |
| Seeds (kg) | 120 | 180 | 120 | 180 |
| Erbicide (kg) | 2 | 1,5 | / | / |
| Water (l) | 600 | 800 | / | / |
| Pesticide (kg) | 1 | 2 | / | / |
| | *Output* | | | |
| Soybean stalks (kg) | 5000 | / | 3500 | / |
| Barley straw (kg) | / | 7000 | / | 5500 |
| yields (kg) | 5000 | 7000 | 3500 | 5500 |

[1] total amount for the annual cultivation

Secondary data were taken from international databases in order to estimate data that it was not possible to collect on the field, mainly from the Ecoinvent database. Processes included in this database, developed by the Swiss Centre for Life Cycle Inventory, are mainly representative of Switzerland and Europe situation, therefore they can be considered appropriate for the case study.

Some assumptions were made for the description of the fertilizers and pesticides supply: a distance of 20 km from the farmers' cooperative to the farmland with a trailer truck (28 tons) was hypothesized. Carbon enrichment in the soil was considered irrelevant for the period of time considered, which is lower than 20 years (IPCC, 2006). No net change in carbon content of the soil from year to year was assumed; however changes in soil composition due to the agriculture activities were included in the impacts assessment through the evaluation of heavy metals in soil.

### *2.2.3 Emissions from field*

One critical aspect in LCA applied to the agricultural sector is the quantification of emissions from field (Audsley et al., 1997; Williams et al., 2006) in every environmental compartment (air, water and soil). They were evaluated with specific models (IPCC, 2006, Nemecek and Kaegi, 2007) starting from the primary data provided by the farmers.

As far as emissions into air are concerned, we included the emissions from Nitrogen compounds: $N_2O$, $NO_X$ and $NH_3$ (IPCC, 2006).

Nitrous oxide is produced naturally in soils through the processes of nitrification and denitrification. Nitrification is the aerobic microbial oxidation of ammonium to nitrate, and denitrification is the anaerobic microbial reduction of nitrate to nitrogen gas ($N_2$). Nitrous oxide is a gaseous intermediate in the reaction sequence of denitrification and a by-product of nitrification that leaks from microbial cells into the soil and ultimately into the atmosphere. One of the main controlling factors in this reaction is the availability of inorganic N in the soil.

IPCC methodology estimates $N_2O$ emissions using human-induced net N additions to soils (e.g., synthetic or organic fertilisers, deposited manure, crop residues, sewage sludge), or of mineralisation of N in soil organic matter following drainage/management of organic soils, or cultivation/land-use change on mineral soils (e.g., Forest Land/Grassland/Settlements converted to Cropland).

The emissions of $N_2O$ that result from anthropogenic N inputs or N mineralisation occur through both a direct pathway (i.e., directly from the soils to which the N is added/released), and through two indirect pathways:

- following volatilisation of $NH_3$ and $NO_x$ from managed soils and from fossil fuel combustion and biomass burning, and the subsequent redeposition of these gases and their products $NH_4^+$ and $NO_3^-$ to soils and waters;
- after leaching and runoff of N, mainly as $NO_3^-$, from managed soils.

Direct emissions of $N_2O$ from managed soils are estimated separately from indirect emissions, though using a common set of activity data. The Tier 1 methodologies was used (IPCC, 2006), which do not take into account different land cover, soil type, climatic conditions or management practices. The following equation provides the basis for the calculation of anthropogenic emissions of direct $N_2O$ (Eq. 2.1) from managed soils:

$$N_2O_{Direct}\text{–}N = N_2O\text{–}N_{N\,inputs} + N_2O\text{–}N_{OS} + N_2O\text{–}N_{PRP} \tag{2.1}$$

Where:

$$N_2O\text{–}N_{N\,inputs} = \begin{bmatrix} \left[\left(F_{SN} + F_{ON} + F_{CR} + F_{SOM}\right) \bullet EF_1\right] + \\ \left[\left(F_{SN} + F_{ON} + F_{CR} + F_{SOM}\right)_{FR} \bullet EF_{1FR}\right] \end{bmatrix}$$

$$N_2O\text{–}N_{OS} = \begin{bmatrix} \left(F_{OS,CG,Temp} \bullet EF_{2CG,Temp}\right) + \left(F_{OS,CG,Trop} \bullet EF_{2CG,Trop}\right) + \\ \left(F_{OS,F,Temp,NR} \bullet EF_{2F,Temp,NR}\right) + \left(F_{OS,F,Temp,NP} \bullet EF_{2F,Temp,NP}\right) + \\ \left(F_{OS,F,Trop} \bullet EF_{2F,Trop}\right) \end{bmatrix}$$

$$N_2O\text{–}N_{PRP} = \left[\left(F_{PRP,CPP} \bullet EF_{3PRP,CPP}\right) + \left(F_{PRP,SO} \bullet EF_{3PRP,SO}\right)\right]$$

where:

$N_2O_{Direct}$-$N$ = annual direct $N_2O$-N emissions produced from managed soils, *kg* $N_2O$-$N\ yr^{-1}$

$N_2O$-$N_{N\ inputs}$ = annual direct $N_2O$–N emissions from N inputs to managed soils, *kg* $N_2O$-$N\ yr^{-1}$

$N_2O$-$N_{OS}$ = annual direct $N_2O$–N emissions from managed organic soils, *kg* $N_2O$-$N\ yr^{-1}$

$N_2O$-$N_{PRP}$ = annual direct $N_2O$–N emissions from urine and dung inputs to grazed soils, *kg* $N_2O$-$N\ yr^{-1}$

$F_{SN}$ = annual amount of synthetic fertiliser N applied to soils, *kg* $N\ yr^{-1}$

$F_{ON}$ = annual amount of animal manure, compost, sewage sludge and other organic N additions applied to soils, *kg* $N\ yr^{-1}$

$F_{CR}$ = annual amount of N in crop residues (above-ground and below-ground), including N-fixing crops, and from forage/pasture renewal, returned to soils, *kg* $N\ yr^{-1}$

$F_{SOM}$ = annual amount of N in mineral soils that is mineralised, in association with loss of soil C from soil organic matter as a result of changes to land use or management, *kg* $N\ yr^{-1}$

$EF_1$ = emission factor for $N_2O$ emissions from N inputs, *kg* $N_2O$-$N(kg\ N\ input)^{-1}$

$EF_{1FR}$ = emission factor for $N_2O$ emissions from N inputs to flooded rice (default value), *kg* $N_2O$-$N\ ha^{-1}\ yr^{-1}$

$F_{OS}$ = annual amount of organic drained/managed soil, ha. [2]

$EF_2$ = emission factor for $N_2O$ emissions from drained/managed organic soils (default), *kg* $N_2O$-$N\ (kg\ N\ input)^{-1}$, [3]

$F_{PRP}$ = annual amount of urine and dung N deposited by grazing animals on pasture, range and paddock, *kg* $N\ yr^{-1}$.

$EF_{3PRP}$ = emission factor for $N_2O$ emissions from urine and dung N deposited on pasture, range and paddock by grazing animals (default) *kg* $N_2O$-$N\ (kg\ N\ input)^{-1}$ , [2]

In addition to the direct emissions of $N_2O$ from managed soils that occur through a direct pathway (i.e., directly from the soils to which N is applied), emissions of $N_2O$ also take place through two indirect pathways:

- the volatilisation of N as $NH_3$ and oxides of N ($NO_X$), and the deposition of these gases and their products $NH_4^+$ and $NO_3^-$ onto soils and the surface of lakes and other waters;
- the leaching and runoff from land of N from synthetic ($F_{SN}$) and organic fertilizer additions ($F_{ON}$), crop residues ($F_{CR}$), mineralization of N associated with loss of soil C in mineral and drained/managed organic soils through land-use change or management practices ($F_{SOM}$), and urine and dung deposition from grazing animals ($F_{PRP}$).

[2] Pedices CG, F, Temp, Trop, NR, NP refer respectively to Cropland and Grassland, Forest Land, Temperate, Tropical, Nutrient Rich, Nutrient Poor

[3] Suffixes CPP and SO refer respectively to Cattle, Poultry, Pig and to Sheep and Other animals

Conversion of $N_2O$–N emissions to $N_2O$ emissions is performed by using the following equation:

$$N_2O = N_2O\text{–}N \bullet 44/28 \quad (2.2)$$

In Tables 2.2 and 2.3 are reported the synthesis of the calculation of respectively direct and indirect annual $N_2O$ emissions for the four systems considered in the study.

**Table 2.2**. *Calculation of direct $N_2O$ according to the IPCC model (IPCC, 2006)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| $F_{SN}$ (kgN/yr)= | 0 | 0 | 92 | 0 |
| $F_{ON}$ (kgN/yr)= | 0 | 16.5 | 0 | 22 |
| $F_{CR}$ (kgN/yr)= | 33.86 | 23.71 | 42.75 | 33.59 |
| crop (kg d.m./ha) = | 4550 | 3185 | 6230 | 4895 |
| harvested area (ha) = | 1 | 1 | 1 | 1 |
| area burnt (ha) = | 0 | 0 | 0 | 0 |
| $C_f$ (-)= | 0 | 0 | 0 | 0 |
| $Frac_{Renew}$ (-)= | 1 | 1 | 1 | 1 |
| $R_{AG}$ (kg d.m/kg d.m.)= | 930.30 | 930.42 | 980.09 | 980.12 |
| $AG_{DM}$ (Mg/ha)= | 4232.85 | 2963.40 | 6105.99 | 4797.69 |
| $N_{AG}$ (kgN/kg d.m.)= | 0.008 | 0.008 | 0.007 | 0.007 |
| $Frac_{Removed}$ (-) = | 0 | 0 | 0 | 0 |
| $R_{BG}$ (kg d.m/kg d.m.)= | 176.95 | 176.97 | 215.84 | 215.85 |
| $R_{BG\text{-}bio}$(kg d.m/kg d.m.)= | 0.19 | 0.19 | 0.22 | 0.22 |
| $N_{BG}$ (kgN/kg d.m.)= | 0.008 | 0.008 | 0.014 | 0.014 |
| $F_{SOM}$ (kgN/yr)= | 0 | 0 | 0 | 0 |
| $EF_1$ (-) = | 0.01 | 0.01 | 0.01 | 0.01 |
| **$N_2O$-N $_{N\ inputs}$ (kgN2O-N/yr)** = | 0.34 | 0.40 | 1.35 | 0.56 |
| $F_{OS,CG,Temp}$ (ha) = | 1 | 1 | 1 | 1 |
| $EF_{2CG,Temp}$ (kgN0-N/(ha*yr)) = | 8 | 8 | 8 | 8 |
| **$N_2O$-$N_{OS}$ (kgN2O-N/yr) =** | 8 | 8 | 8 | 8 |
| **$N_2O$-$N_{PRP}$(kgN2O-N/yr) =** | 0 | 0 | 0 | 0 |
| **$N_2O$direct-N (kgN2O-N/yr)** = | 8.339 | 8.402 | 9.347 | 8.556 |
| $N_2O$ direct (kg/yr) = | 13.104 | 13.203 | 14.689 | 13.445 |

**Table 2.3**. *Calculation of indirect $N_2O$ according to the IPCC model (IPCC, 2006)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| **$N_2O$ from atmospheric deposition of N volatilised from managed soils** | | | | |
| $F_{SN}$ (kgN/yr)= | 0 | 0 | 92 | 0 |
| $Frac_{Gasf}$ (kgN applied)^-1= | 0.1 | 0.1 | 0.1 | 0.1 |
| $F_{ON}$ (kgN/yr)= | 0 | 16.5 | 0 | 22 |
| $F_{PRP}$ (kgN/yr)= | 0 | 0 | 0 | 0 |
| $Frac_{Gasm}$ (kgN applied)^-1= | 0.2 | 0.2 | 0.2 | 0.2 |
| $EF_4$ (-)= | 0.01 | 0.01 | 0.01 | 0.01 |
| ***$N_2OATD$-N* (kgN2O-N/yr) =** | 0 | 0.033 | 0.092 | 0.044 |
| **$N_2O$ ATD (kg/yr)** | 0,000 | 0.052 | 0.145 | 0.069 |
| **$N_2O$ from n leaching/runoff from managed soils in regions where leaching/runoff occurs** | | | | |
| $F_{SN}$ (kgN/yr)= | 0 | 0 | 92 | 0 |
| $F_{ON}$ (kgN/yr)= | 0 | 16.500 | 0 | 22 |
| $F_{PRP}$ (kgN/yr)= | 0 | 0 | 0 | 0 |
| $F_{CR}$ (kgN/yr)= | 33.864 | 23.709 | 42.745 | 33.587 |
| $F_{SOM}$ (kgN/yr)= | 0 | 0 | 0 | 0 |
| $Frac_{Leach-(H)}$(kgN additions/yr)= | 0.3 | 0.3 | 0.3 | 0.3 |
| $EF_5$ (-)= | 0.0075 | 0.008 | 0.0075 | 0.0075 |
| ***$N_2O(L)$ - N =*** | 0.0762 | 0.090 | 0.3032 | 0.1251 |
| **$N_2O$ (L) (kg/yr) =** | 0.120 | 0.142 | 0.476 | 0.197 |
| **Total $N_2O$ Indirect (kg/yr)** | **0.120** | **0.194** | **0.621** | **0.266** |

In the case of conventional barley, also $CO_2$ emissions into air from urea fertilization were calculated, considering the annual amount of urea used (200 kg/year) multiplied by the emission factor (0.2 C/ton urea), giving as a result 0.04 ton $CO_2$-C/year. Finally this value was transformed in terms of total $CO_2$ emissions (0.147 ton/year).

For the calculation of $NO_X$ emissions the model proposed by the Ecoinvent database was used (Nemecek et al., 2007), according to equation 2.3:

$$NO_X = 0.21 \cdot N_2O \tag{2.3}$$

The values of $NO_X$ emissions are reported in Table 2.4.

**Table 2.4**. *Calculation of $N_X0$ according to the Ecoinvent model (Nemecek and Kaegi, 2007)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| **$N_XO$ (kg/kg crop)** | 5.554E-04 | 8.038E-04 | 4.593E-04 | 5.235E-04 |

Regarding $NH_3$ emissions, the model suggested by Ecoinvent database (Nemecek et al., 2007) was used, considering only emissions coming from mineral compost. In the case of conventional soybean only phosphates-based compost are used, therefore no ammonia emissions are present in this case.

**Table 2.5**. *Calculation of $NH_3$ emission according to the Ecoinvent model (Nemecek and Kaegi, 2007)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| Urea [kg] = | / | / | 200 | / |
| % N in urea = | / | / | 46% | / |
| kg N in urea = | / | / | 92 | / |
| Compost [kg] = | / | 150 | / | 350 |
| % N in compost = | / | 11% | / | 11% |
| kg N in compost = | / | 16.5 | / | 38.5 |
| Emission factor for NH3-N= | 0 | 0.66 | 13.8 | 1.54 |
| $NH_3$ [kg] = | 0 | 0.801 | 16.757 | 1.870 |
| Yield to kg conversion factor | 0 | 0.000286 | 0.000143 | 0.000182 |
| kg $NH_3$/kg crop = | 0 | 0.00023 | 0.00240 | 0.00034 |

Concerning emissions into water, nitrate and phosphate emissions were included, according to the model proposed by the Ecoinvent database (Nemecek et al., 2007), but it was possible to include only the contribution from:

- nitrate leaching in groundwater (which depends on Nitrogen mineralization from organic matter into soil according to different months, nitrogen absorption from vegetation, N input from fertilizers and soil depth);,
- phosphorus emissions into groundwater (which depends on type of cultivated soil, therefore it is the same value for all crops);
- phosphate emissions into ground water and surface water.

The values considered in the calculation are reported in Tables 2.6, 2.7 and 2.8.

**Table 2.6**. *Nitrate emissions into groundwater according to Ecoinvent model (Nemecek and Kaegi, 2007)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| Harvesting month | September | September | June | June |
| Nmin [kg N per ha] | 125 | 75 | 75 | 40 |
| Clay content | > 40% | > 40% | > 40% | > 40% |
| Humus content | 8-15% | 8-15% | 8-15% | 8-15% |
| Corrective factor [-] | -15 | -15 | -15 | -15 |
| Value N mineral potential [kg N/ha] | 110 | 60 | 60 | 25 |
| Nupt [kg N per ha] | 10 | 10 | 75 | 75 |
| Risk of nitrogen leaching | 0 | 13.2 | 9.2 | 17.05 |
| Notes | *No N fertilizer* | *150kg/ha organic compost (11%N)* | *200kg/ha urea (46%N)* | *Organic compost (150 kg/ha + 200kg/ha)* |
| Nitrogen leaching to ground water [kg N/ha] | 120 | 83.2 | 144.2 | 117.05 |
| Corrective factor [-] | 0.8 | 0.8 | 0.8 | 0.8 |
| Final Nitrogen leaching to ground water [kg N/ha] | 96 | 66.56 | 115.36 | 93.64 |
| Nitrate leaching to ground water [kg $NO_3$] | 425.143 | 294.766 | 510.880 | 414.691 |
| Yield to kg conversion factor | 0.0002 | 0.000286 | 0.000143 | 0.000182 |
| ***kg $NO_3$/kg crop*** | 0.0850 | 0.0843 | 0.0731 | 0.0755 |

**Table 2.7**. *Phosphorus leachate to groundwater according to Ecoinvent model (Nemecek and Kaegi, 2007)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| Pgw [kgP/ha]= | 0.07 | 0.07 | 0.07 | 0.07 |
| Pwl [kgP/ha] = | 0.07 | 0.07 | 0.07 | 0.07 |
| $P_2O_5$sl [kg/ha] = | 0 | 0 | 0 | 0 |
| $F_{gw}$ [-] = | 1 | 1 | 1 | 1 |
| phosphate,to groud water [kg$PO_4$/ha] = | 0.21452 | 0.21452 | 0.21452 | 0.21452 |
| Yield to kg conversion factor | 0.0002 | 0.000286 | 0.000143 | 0.000182 |
| ***kg $PO_4$/kg crop*** | 4.290E-05 | 6.135E-05 | 3.068E-05 | 3.904E-05 |

**Table 2.8**. *P runoff to surface waters according to the Ecoinvent model (Nemecek and Kaegi, 2007)*

| Parameter | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|
| Prol [kg/ha]= | 0.175 | 0.175 | 0.175 | 0.175 |
| $P_2O_5$min [kg/ha] = | 138 | 2.25 | 207 | 5.25 |
| $P_2O_5$sl [kg/ha] = | 0 | 0 | 0 | 0 |
| $P_2O_5$man [kg/ha] = | 0 | 0 | 0 | 0 |
| Fro = | 1.3450 | 1.0056 | 1.5175 | 1.0131 |
| Pro [kgP/ha]= | 0.23538 | 0.17598 | 0.26556 | 0.17730 |
| phosphate,to river [kgPO4/ha] = | 0.72131 | 0.53931 | 0.81382 | 0.54333 |
| Yield to kg conversion factor | 0.0002 | 0.000286 | 0.000143 | 0.000182 |
| ***kg PO4/kg crop*** | 1.443E-04 | 1.542E-04 | 1.164E-04 | 9.889E-05 |

Finally, for the calculation of P run-off to surface waters, as well as for heavy metals emissions into water (cadmium, ion; chromium, ion; copper, ion; lead; mercury; nickel, ion; zinc, ion; cadmium, ion; chromium, ion; copper, ion; lead; mercury; zinc, ion) and heavy metals emissions into agricultural soil (cadmium, chromium, copper, lead, mercury, nickel, zinc) secondary data from the Ecoinvent database (Nemecek and Kaegi, 2007) were used, as it was not possible to apply the specific formulae for the calculation. The following datasets were considered as basis:

- soy beans IP, at farm/kg/CH for conventional soybean;
- soy beans organic, at farm/kg/CH for organic soybean;
- barley grains IP, at farm/CH for conventional barley;
- barley grains organic, at farm/kg/CH for organic barley.

### *2.2.3 Sensitivity analysis*

Determining the environmental impact of an agri-food product is complex for several reasons. One of the challenges is the variability in natural processes. Variability is an inherent property of a system and, unlike uncertainty, it cannot be reduced by more accurate modelling of the system or collection of the data. While some variations arise from differences in cultivation practices, others are less easily explained, i.e. the difference in yield from similar field. As suggested by Röös at al. (2010) uncertainty and sensitivity analysis can be used to determine the contribution to the end result uncertainty from uncertainties in the input data and model parameters.

With regard to sensitivity analysis, different scenarios were compared in order to evaluate how the LCIA results change according to the variation of one of the most influencing parameters in the LCI of the system under study, namely the geo-climatic spring rainfall

index, which is a calculation of how much precipitation has fallen over a specific area.
The value of the rainfall index for the reference case is 232 mm (referred to year 2009). Two further extreme scenarios were introduced, considering the data for the meteorological station of Trecenta in the Province of Rovigo, provided by ARPAV (Veneto Regional Agency for the Environmental Prevention and Protection), for the spring period between the 21st March and 21st June:

- High Rainfall (HR) case (245 mm of rainfall): condition which is particularly adverse for barley as high rainfall causes a higher growth of weeds and favours parasites spread;
- Drought (D) case: (195 mm of rainfall): most unfavourable case for soybean cultivation. Low moisture index and rainfall result in a low develop of weeds and parasites.

The main differences at inventory levels from the base scenario are reported in Table 2.9, according to the data provided by the farmers under study.

**Table 2.9**. *Main differences in inventory data for the scenarios considered in the sensitivity analysis.*

| Parameter | Case | Conventional soybean | Organic soybean | Conventional barley | Organic barley |
|---|---|---|---|---|---|
| Yield (kg/ha) | HR | 5000 | 3850 | 5500 | 4320 |
| | S | 5000 | 3500 | 7000 | 5500 |
| | D | 2500 | 1750 | 7000 | 5500 |
| N° pest treatments | HR | 2 | / | 2 | / |
| | S | 1 | / | 1 | / |
| | D | 0 | / | 0 | / |
| N° weddings | HR | / | / | / | 3 |
| | S | / | / | / | 2 |
| | D | / | / | / | 2 |

Starting from these three extreme scenarios all possible combinations were considered in a three-year cycle perspective (soybean – barley – soybean).
As the first and third year refer to soybean considering the same data, some combinations are equal in terms of data inventory and therefore they have been neglected. As a consequence out of total 27 possible combinations, only 18 different scenarios have effectively been analyzed.

## 2.2.4 Mixed approach for uncertainty analysis

From the methodological point of view the uncertainty analysis in the LCI was implemented with a mixed approach combining qualitative and stochastic quantitative methods. The method adopted in the present study is derived from Sonnemann et al. (2003), integrating it with a simplified procedure used to quantify data uncertainty derived from the Ecoinvent database (Frischknecht et al., 2007).

This simplified approach developed by Ecoinvent includes a qualitative assessment by data quality indicators on the basis of a pedigree matrix. This matrix was developed and introduced by Weidema and Wesnae and has been so named (pedigree matrix), as the data quality indicators refer to the history or origin of the data just as a genealogic tree traces the pedigree of an individual (Weidema and Wesnaes, 1996). The Pedigree Matrix as implemented by Ecoinvent is reported in Table 2.10.

**Table 2.10** *Pedigree Matrix used for data quality assessment (Frischknecht et al., 2007).*

| Indicator score | 1 | 2 | 3 | 4 | 5 | Remarks |
|---|---|---|---|---|---|---|
| Reliability | Verified data based on measurements | Verified data partly based on assumptions OR non-verified data based on measurements | Non-verified data partly based on qualified estimates | Qualified estimate (e.g. by industrial expert); data derived from theoretical information (stoichiometry, enthalpy, etc.) | Non-qualified estimate | verified means: published in public environmental reports of companies, official statistics, etc<br>unverified means: personal information by letter, fax or e-mail |
| Completeness | Representative data from all sites relevant for the market considered over an adequate period to even out normal fluctuations | Representative data from >50% of the sites relevant for the market considered over an adequate period to even out normal fluctuations | Representative data from only some sites (<<50%) relevant for the market considered OR >50% of sites but from shorter periods | Representative data from only one site relevant for the market considered OR some sites but from shorter periods | Representativeness unknown or data from a small number of sites AND from shorter periods | Length of adequate period depends on process/technology |
| Temporal correlation | Less than 3 years of difference to our reference year (2000) | Less than 6 years of difference to our reference year (2000) | Less than 10 years of difference to our reference year (2000) | Less than 15 years of difference to our reference year (2000) | Age of data unknown or more than 15 years of difference to our reference year (2000) | less than 3 years means: data measured in 1997 or later;<br>score for processes with investment cycles of <10 years;<br>for other cases, scoring adjustments can be made accordingly |
| Geographical correlation | Data from area under study | Average data from larger area in which the area under study is included | Data from smaller area than area under study, or from similar area | | Data from unknown OR distinctly different area (north america instead of middle east, OECD-Europe instead of Russia) | Similarity expressed in terms of enviornmental legislation. Suggestion for grouping:<br>North America, Australia;<br>European Union, Japan, South Africa;<br>South America, North and Central Africa and Middle East;<br>Russia, China, Far East Asia |
| Further technological correlation | Data from enterprises, processes and materials under study (i.e. identical technology) | | Data on related processes or materials but same technology, OR<br>Data from processes and materials under study but from different technology | Data on related processes or materials but different technology, OR data on laboratory scale processes and same technology | Data on related processes or materials but on laboratory scale of different technology | Examples for different technology:<br>- steam turbine instead of motor propulsion in ships<br>- emission factor B(a)P for diesel train based on lorry motor data<br>Examples for related processes or materials:<br>- data for tyles instead of bricks production<br>- data of refinery infrastructure for chemical plants infrastructure |
| Sample size | >100, continous measurement, balance of purchased products | >20 | > 10, aggregated figure in env. report | >=3 | unknown | sample size behind a figure reported in the information source |

In the Pedigree Matrix data sources are assessed according to the six characteristics:

- reliability, which relates to the sources, acquisition methods and verification procedures used to obtain the data;
- completeness, which relates to the statistical properties of the data, i.e. how representative is the sample, if the sample include a sufficient number of data and if the period is adequate to even out normal fluctuations;
- temporal correlation, which represent the time correlation between the year of the study and the year of the obtained data;

- geographical correlation, which illustrates the geographical correlation between the defined area and the obtained data;
- further technological correlation, which is concerned with all other aspects of correlation than the temporal and geographical considerations;
- sample size, which refer to the size of the sample from which data are collected.

Each characteristic is divided into five quality levels with a score between 1 (best score) and 5 (worst score). Furthermore there is another factor, the so-called basic uncertainty factor, which depends on the type of process (namely agricultural, industrial process or combustion) and the type of input and output as reported in Table 2.11.

**Table 2.11** *Basic uncertainty factor as reported by Frischknecht et al. 2007.*

| Input / output group | c | p | a |
|---|---|---|---|
| **Demand of:** | | | |
| Thermal energy, electricity, semi-finished products, working material, waste treatment services | 1.05 | 1.05 | 1.05 |
| Transport services (tkm) | 2.00 | 2.00 | 2.00 |
| Infrastructure | 3.00 | 3.00 | 3.00 |
| **Resources:** | | | |
| Primary energy carriers, metals, salts | 1.05 | 1.05 | 1.05 |
| Land use, occupation | 1.50 | 1.50 | 1.50 |
| Land use, transformation | 2.00 | 2.00 | 2.00 |
| **Pollutants emitted to air:** | | | |
| $CO_2$ | 1.05 | 1.05 | |
| $SO_2$ | 1.05 | | |
| NMVOC total | 1.50 | | |
| $NO_X$, $N_2O$ | 1.50 | | 1.40 |
| $CH_4$, $NH_3$ | 1.50 | | 1.20 |
| Individual hydrocarbons | 1.50 | 2.00 | |
| Pm>10 | 1.50 | 1.50 | |
| Pm10 | 2.00 | 2.00 | |
| Pm2.5 | 3.00 | 3.00 | |
| Polycyclic aromatic hydrocarbons (PAH) | 3.00 | | |
| CO, heavy metals | 5.00 | | |
| Inorganic emissions, others | | 1.50 | |
| Radionuclides (e.g. Radon-222) | | 3.00 | |
| **Pollutants emitted to water:** | | | |
| BOD, COD, DOC, TOC, inorganic compounds ($NH_4$, $PO_4$, $NO_3$, Cl, Na, etc.) | | 1.50 | |
| Individual hydrocarbons, PAH | | 3.00 | |
| Heavy metals | | 5.00 | 1.80 |
| Pesticides | | | 1.50 |
| $NO_3$, $PO_4$ | | | 1.50 |
| **Pollutants emitted to soil:** | | | |
| Oil, hydrocarbon total | | 1.50 | |
| Heavy metals | | 1.50 | 1.50 |
| Pesticides | | | 1.20 |

Finally, six uncertainty factors are assigned to each input and output: five refer to the Pedigree Matrix, according to the correspondence between the score and the value of the factor in Table 2.12.

**Table 2.12** *Uncertainty factors as derived from the Pedigree Matrix (Frischknecht et al., 2007)*

| Indicator score | 1 | 2 | 3 | 4 | 5 |
|---|---|---|---|---|---|
| Reliability | 1.00 | 1.05 | 1.10 | 1.20 | 1.50 |
| Completeness | 1.00 | 1.02 | 1.05 | 1.10 | 1.20 |
| Temporal correlation | 1.00 | 1.03 | 1.10 | 1.20 | 1.50 |
| Geographical correlation | 1.00 | 1.01 | 1.02 | | 1.10 |
| Further technological correlation | 1.00 | | 1.20 | 1.50 | 2.00 |
| Sample size | 1.00 | 1.02 | 1.05 | 1.10 | 1.20 |

The uncertainty factors are used to calculate the square of the geometric standard deviation according to the following equation:

$$SD_{g95} := \sigma_g^{\ 2} = \exp^{\sqrt{[\ln(U_1)]^2+[\ln(U_2)]^2+[\ln(U_3)]^2+[\ln(U_4)]^2+[\ln(U_5)]^2+[\ln(U_6)]^2+[\ln(U_b)]^2}} \tag{2.4}$$

where (Weidema and Wesnaes, 1996):
- $U_1$: uncertainty factor of reliability (R);
- $U_2$: uncertainty factor of completeness (C);
- $U_3$: uncertainty factor of completeness of temporal correlation (TC);
- $U_4$: uncertainty factor of geographic correlation (G);
- $U_5$: uncertainty factor of other technological correlation (T);
- $U_6$: uncertainty factor of sample size (S);
- $U_b$: basic uncertainty factor, as reported in Table 2.11.

Once defined the value of the geometric standard deviation, a lognormal distribution is assigned to the input and output values. The lognormal distribution is the probability distribution where the natural logarithm of the observed values are normally distributed. The reason for choosing lognormal distribution is that, according to Hofstetter P. (1998), several reports in the field of risk assessment and impact pathway analysis have shown that the lognormal distribution seems to be more realistic approximation for the variability in fate and effect factors than the normal distribution.

Furthermore, the lognormal distribution is the predominant distribution used to model uncertainties in the Ecoinvent database for a number of reasons:

- the lognormal distribution is frequently observed in real life populations. One reason for this is that many real life effects are multiplicative rather than additive, and in parallel to the central limit theorem for additive effects, it can be shown that multiplicative effects will result in a lognormal distribution;
- most parameters for real life populations are always positive, and this constraint will result in a skewed distribution with a longer tail towards the higher values;
- the standard deviation of the underlying normal distribution is scale independent. This means that for a lognormally distributed vector of random values X, multiplying by a

constant a does not change the standard deviation, also not the standard deviation of the underlying normal distribution:

The procedure developed for parameter uncertainty analysis is based on 4 steps:

1. selection of the most significant LCI data for each impact category by the means of a contribution analysis, inserting a 1% cut-off, meaning that only the input out output contributing more than 1% on the overall impact score are considered;
2. definition of the lognormal probability distribution with calculation of the square of SDg from uncertainty factors defined according to qualitative assessment on LCI data, based on the Ecoinvent approach previously described (with reference to Tables 2.11, 2.12 and eq. 2.4);
3. Monte Carlo analysis, implemented by Simapro® software (PRé Consultants, 2008), which consists in randomly sampling the probability distribution of each uncertain parameter and then computing the result using the model. By performing this procedure a large number of times (in this case 1000 runs were chosen, as suggested by Frischknecht et al., 2007) a frequency histogram is constructed from the results and a probability distribution representing model results can be computed;
4. recalculation of the LCIA results and graphical representation with probability distribution (with representation of the 95% interval confidence).

## 2.3 Results and discussion

### *2.3.1 Life Cycle Impact Assessment results*

The potential impacts of the two farming techniques were calculated both at midpoint and endpoint level with the Recipe method (Goedkoop et al., 2009).

The results of the comparison between the three-year cycle considering conventional and organic farming for the 18 impact categories at mid-point level are reported in Table 2.13, meanwhile the results at endpoint level, summing the impact categories which influence the three areas of protection, are reported in Table 2.14 and Figure 2.3.

**Table 2.13** *Comparative LCIA at midpoint level for the organic cycle and conventional cycle.*

| *Impact category* | *Unit* | *Conventional cycle* | *Organic cycle* |
|---|---|---|---|
| Climate change | kg $CO_2$ eq | 1.127 | 0.990 |
| Ozone depletion | kg CFC-11 eq | 1.565E-09 | 1.822E-08 |
| Human toxicity | kg 1,4-DB eq | 2.443E-02 | 9.968E-02 |
| Photochemical oxidant formation | kg NMVOC | 2.152E-03 | 2.282E-03 |
| Particulate matter formation | kg PM10 eq | 6.322E-04 | 1.347E-03 |

| *Impact category* | *Unit* | *Conventional cycle* | *Organic cycle* |
|---|---|---|---|
| Ionising radiation | kg U235 eq | 2.752E-03 | 4.741E-02 |
| Terrestrial acidification | kg $SO_2$ eq | 2.239E-03 | 5.092E-03 |
| Freshwater eutrophication | kg P eq | 1.121E-04 | 9.410E-04 |
| Marine eutrophication | kg N eq | 1.987E-02 | 1.969E-02 |
| Terrestrial ecotoxicity | kg 1,4-DB eq | 1.750E-04 | 4.049E-04 |
| Freshwater ecotoxicity | kg 1,4-DB eq | 5.132E-03 | 5.314E-03 |
| Marine ecotoxicity | kg 1,4-DB eq | 4.171E-04 | 2.010E-03 |
| Agricultural land occupation | m2a | 2.627 | 1.885 |
| Urban land occupation | m2a | 1.186E-03 | 3.034E-03 |
| Natural land transformation | m2 | 3.869E-06 | 4.774E-05 |
| Water depletion | $m^3$ | 5.158E-05 | 6.710E-03 |
| Metal depletion | kg Fe eq | 1.309E-03 | 2.142E-02 |
| Fossil depletion | kg oil eq | 2.822E-02 | 7.630E-02 |

**Table 2.14** *Comparative LCIA at endpoint level for the organic cycle and conventional cycle.*

| *Impact category* | *Unit* | *Conventional cycle* | *Organic cycle* |
|---|---|---|---|
| Human Health | DALY | 1.81E-06 | 1.76E-06 |
| Ecosystems | species.yr | 4.33E-08 | 5.58E-08 |
| Resources | $ | 1.23 | 0.45 |

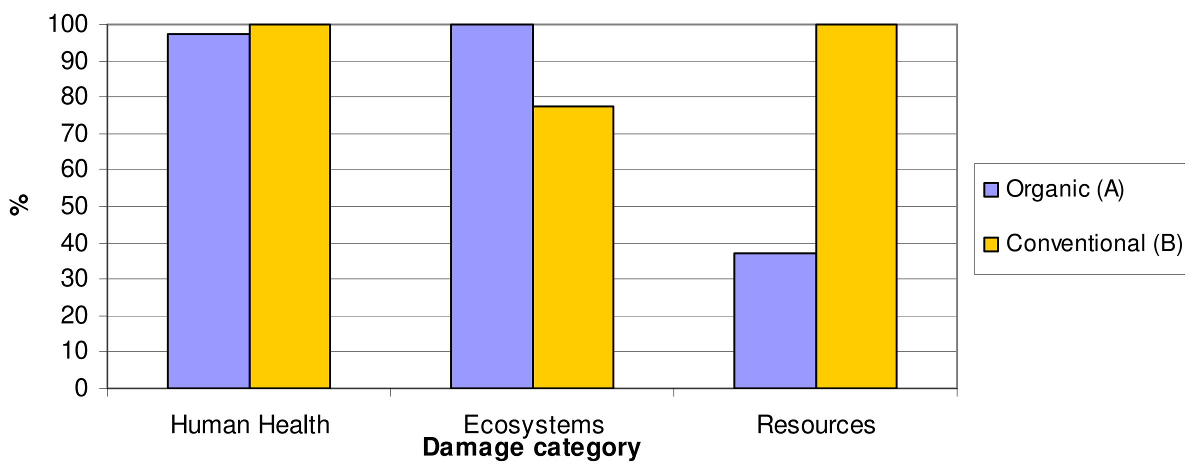


**Figure 2.3** *LCIA results from the comparative LCA between organic and conventional farming of soybean and barley*

For the two damage categories Human Health and Resources the conventional cycle resulted in higher impacts than the organic one, +2.6% and +63%, respectively, meanwhile the

damage to Ecosystems is higher for the organic cycle (+22.5%).
In order to detect the cause of the differences it is necessary to determine the most relevant unit processes inherent the whole life cycle in terms of environmental impacts. For the damage to Human Health results are similar and the main cause of damage for both cycles are Dinitrogen monoxide ($N_2O$) emissions from field derived from fertilizer utilization (triple superphosphate for conventional cycle and organic compost for organic cycle).
The difference between the two farming systems for the damage to Ecosystems is due to the lower yields (in terms of kg of product/ha) for the organic production. Considering the value of the impact category Climate Change Ecosystems similar results in terms of emissions of $N_2O$ were obtained for 1 ha of cultivated soil, but once these values are connected to the functional unit, through the yield parameter, higher impacts for the organic cycle are obtained. Furthermore, with regard to the need for arable land with reference to the functional unit, higher values are connected with the organic crops.
Finally, considering the damage to Resource*s* a significant process responsible of the impact for the two cycles is the use of diesel in the tractors for the various agricultural steps. However, the difference between conventional and organic farming is connected with the resources (oil and gas) used in the production of chemical fertilizers triple superphosphate and urea.
Considering the results of the comparative LCA, no unambiguous environmental claim regarding the superiority of one farming system versus the other one performing the same function can be stated. This result is aligned with the results of earlier LCA studies comparing organic versus conventional farming, which suggest that the outcome depends on climate conditions (Meisterling et al., 2009; Liu et al., 2010). The variability of the results is even more relevant during the conversion from conventional to organic farming (Hokazono et al., 2012).
Despite some other studies outlined the contribution of pesticide to the overall impact for conventional farming in the olive sector (Salomone and Ioppolo, 2012) and bread wheat, oilseed rape and potatoes (Williams et a., 2010) in this case it is interesting to note that the use of pesticide in the conventional cycle is not so relevant in terms of damage. This fact can be explained considering their limited amount of pesticide used in the weed and pest treatment.

### *2.3.2 Sensitivity analysis results*

The results of the sensitivity analysis with the modelling of the 18 scenarios describing the mixed cycles according to the variation of the rainfall index are shown in Figures 2.4, 2.5 and 2.6, with reference to Human health, Ecosystems and Resources, respectively.

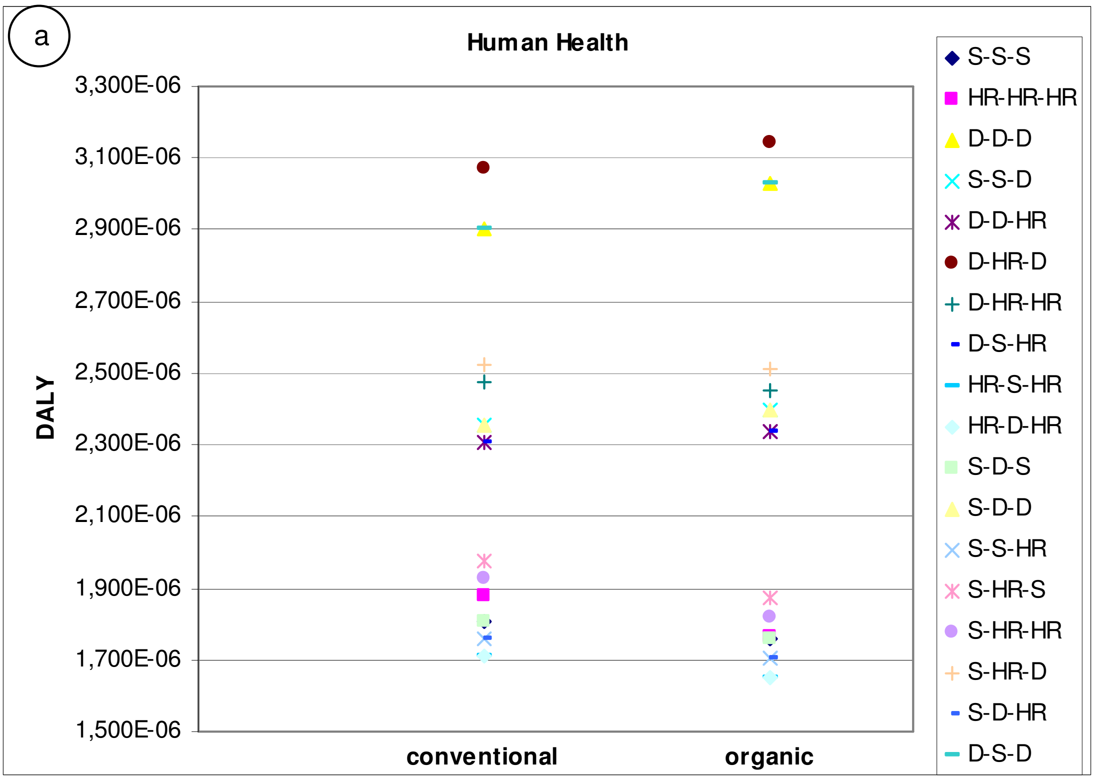


**Figure 2.4** *Results of the sensitivity analysis for Human health*

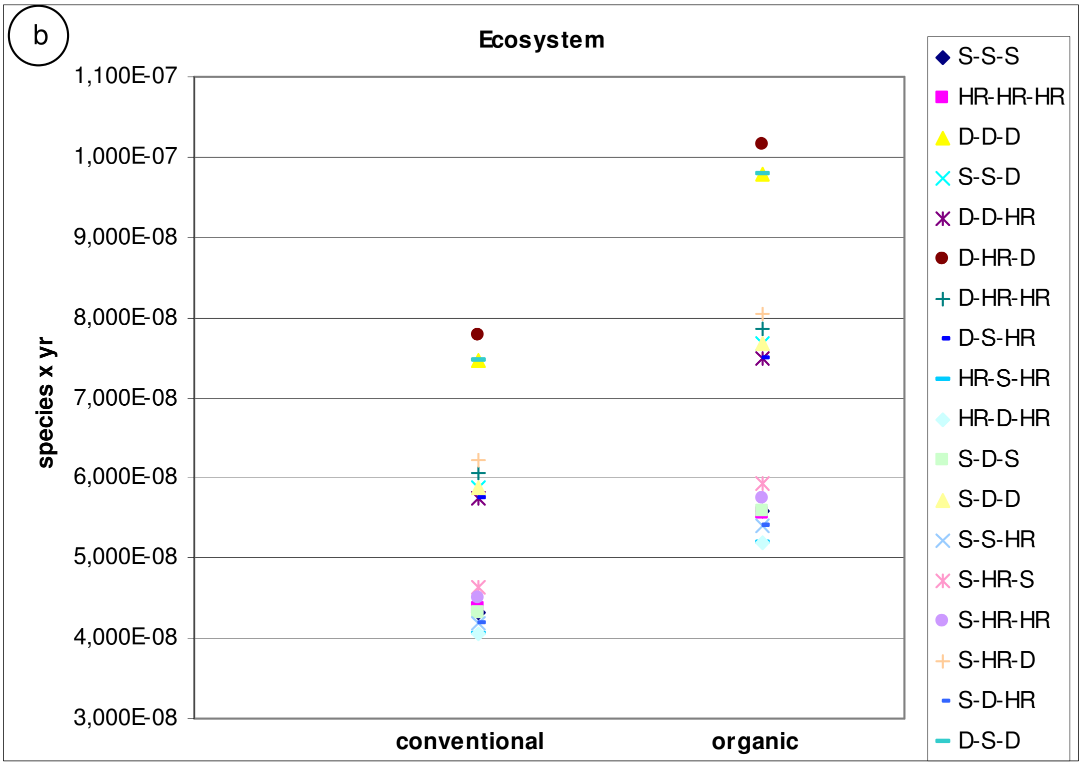


**Figure 2.5** *Results of the sensitivity analysis for Ecosystems*

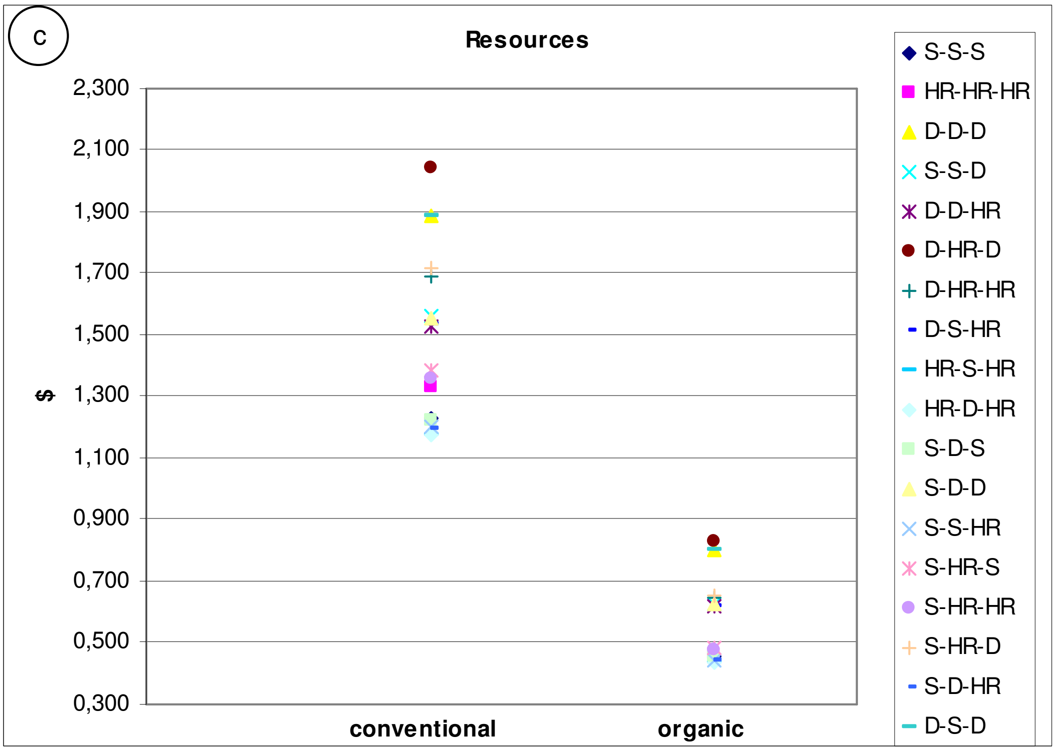


**Figure 2.6** *Results of the sensitivity analysis for Resources*

The results of the sensitivity analysis, conducted considering the effect of the spring rainfall index allowed to define a variability range of the LCIA results according to the different combinations of the different conditions high rainfall (HR), standard (S) and drought (D).
Both for conventional cycle and organic cycle the case with the highest environmental impact is given by the scenario D-HR-D, showing extreme conditions for soybean and barley with a total yield for the three-year cycle equal to 10500 kg/ha (conventional) and 7820 kg/ha (organic). On the contrary, the lower limit is given by the scenario HR-D-HR with a total yield for the three-year cycle equal to 18000 kg/ha (conventional) and 13200 kg/ha (organic). As a reference value, the total yield for the standard case (S-S-S) was 17000/ha kg for the conventional cycle and 12500/ha for the organic one. These two scenarios can be considered as the limits of an hypothetical variability range of the comparative LCA results in relation with the change of the spring rainfall parameter for the cultivation cycle under study. The main reason of this variability is connected whit the specific yields considered for the different scenarios, as summarized in Table 2.9.
As shown in Figure 2.4, the sensitivity analysis performed on the damage category *Human Health* reveals that 11 scenarios presents an higher damage for the conventional cycle than the same organic cycle, with percentage difference between 0,5% and 6,1%. For the other seven mixed-cycles considered (D-D-D, S-S-D, D-D-HR, D-HR-HR, D-S-HR, S-D-D, D-S-D) higher results for the organic cycles were obtained, with a range of difference between 1,4% and 4,4%. From Figure 2.5 for the damage category *Ecosystem*, the results of the

comparative LCA are confirmed, as for every scenario the results for the organic cycle are higher than the conventional cycle, with a percentage difference between 21,5% and 23,9%.
For the damage category *Resource* every conventional three-year cycle always results in higher damage values if compared with the same organic cycle: this differences are between 57,5% and 65,3%.
According to the results obtained by Hokazono et al. (2012) this study confirms that year-to-year variation of agricultural production has to be considered while comparing organic and conventional farming. The sensitivity analysis showed indeed that the geo-climatic parameter spring rainfall index can significantly influence yields (Austin et al., 1997; Basso et al., 2012) and therefore the comparative LCA results, shifting the preference from one farming system to the other.

### 2.3.3 Uncertainty analysis results

The 4-step procedure for uncertainty analysis defined in §2.4.4 was applied to the case study in order to quantify parameter uncertainty. The first step consisted in the selection of the most significant inventory data, by the means of the contribution analysis at damage level with a 1% cut-off. The results are reported in Table 2.15.

**Table 2.15** *Contribution analysis at endpoint level for the organic cycle and conventional cycle.*

| **Human health (DALY)** | | |
|---|---|---|
| **Process** | **Organic** | **Conventional** |
| Emission from soil management (N2O, NOX) – soybean | 1.111E-06 | 7.927E-07 |
| Emission from soil management (N2O, NOX) – barley | 3.784E-07 | 3.979E-07 |
| Diesel for tractors | 1.546E-07 | 1.143E-07 |
| Organic compost | 2.927E-08 | 1.143E-07 |
| Triple superphosphate | / | 3.794E-07 |
| Urea | / | 5.990E-08 |
| **Ecosystems (species.yr)** | | |
| **Process** | **Organic** | **Conventional** |
| Occupation arable soil - soybean | 3.997E-08 | 2.871E-08 |
| Occupation arable soil – barley | 1.277E-08 | 1.056E-08 |
| Triple superphosphate | / | 1.716E-09 |
| **Resources ($)** | | |
| **Process** | **Organic** | **Conventional** |
| Triple superphosphate | / | 6.323 E-01 |
| Diesel for tractors | 4.048E-01 | 2.994E-01 |
| Urea | / | 2.325E-01 |
| Benzo[thia]diazole-compounds | / | 1.511E-02 |
| Compost | 7.831E-03 | / |

Each of the inventory data listed in Table 2.15 was attributed a probability distribution, according to the Pedigree matrix approach. Input and output data from Ecoinvent database at unit level already incorporate lognormal probability distribution with their relative value of geometric standard deviation. The results of the calculation of the square of the geometric standard deviation for inventory data not including probability distribution, namely from other database and emissions and land use data, are reported in Table 2.16.

**Table 2.16** *Calculation of the square of the geometric standard deviation*

| ***Inventory data*** | ***SDg95*** |
|---|---|
| Diesel equipment_tractor | 1.228 |
| $N_2O$ in air | 1.184 |
| $CO_2$ in air | 1.123 |
| $NO_X$ in air | 1.184 |
| occupation arable, organic | 1.214 |

The third step of the procedure consisted in the quantitative uncertainty quantification by the means of Monte Carlo simulation with a stop criterion equal to 1000 run.
The results of the Monte Carlo simulation in terms of probability distributions are reported in Figures 2.7, 2.8 and 2.9 with regard to the organic farming. Uncertainty analysis results are given using bar charts; red lines show 95% confidence interval for the damage categories, that is the interval that includes 95% of the results and their probability distribution is related to the three damage categories.

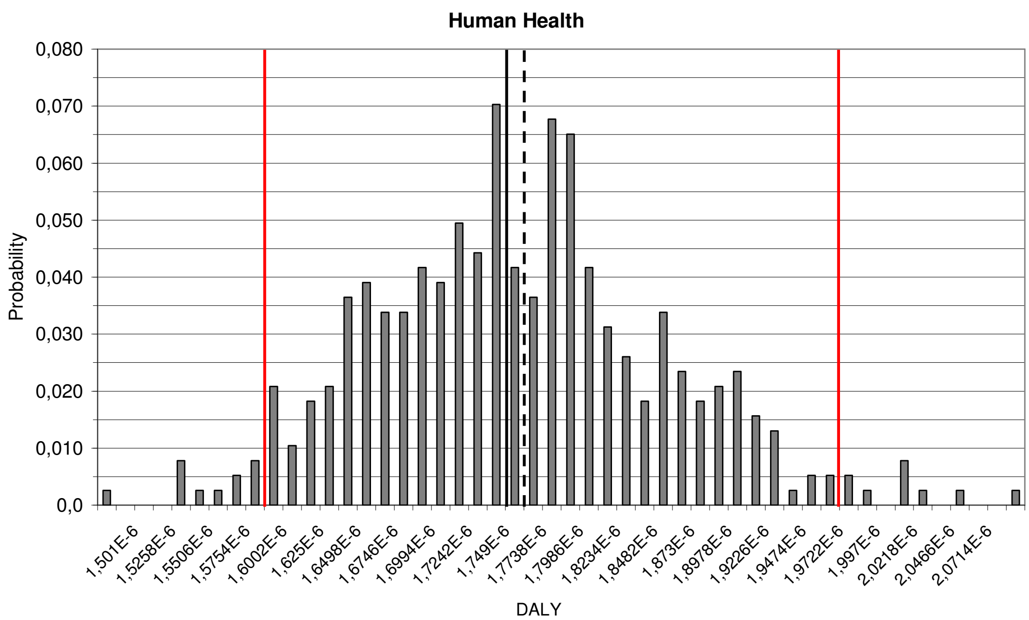


**Figure 2.7** *Probability distribution for Human health damage category.*

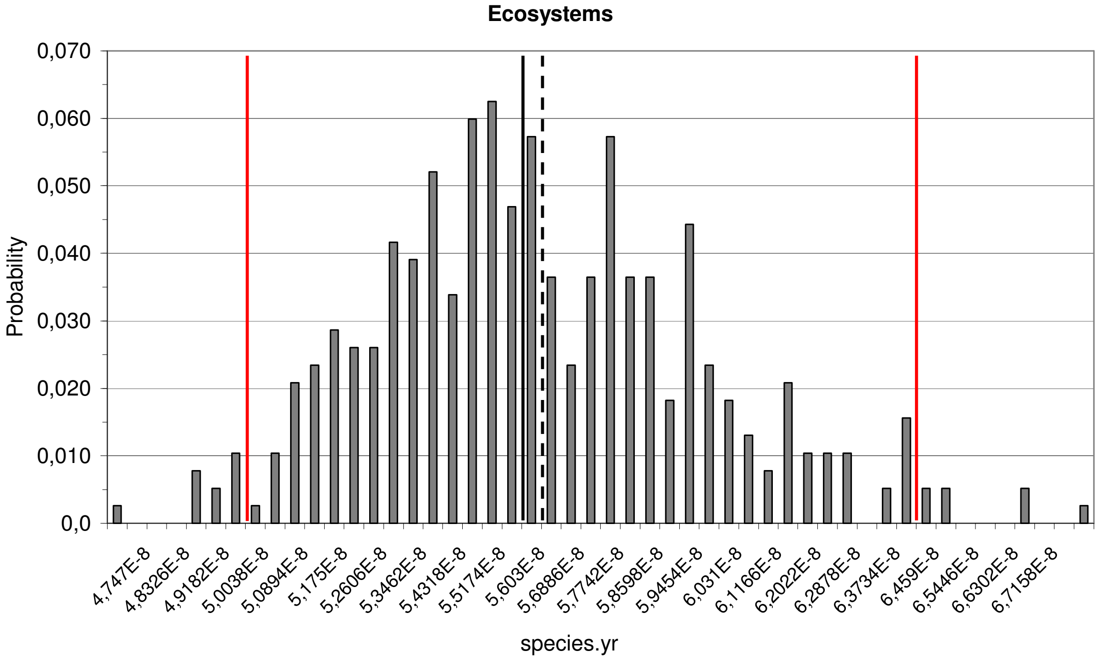


**Figure 2.8** *Probability distribution for Ecosystems damage category.*

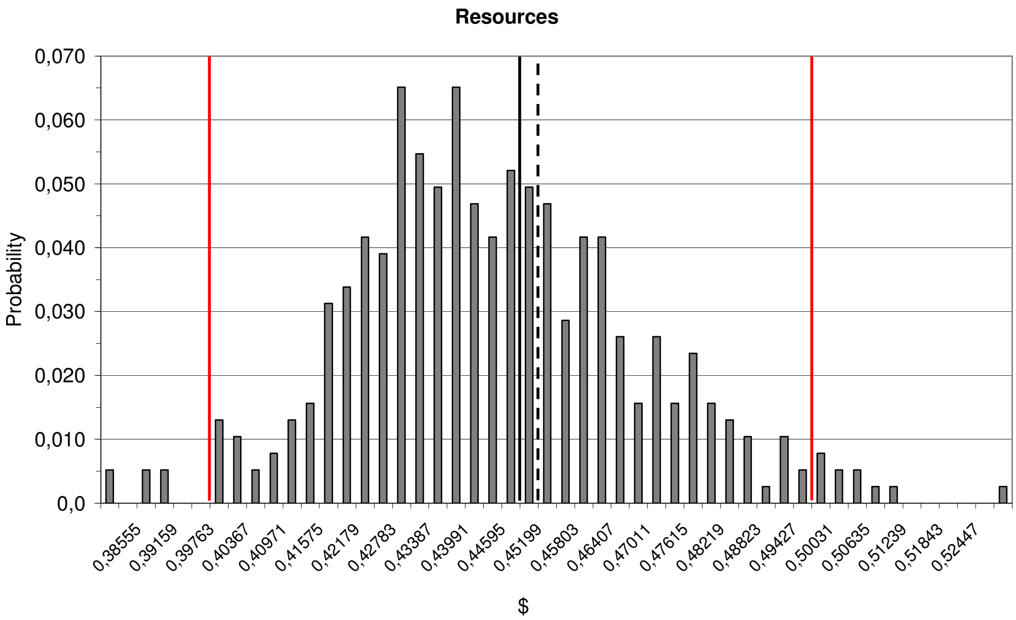


**Figure 2.9** *Probability distribution for Resources damage category.*

From the probability distribution results it emerged that for Human health impact category median value (1,75E-06 DALY) is slightly lower than the actual value from damage assessment (1,76E-06 DALY) with a standard error of 2,87E-03. For Ecosystems median value (5,54E-08 species.yr) is lower if compared with the score from actual LCA (5,58E-08 species.yr), with a standard error of 3,26E-03. Finally for Resources the median value (0,441 $) is slightly different from the one obtained from damage assessment (0,454$) and the standard error is equal to 2,74E-03.

The results of the comparative LCA with the 95% interval confidence are reported in Figures 2.10, 2.11 and 2.12.

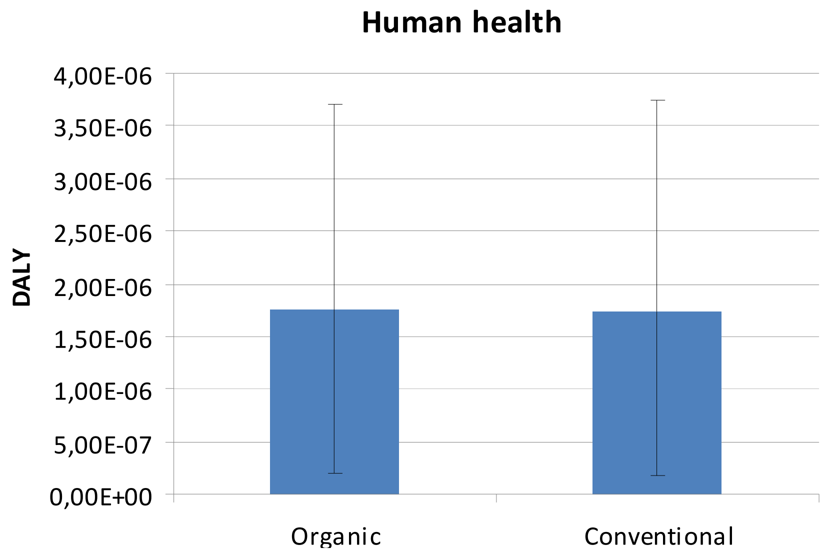


**Figure 2.10** *Confidence interval for the comparative LCA results for Human health.*

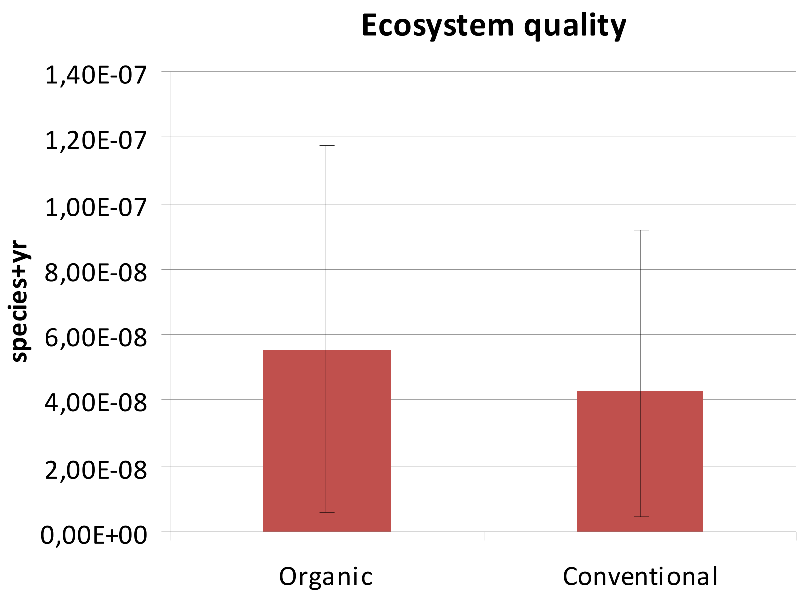


**Figure 2.11** *Confidence interval for the comparative LCA results for Ecosystems*

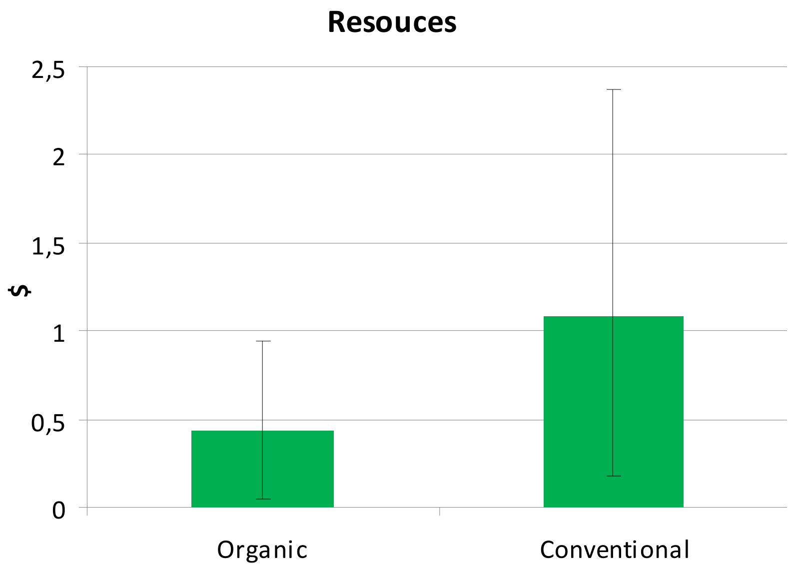


**Figure 2.12** *Confidence interval for the comparative LCA results for Ecosystems*

The interval confidence are of the same order of magnitude for Human health impact category, meanwhile significant differences can be seen for Ecosystems and Resources.
In the case of comparative LCA it is important to determine wheatear the differences among the system analyzed are consistent or not, therefore it is possible to calculate the number of comparison runs in which one product system is higher than the other. These results are shown in Figure 2.13.

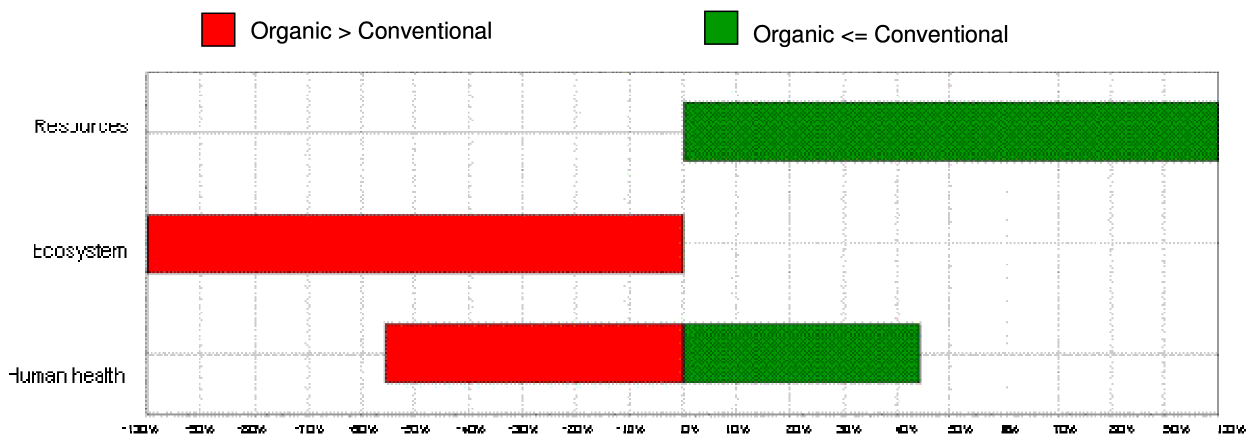


**Figure 2.13** *Results of the comparative Monte Carlo simulation.*

The output of the uncertainty analysis are the following (Niero et al., 2012):

- Resources: 100% of cases organic farming is better than conventional farming;
- Ecosystem: 100% of cases conventional farming is better than organic farming;
- Human health: 55% of cases: conventional farming has lower impacts than organic.

This graphical representation allows to see whether the differences shown in the previous figures are indeed significant. In general it can be assumed that if 90 to 95% of the Monte Carlo runs are favourable for a product, the difference may be considered significant.
Applying this rule means that only the difference between the two processes are not significant for Human health, therefore confirming that for this impact category there are no significant differences among the two farming systems.
It should be noted that also the calculation procedure developed presents some limitations, for instance it does not consider correlation influence on uncertainty (Frischknecht et al., 2007). It happens indeed with some frequency that the input and output of a unit process (and thus the correlated uncertainty) are dependent upon each other. For instance, input fuels and $CO_2$ emissions present a linear relation, but are treated as independent in the current uncertainty calculations, which tends to overestimate the uncertainty of real processes.
Furthermore, basic uncertainty factors $U_b$ were derived from subjective evaluations of a group of independent analysts, therefore thy tend to under estimate the "real" uncertainty.

## 2.4 Conclusions

LCA methodology was applied to compare the environmental impacts of soybean and barley production by conventional or organic farming using data from a three year cycle that was provided by a farm in northeast Italy. The results showed that no unambiguous comparative assertion between the two farming systems can be stated. The conventional cycle revealed better performances for ecosystems, due to the higher yield per hectare, meanwhile organic farming showed better performance for resources, due to the lesser fossil fuels consumption.
For human health damage category the difference among the two production system is very small (2,6%).
As crop yields are influenced by geo-climatic parameters, the effect of the spring rainfall index on the LCIA results was evaluated by the means of sensitivity analysis performed at endpoint level. Two further borderline scenarios were identified, High Rainfall and Drought, based on the value of the spring rainfall index. All of the possible combinations of the three scenarios (standard, high rainfall, drought) in a three year cycle (soybean - barley - soybean) were taken into account. As a result, 18 different scenarios were evaluated, and a variable range of LCIA results was defined. The effect of other geo-climatic parameters, such as spring sudden changes in temperature and water stock in winter time, needs to be further separately investigated for, finally, trying to evaluate all together these parameters.
Finally a 4-step procedure was elaborated for quantifying the uncertainty connected with the Life Cycle Inventory, based on a mixed approach combining qualitative and stochastic quantitative methods. The qualitative analysis with the use of the Pedigree matrix allowed to define a qualitative judgement on inventory data. Secondly, the quantitative uncertainty

analysis (geometric standard deviation quantification and Monte Carlo simulation) quantified how data quality translated into uncertainty level on the Life Cycle Impact Assessment results. For ecosystems and resources the results of the comparative LCA are confirmed with a level of statistical significance of 100%, meanwhile for human health the results confirmed that there are no significant differences among the two farming systems. The results of uncertainty analysis indicate that uncertainty intervals are useful in understanding the stability of results.

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# Chapter 3

# Comparative LCA of tissue paper wipers

This chapter[4] introduces a case study in the tissue paper sector, where LCA methodology was applied in order to compare three different type of wipers for professional use. They differ for the raw materials used to manufacture the product: virgin pulp, waste paper and fibers recovered from the recycling of beverage cartons. The results of the comparative LCA are discussed considering the outcomes of the sensitivity analysis, changing the LCIA method. Furthermore the 4-step procedure for uncertainty quantification described in Chapter 2 is applied comparing two products at a time.

## 3.1 Introduction

One of the most active industrial sectors towards the minimization of its impacts on the environment is the paper tissue industry. Tissue products are used help to ensure health, hygiene, and well-being at home (e.g. facial tissues, bathroom tissue and paper towel) and away from home (e.g. hand towels, wipers and washroom products). Tissue products can be made starting from virgin wood fibre, fibres derived from paper recycling operations or a combination of the two. The use of recycled material by industries is driven by a long standing commitment to make the best use of all available resources that balance the sometimes competing business requirements of responding to customer and consumer mandates, expectations and perceptions that recycled materials offer environmental benefits, meeting product performance requirements in a highly competitive market and controlling raw material costs to maintain profit margins. In order to develop a more complete understanding of the environmental performance of tissue products containing responsibly managed virgin and recycled fibres, Life Cycle Assessment can be used to provide solutions to this open issue, whether the use of recovered fibres to manufacture new tissue paper products is actually better for the environment than using new fibres.

Many researchers have focused on the comparison of different hand drying systems, including not only tissue papers hand dryers, but also other systems, such as textile towel rolls or

[4] *[1] The topic addressed in this section is part of publication Niero et al., (2012).*

electric dryers (Eberle and Möller, 2006, Dettling and Margni, 2009, Neitzel, 1997). These studies are based on a cradle to grave perspective, therefore focusing on the whole life cycle performance of the system, including not only the production stage but also use, distribution and end of life.

For companies manufacturing tissue paper products there is an interest in understanding which is the more sustainable production system between the use of virgin and recycled fibres. A previous LCA study from Environmental Resources Management (2007) concluded that there is no environmental preference between using recycled or virgin fibres in the manufacture of a selection of tissue products (bathroom tissue, washroom towel, facial tissue, kitchen towel, commercial wipers ). Within the tissue product systems, five impact categories were closely related to the burning of fossil fuels: natural resource depletion, acidification, global warming potential, photochemical oxidation (smog) and ozone layer depletion. Combining those five categories related to the burning of fossil fuels for all products results in showing that the use of virgin fibres is better than the use of recycled fibres. The same situation is revealed for eutrophication impact category. On the contrary for human toxicity better performances are connected with the use of recycled fibres. Finally no preference was shown for water use and solid waste production.

An important guidance for the conduction of LCA study within the tissue paper sector is given by Product Footprint Category Rules (PFCRs) for Intermediate Paper Products (CEPI, 2011). PFCR is linked to the forthcoming European methodology for the calculation of environmental footprint, currently being developed by the European Commission. As defined in ISO 14025 (2006), PCRs (Product Category Rules) include sets of specific rules, guidelines and requirements that are aimed at developing Type III environmental declarations, which are based on LCA study. Even though there is a guidance on how to assess the potential environmental impacts of tissue paper products, it must be recognized that the most critical phase in the conduction of an LCA study is data collection. In particular, when data are collected from a specific plant and are considered as representative of a well defined production process, it is important to assess how the assumptions made on data can influence the outcomes of the study. Therefore the relevance of assumptions should be tested by the means of sensitivity analysis and uncertainty analysis, whose role is relevant when the goal of the LCA is to make an environmental claim regarding the superiority of one product versus another product with the same function.

Starting from these premises, this study contributes to the debate on the use of virgin and recycled fibres in the tissue paper sector, considering the case study of an Italian company which conducted a comparative LCA between three different wipers, made using different raw materials: virgin pulp, waste paper and paper obtained from the recycling of beverage cartons (Niero et al., 2012). The aims of the study were twofold: (i) to define which is the raw materials leading to the best environmental performances, and (ii) to test the reliability of the comparative LCA results by the means of sensitivity and uncertainty analysis, with the 4-step procedure described in Chapter 2.

## 3.2 Materials and methods

### *3.2.1 Goal and scope definition*

The aim of the LCA study was to conduct a comparative LCA of three different types of wipers for professional use, made using different raw materials: virgin pulp (hereafter product A), waste paper (hereafter product B) and recovered fibres from beverage cartons recycling, both from post-consumer use and waste from laminated carton containers manufacturing (hereafter product C). The main characteristics of the wipers are reported in Table 3.1.

**Table 3.1**. *Main characteristics of the wipers for commercial use included in the study.*

| Product | N° plies | Grammage | Sheets x package | Sheet dimension | N° package x item |
|---|---|---|---|---|---|
| A. Virgin pulp | 2 | 21.5 $g/m^2$ | 800 | 250mm x 370 mm | 2 |
| B. Waste paper | 2 | 22 $g/m^2$ | 800 | 250mm x 370 mm | 2 |
| C. Pulp fibers recovered from beverage cartons | 2 | 19 $g/m^2$ | 800 | 250mm x 250 mm | 2 |

The function of tissue paper is manifold and normally separated into primary and secondary functions. Primary functions include: hygiene, absorbency, strength and softness.

Secondary functions include image, luxury, quality and consumer satisfaction. In this case, as the product under study are used for professional use, the function is surface cleaning, and the functional unit is 1 kg of tissue paper. In the system boundaries definition a cradle to gate perspective was adopted, including all the processes from raw and auxiliary materials extraction, manufacturing, packaging of the final product, and waste management and transports.

A schematic overview of the system boundaries with inclusion of the main life cycle stages considered in the study, is reported in Figure 3.1. The input considered inside system boundaries are: raw and auxiliary materials, electricity and fuels, water, chemicals (both for the manufacturing and wastewater treatment).

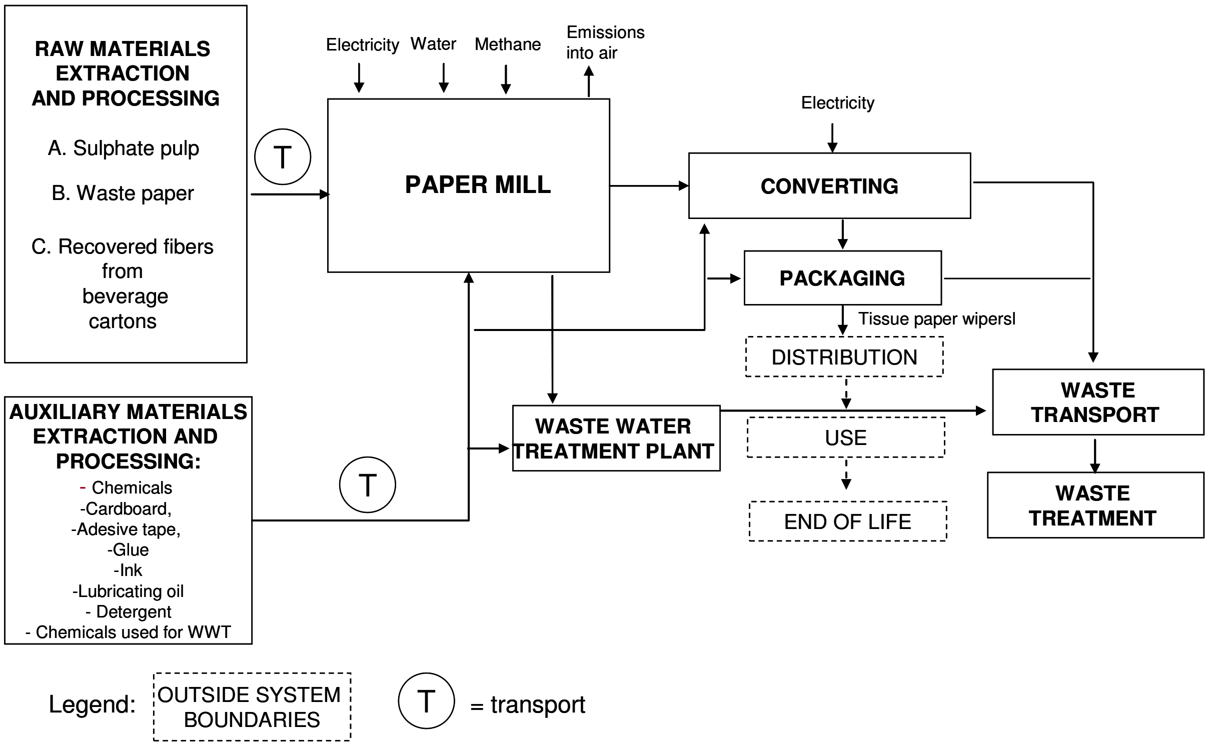


**Figure 3.1** *System boundaries of the product system analyzed.*

The manufacture, maintenance and decommissioning of capital equipment, such as buildings or machines, were not included in the investigated system. The reason for excluding capital equipment, besides the practical aspects, was that the environmental impact related to the functional unit is negligible.

The cut-off criteria for initial inclusion of inputs and outputs was on based on 1% mass basis (the process is neglected if it reaches less than 1% of the total known mass), but all processes where data are available were taken into account, even if their contribution was less that 1%. Therefore in accordance with Humbert et al. (2009), the cut-off rule is used to avoid gathering unknown data, but not to neglect known data.

### *3.2.2 Life Cycle Inventory*

This step includes the description of the process units which define the system products under study.

For product A the main raw material is virgin pulp, to which auxiliary non fibrous raw materials are added. The formulation of the pulp with the auxiliary chemicals is essential to determine the characteristics of the finished sheet, such as appearance, quality, as well as the strength of paper. Raw auxiliary materials are fed into the first parte of the paper mill, namely the pulper (or kneader). The pulper contains water and has the purpose of pulping, i.e. uniformly disperse the fibrous material in the water.

The fibrous raw materials are subsequently refined, blended, diluted and purified and go to paper production. The fibrous slurry in suspension, kept in agitation, is diluted, admixed with any chemicals to bring it to a consistency of 0.2 to 0.3 g/l and then it is sent to the paper machine, which has the function to uniformly distribute the fibers on the forming. The water is drained through the forming fabric while the fiber is distributed on it forming the sheet. The sheet format is thus transferred on the felt, which is made of synthetic and porous material, and has the function of carrying the sheet into the part of the drying section, of the continue machine. Through the pressing cylinders, the paper and the felt are crushed to remove the excess of water and transferred to the dryer for final drying.

With regard to product B, as the raw material is recovered paper, there is a cleaning machine, where contaminants such as plastic parts, metallic, sand stickies are removed in the so-called pulper waste. Once the fibre is cleaned, it is sent to the deink process, where coloured parts are removed. At this stage the pulp is sent to the continue machine and follows the same pattern of product A.

Finally, in the case of product C, as raw material are the fibres from the recycling of beverage cartons, after the pulper there is a plant for the separation of the plastic and aluminium component of the laminated carton container.

For all the three products, once the chemicals are added, the semi finished product in the form of reel is stocked and then sent at the converting stage, where packaging and preparation into pallet of the wipers takes place.

The life cycle of the tissue paper wipers was modelled with Simapro software (PRé Consultants, 2008) into 9 life cycle stages, the first 6 connected with the semi finished product manufacturing, and the other three connected with the converting stage and production of the final product:

1. Raw materials transport, which includes the transport of raw materials from their production site to the paper mill. in the case of product A and B transport are by both by ship and truck, meanwhile for product C only truck are used.
2. Auxiliary materials and chemicals transport, which includes the transport by truck of chemicals used in the paper mill;
3. Tissue production, which includes all the operations for transforming raw materials into the semi finished reel. It includes raw materials, as well as chemicals and resources consumption, as well as the operation for wastewater treatment;
4. Emission into air, which includes the emissions of nitrous oxides, particulates, carbon dioxide and carbon monoxide into air;
5. Waste transport and treatment, which includes pulper waste, as well as paper sludge final disposal;
6. Warehousing and internal movement, which includes consumption of diesel for internal movement as well as packaging film for packaging of the semi finished reel and lubricating oil for maintenance operations;

7. Transport of chemicals and auxiliary materials used in the converting phase, which includes transport by truck;
8. Converting, which includes chemicals and auxiliary materials (labels, glue, adhesive tape, ink), as well as electricity consumption in the converting stage;
9. Packaging, which includes the consumption of packaging for the preparation of the final product.

Data for the LCI step were mainly primary data from one specific plant which manufactures all the three types of tissue wipers, considering 2010 as reference year.

Data collection was particularly challenging, because data on resources consumption were aggregated at plant level, therefore it was necessary to find out a procedure for quantifying the differences between the products.

The procedure is shown in Figure 3.2 and is based on a modelling processes which differs according to the life cycle stage. As far as upstream data are concerned, namely data about raw and auxiliary materials production, they were available at product level, meanwhile data about consumption in the paper mill and converting phase were available at plant level. Therefore the process started with the analysis of historical data about product A, which were considered as the reference values, then the main differences in the production between product A and products B and C, respectively were quantified. The third step consisted in the effect analysis, i.e. the quantification of the way in which the different raw materials influence resource consumption.

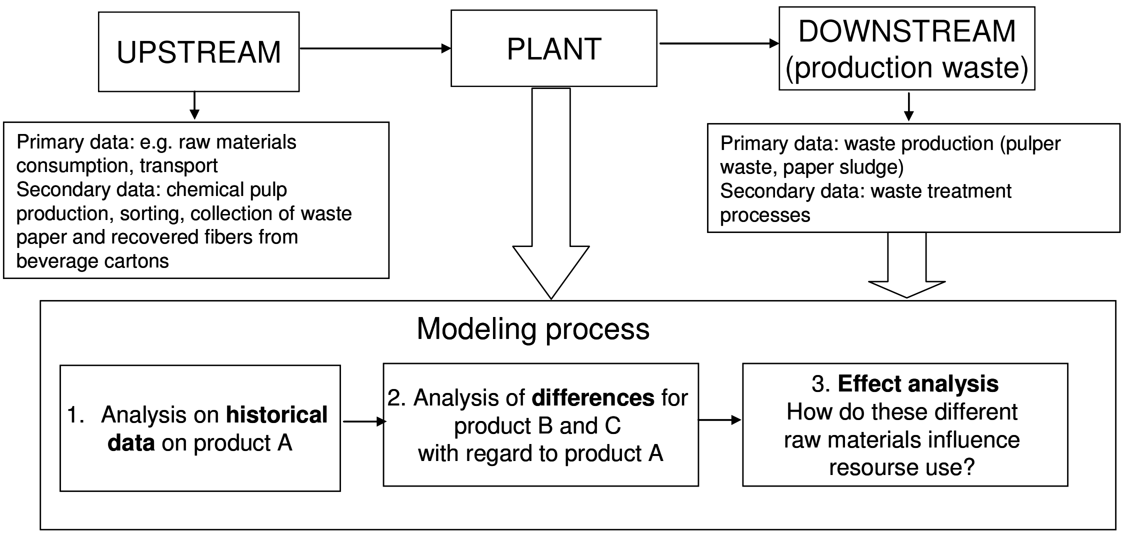


**Figure 3.2** *Modelling process adopted for data collection*

According to the modelling process previously defined it was possible to define the main input and output for each product system, as summarized in Table 3.2.

**Table 3.2**. *Main input and output data referred to 1 ton of tissue paper*

| Inventory data | Product A | Product B | Product C |
|---|---|---|---|
| | | Input | |
| Raw materials | 1.05 ton chemical pulp | 1.30 waste paper | 2.02 fibres from beverage carton recycling |
| Electricity | 1190.39 MWh | 1120.49 MWh | 1131.58 MWh |
| Water | 15.71 $m^3$ | 20.42 $m^3$ | 21.99 m3 |
| Methane | 277.75 $m^3$ (cogeneration) + 96.93 $m^3$ (converting + boiler) [a] | | |
| | | Output | |
| Pulper waste | 5% of input virgin pulp | 30% of input waste paper | 50% of input beverage cartons |
| Paper sludge | 54.6 kg | 54.6 kg | 54.6 kg |
| Pulper waste end of life | Landfill (43%) + Incineration with energy recovery (57%) [a] | | |
| Paper sludge end of life | Landfill (0.54%) + Compost (10.71%) + Environmental recovery (35.14%) + Brick production (53.61%) [a] | | |

[a] These values are common for all the three products A, B and C.

Despite the adoption of the modelling process described in Figure 3.2, for some operations, such as waste water treatment and emissions into air, the average plant values of the year 2011 were taken into account, as reported in Table 3.3.

**Table 3.3**. *Average plant value taken as reference for all the three products*

| Inventory data | Value |
|---|---|
| Wastewater | |
| Inlet wastewater | 19,6 $m^3$/ton tissue paper |
| Electricity | 36,7 kWh/ton tissue paper |
| Antifouling | 0.0111 kg/ton tissue paper |
| Polialluminium chloride | 0.964 kg/ton tissue paper |
| Poliammine | 0.113 kg/ton tissue paper |
| Cationic polymer | 0.120 kg/ton tissue paper |
| Anionic polymer | 0.077 kg/ton tissue paper |
| Liquid oxygen | 1.482 kg/ton tissue paper |
| Bacterial nutrient | 0.157 kg/ton tissue paper |
| Sodium hypochloride | 0.115 kg/ton tissue paper |
| Outlet water | 15.090 $m^3$/ton tissue paper |
| COD | 53 mg/l/ |

| Inventory data | Value |
|---|---|
| Suspended solids | 10 mg/l/ |
| Total N | 0.61 mg/l |
| Total P | 0.27 mg/l |
| Emissions into air | |
| $NO_X$ | 99.9 ton |
| Particulates | 7.46 ton |
| $CO_2$ | 76.805 ton |
| CO | 1.56 ton |

Concerning chemicals consumption, a distinction can be made according to the types of chemicals used by all the products (biocides, anionic polymers, cationic polyamine, resins for wet resistance, antifouling, auxiliaries for mechanical resistance) and specific chemicals:

- coating, protective (i.e. monoammonium phosphate) for product A;
- antistickies, sodium ipochloride for product B and C;
- hydrogen peroxide for product C.

Some relevant assumptions within the LCI phase are connected with the allocation problem. Allocation is the partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems (ISO 2006a). In the case of the production process for product C, there is also a co-product, namely the residual plastic and aluminium component. Allocation on mass basis was chosen in order to define the amount of resources attributable to product C. Therefore the following subdivision was done, based on the final mass of the two coproducts:

- 75% of impacts of the production process allocated to the production of the semi finished reel of product C;
- 25% of impacts allocated to the production of the residual aluminium and plastic component.

From a methodological point of view, the cut-off approach was taken into account with regard to the recycling of paper. The cut off approach is the one defined within the Ecoinvent database (Frischknecht et al. 2005), which attributes the environmental impacts of raw materials extraction and processing of primary paper to the first use of that paper product, meanwhile the second use of the paper bears the environmental impacts of collection and re-processing of scraps (Frischknecht R., 2010).

## 3.3 Results and discussion

### 3.3.1 Life Cycle Impact Assessment results

ReCiPe method (Goedkoop et al., 2009) was used to perform impact assessment at midpoint level, considering the following impact categories: climate change, photochemical oxidant formation, terrestrial acidification, freshwater eutrophication, particulate matter formation, agricultural land occupation, fossil depletion. The first four impact categories were selected in accordance to the requirements of the PFCR for Intermediate Paper product (CEPI, 2011), meanwhile the last three were considered significant by the company who provided the data for the LCA study.

The results of the comparison between the three products for the selected impact categories at mid-point level are reported in Table 3.4 in absolute value and in Figure 3.3 as percentage.

**Table 3.4** *Comparative LCIA at midpoint level for the three tissue paper wipers with Recipe.*

| *Impact category* | *Unit* | *A* | *B* | *C* |
|---|---|---|---|---|
| Climate change | kg $CO_2$ eq | 3.53 | 3.53 | 2.01 |
| Photochemical oxidant formation | kg NMVOC | 1.03E-02 | 8.01E-03 | 4.51E-03 |
| Particulate matter formation | kg PM10 eq | 4.04E-03 | 2.93E-03 | 1.58E-03 |
| Terrestrial acidification | kg $SO_2$ eq | 1.17E-02 | 9.20E-03 | 4.63E-03 |
| Freshwater eutrophication | kg P eq | 5.16E-04 | 3.87E-04 | 1.98E-04 |
| Agricultural land occupation | m2a | 7.47 | 6.30E-01 | 9.48E-01 |
| Fossil depletion | kg oil eq | 9.96E-01 | 9.34E-01 | 5.25E-01 |

The results of impact assessment show that wipers made by virgin pulp (product A) have the greatest environmental impacts for all the impact categories considered, except for the climate change for which virgin fibre rolls and waste paper wipers (product B) have the same impact. The less impacting products are wipers made by the paper from beverage carton recycling (product C). Furthermore, for Agricultural land occupation product C has slightly higher impacts than product B.

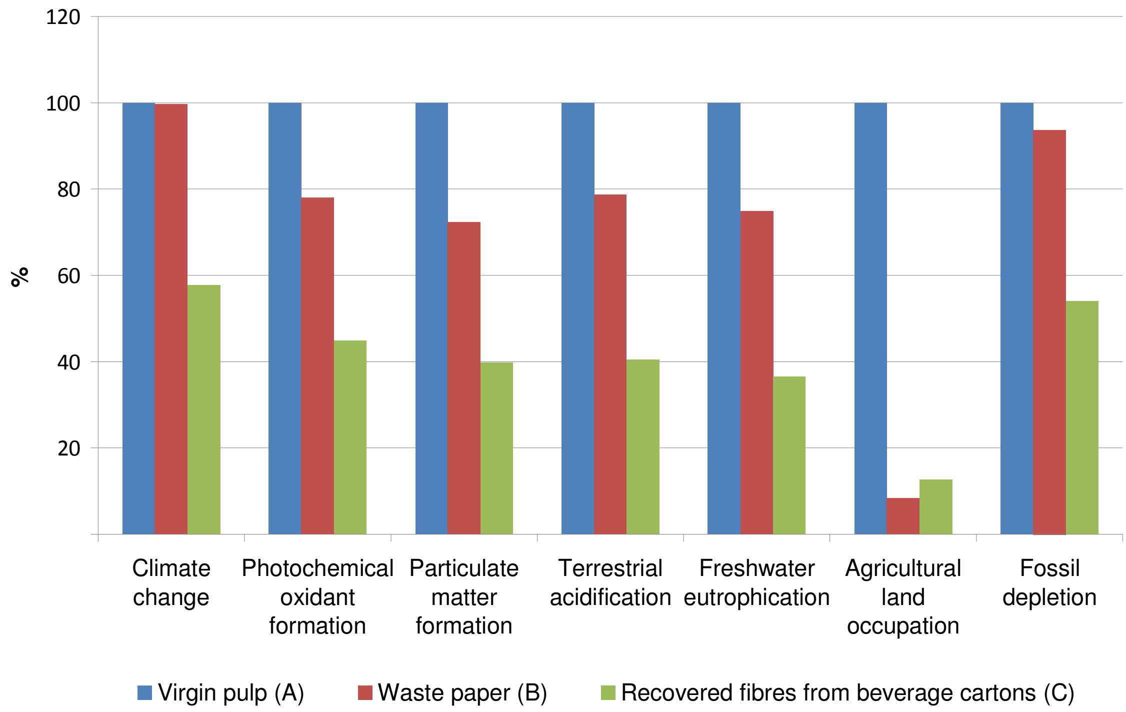


**Figure 3.3** *LCIA results from the comparative LCA between the three tissue paper wipers with Recipe.*

The results of the LCIA for each product are shown in Figures 3.4, 3.5 and 3.6.

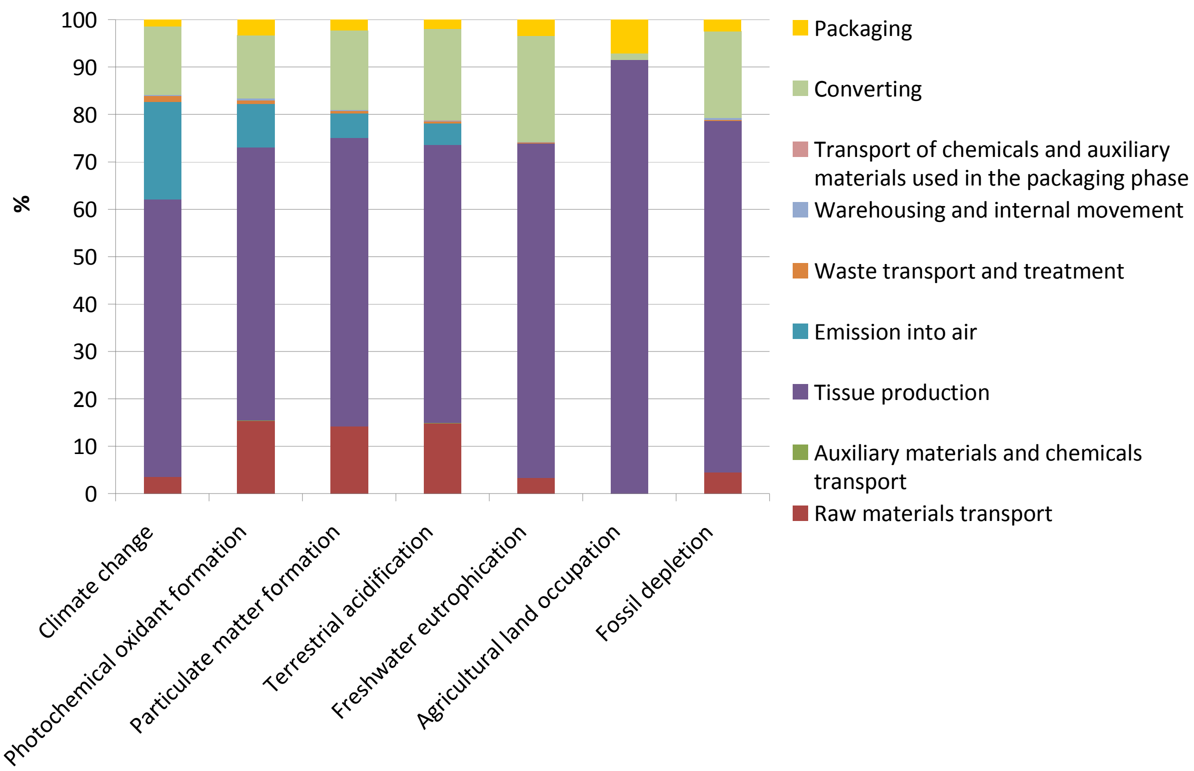


**Figure 3.4** *LCIA results for wipers made from virgin pulp (Product A).*

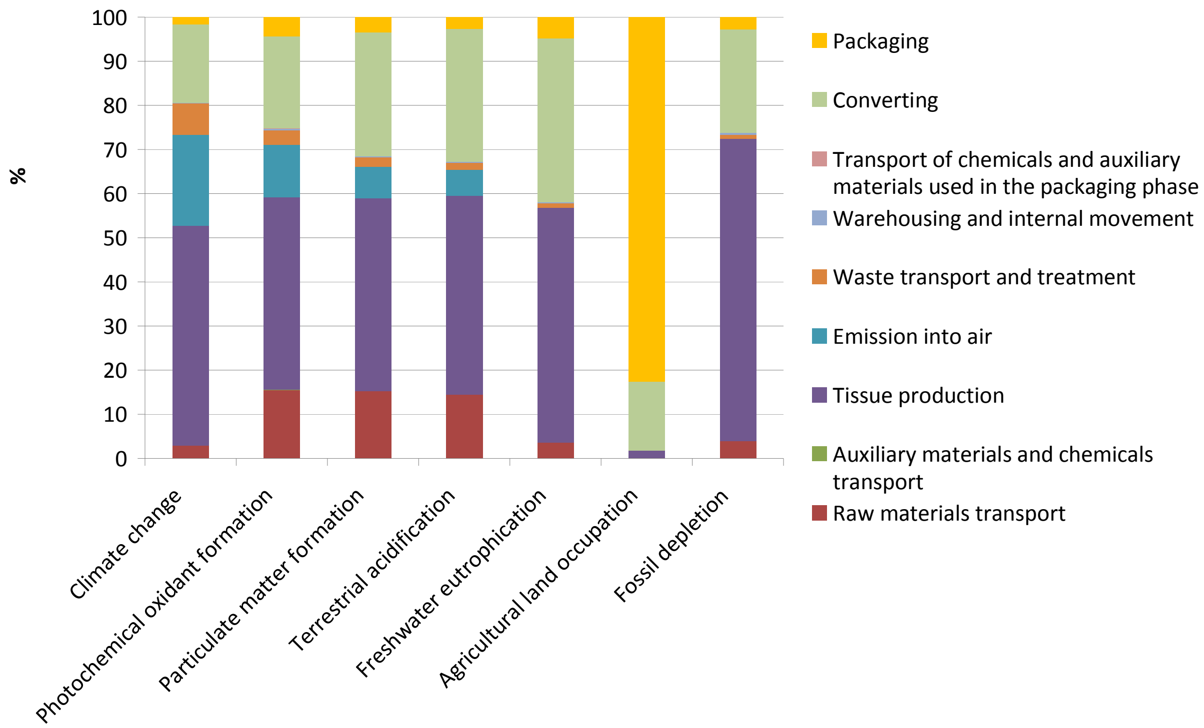


**Figure 3.5** *LCIA results for wipers made from waste paper (Product B).*

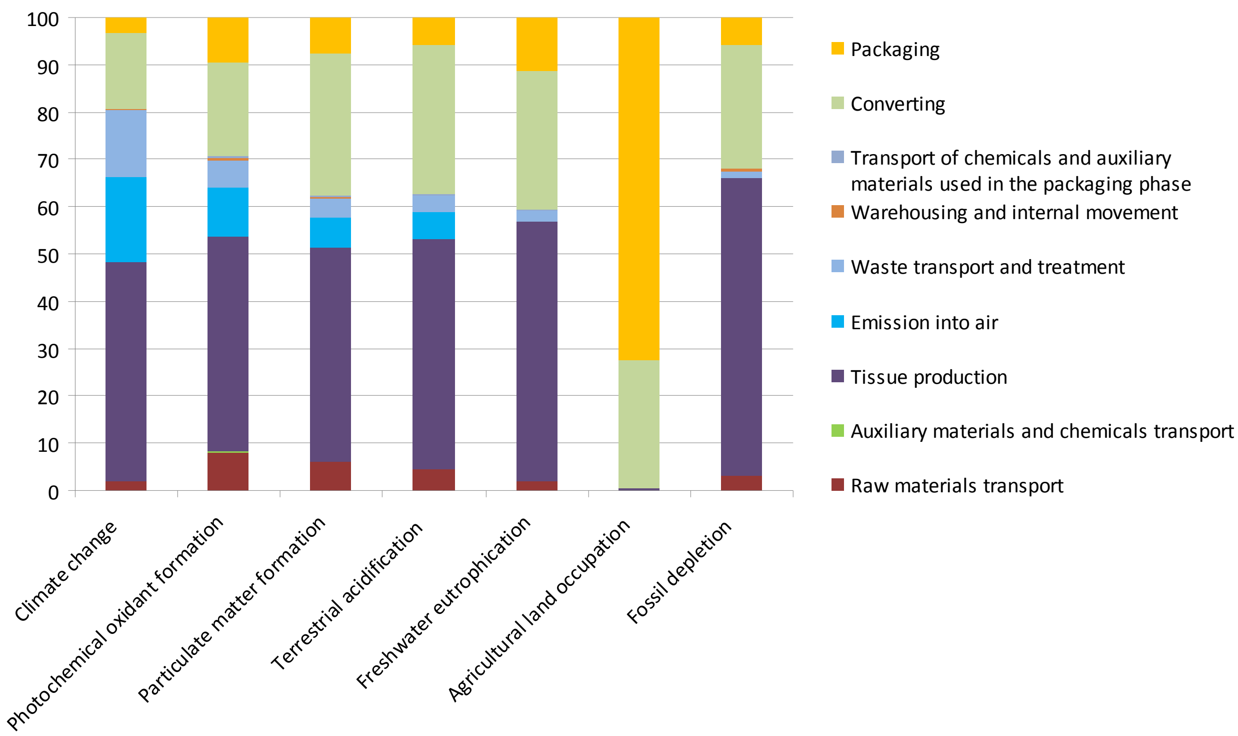


**Figure 3.6** *LCIA results for wipers made from recovered fibres from beverage cartons (Product C).*

For all the products the main impacting life cycle stage is tissue production, for all impact categories. With regard to product A, the substances responsible for the impact for each impact categories are the following:

- fossil carbon dioxide emissions from electricity and methane use for climate change, as well as from chemical pulp production;
- Nitrogen oxides,non-methane volatile organic compounds (NMVOC) and sulphur dioxide emissions from electricity and methane use for photochemical oxidant formation;
- Nitrogen oxides and particulates emissions from electricity production and chemical pulp production for particulate matter formation;
- Sulphur dioxide, nitrogen dioxide and ammonia emissions from electricity production and chemical pulp production for terrestrial acidification;
- Phosphate emissions into water from electricity production and chemical pulp production for freshwater eutrophication;
- Forest occupation from chemical pulp production for agricultural land occupation;
- Natural gas consumption within the electricity and methane production for fossil fuels.

These results are confirmed for product B and C, which do not include the contribution of chemical pulp production, therefore from the contribution analysis the impact connected with tissue production is lower if compared with product A.

The contribution of the life cycle stage is comparable for products B and C, except for the raw materials transport, which is lower for product C, due to the fact that the supply of the beverage cartons is at local level and only the contribution of transport by truck is included.

Furthermore, for agricultural land occupation, the absolute value for product B and C is considerable lower than the value for product A, and is mainly due to forest occupation during pallet and cardboard production (negligible if compared with the contribution of chemical pulp production).

### 3.3.2 Sensitivity analysis results

The sensitivity analysis was performed changing the impact assessment methodology: Impact 2002+ (Jolliet et al., 2005) including the following impact categories: respiratory inorganics, respiratory organics, terrestrial acidification/nutrification, land occupation, global warming, non-renewable energy

The results are shown in Table 3.5 and Figure 3.7.

**Table 3.5** *Comparative LCIA at midpoint level for the three tissue paper wipers with Impact 2002+.*

| ***Impact category*** | ***Unit*** | ***A*** | ***B*** | ***C*** |
|---|---|---|---|---|
| Respiratory inorganics | kg PM2.5 eq | 2.21E-03 | 1.51E-03 | 8.48E-04 |
| Respiratory organics | kg C2H4 eq | 9.11E-04 | 7.50E-04 | 5.21E-04 |
| Terrestrial acid/nutri | kg SO2 eq | 5.18E-02 | 4.00E-02 | 2.26E-02 |
| Land occupation | m2org.arable | 8.54E-01 | 7.66E-02 | 1.16E-01 |
| Aquatic eutrophication | kg PO4 P-lim | 7.54E-04 | 2.37E-04 | 1.40E-04 |
| Global warming | kg CO2 eq | 3.42 | 3.41 | 1.92 |
| Non-renewable energy | MJ primary | 47.08 | 43.48 | 24.19 |

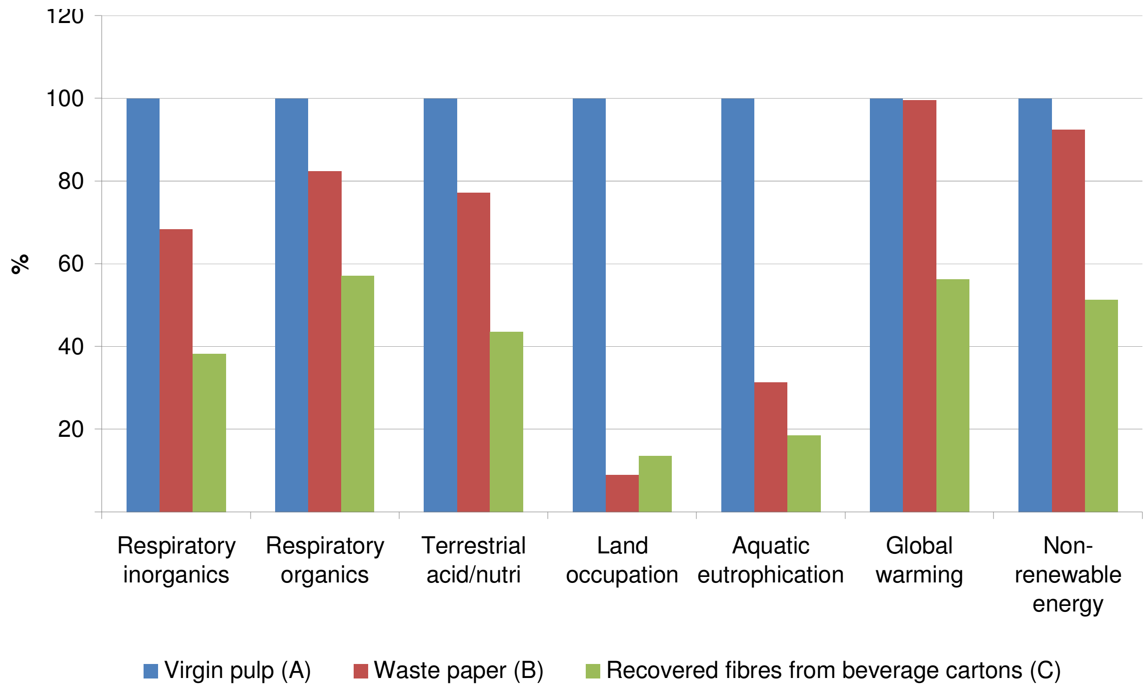


**Figure 3.7** *LCIA results from the comparative LCA between the three tissue paper wiper with Impact 2002+.*

The results show that product A has the worst performance for all impact categories, except for Global warming for which also product B show the same impact.

In order to compare the results of the two methodology, it was necessary to define a correspondence within the different impact category. The reference was the classification provided by the ILCD Handbook (EC JRC, 2010a). Table 3.6 summarizes the correspondence.

**Table 3.6** *Classification of the different impact categories according to Recipe and Impact 2002+.*

| Midpoint impact category (ILCD Handbook) | IMPACT 2002+ | | ReCiPe 2008 | |
|---|---|---|---|---|
| | Category indicator | Unit | Category indicator | Unit |
| Climate change | Global warming | [kg $CO_2$ eq] | Climate change | [kg $CO_2$ eq] |
| Respiratory inorganics/Particulate matter | Respiratory inorganics | [kg PM2.5 eq] | Particulate matter formation | [kg PM 10 eq] |
| Photochemical ozone formation | Respiratory organics | [kg $C_2H_4$ eq] | Photochemical oxidant formation | [kg NMVOC] |
| Acidification | Terrestrial acidification/ nutrification | [kg $SO_2$ eq] | Terrestrial acidification | [kg $SO_2$ eq] |
| Land use | Land occupation | [$m^2$org.arable] | Agricultural land occupation | [$m^2$a] |
| | | | Urban land occupation | [$m^2$a] |
| Eutrophication | Aquatic eutrophication | [kg $PO_4$ P-lim] | Freshwater eutrophication | [kg P eq] |
| Resources consumption | Non-renewable energy | [MJ primary] | Fossil depletion | [kg oil eq] |

The results of the sensitivity analysis with Impact 2002+ confirms the results obtained with Recipe, therefore confirming the results of the comparative LCA.

### *3.3.3 Uncertainty analysis results*

The 4-step procedure for uncertainty analysis defined in §2.4.4 was applied to the case study in order to quantify parameter uncertainty. The first step consisted in the selection of the most significant inventory data, by the means of the contribution analysis at impact category level with a 1% cut-off. The results for climate change are reported in Table 3.7. The same analysis was done also for the other impact category.

**Table 3.7** Contribution analysis at midpoint level for Climate change.

| Process | Product A (%) | Product B (%) | Product C (%) |
|---|---|---|---|
| Electricity, medium voltage, at grid | 31.1 | 35.0 | 16.3 |
| Natural gas, burned in Mini CHP plant | 21.0 | 21.0 | 18.3 |
| CO2 emissions into air | 20.6 | 20.7 | 18.0 |
| Sulphate pulp, ECF bleached, at plant | 9.9 | - | - |
| Natural gas, burned in industrial furnace >100kW | 6.7 | 6.7 | 6.7 |
| Transport, transoceanic freight ship/OCE S | 2.0 | 1.5 | - |
| Transport, lorry >32t, EURO3/RER S | 1.6 | 1.4 | 2.2 |

| Process | Product A (%) | Product B (%) | Product C (%) |
|---|---|---|---|
| Oriented polypropylene film E | 1.2 | 1.2 | 3.1 |
| EUR-flat pallet/RER S | 1.1 | 1.1 | 2.6 |
| Disposal, plastics, mixture, 15.3% water, to municipal incineration | - | 5.6 | 11.7 |
| Melamine formaldehyde resin, at plant/RER S | - | - | 4.1 |
| Core board, at plant/RER S | - | - | 1.6 |
| Disposal, paper, 11.2% water, to sanitary landfill/CH S | - | - | 1.3 |
| Transport, municipal waste collection, lorry 21t/CH U | - | - | 5.2 |

Each of the inventory data listed in Table 3.7, as well as the other inventory data referring to the other selected impact categories was attributed a probability distribution, according to the Pedigree matrix approach. Input and output data from Ecoinvent database at unit level already incorporate lognormal probability distribution with their relative value of geometric standard deviation. The results of the calculation of the square of the geometric standard deviation for inventory data not including probability distribution, namely from other database and emissions and land use data, are reported in Table 3.8.

**Table 3.8** *Calculation of the square of the geometric standard deviation*

| *Inventory data* | *SDg95* |
|---|---|
| $NO_x$ emission into air | 1.218 |
| $CO_2$ emission into air | 1.096 |
| Diesel | 1.175 |
| Nitrogen total emission into water | 1.218 |
| Phosphorus, total emission into water | 1.218 |
| Water, well, in ground | 1.106 |
| Oriented polypropylene film E | 1.096 |

The third step of the procedure consisted in the quantitative uncertainty quantification by the means of Monte Carlo simulation with a stop criterion equal to 1000 runs.

Monte Carlo analysis is a numerical way to process uncertainty data and establish an uncertainty range in the calculation results.

Monte Carlo simulation was conducted comparing two products at a time. The results of the comparison between product A and B are reported in Figure 3.8 and Table 3.9; meanwhile the comparison between product B and C in reported in Figure 3.9 and Table 3.10 and finally the comparison between product A and C is reported in Figure 3.10 and Table 3.11.

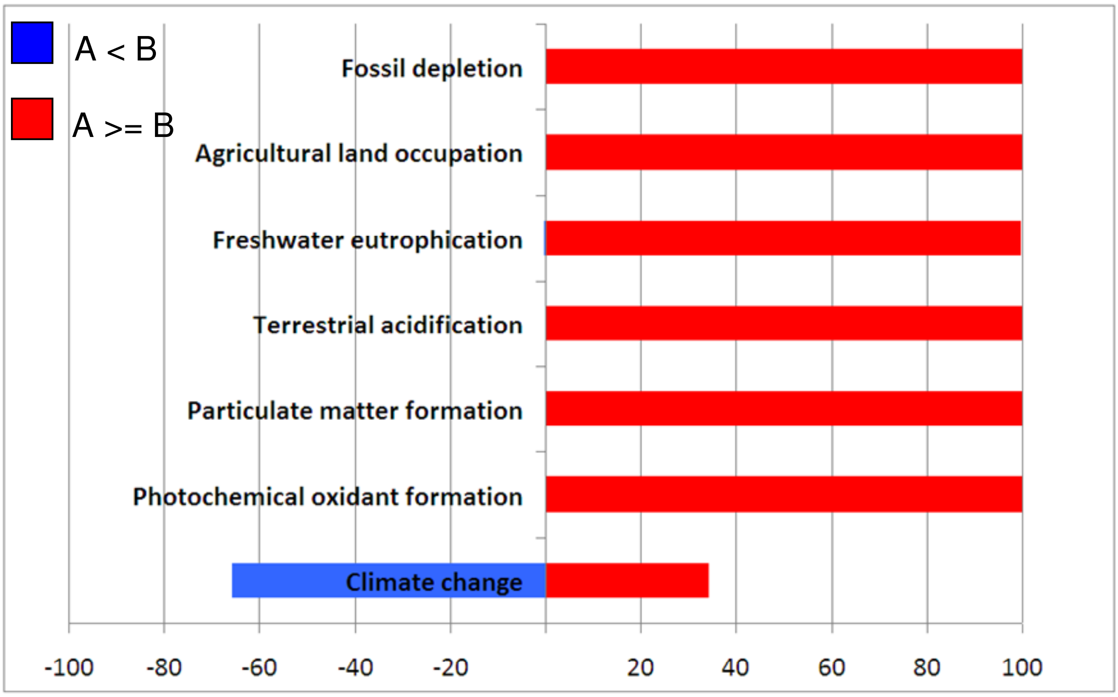


**Figure 3.8** *Results of the comparative Monte Carlo simulation between product A and product B.*

**Table 3.9** *Tabular representation of the difference of distribution between product A and B.*

| Impact category | Mean | Median | Standard deviation | 2.5% IC | 97.5% IC |
|---|---|---|---|---|---|
| Agricultural land occupation | 6.82 | 6.64 | 1.39 | 4.6 | 10 |
| Climate change | -2.79E-02 | -2.74E-02 | 6.41E-02 | -1.51E-01 | 9.64E-02 |
| Fossil depletion | 4.99E-02 | 4.96E-02 | 1.39E-02 | 2.25E-02 | 7.92E-02 |
| Freshwater eutrophication | 0.000103 | 8.36E-05 | 7.44E-05 | 2.65E-05 | 2.96E-04 |
| Particulate matter formation | 1.02E-03 | 9.98E-04 | 1.87E-04 | 7.22E-04 | 1.52E-03 |
| Photochemical oxidant formation | 2.10E-03 | 2.08E-03 | 4.11E-04 | 1.34E-03 | 2.95E-03 |
| Terrestrial acidification | 2.29E-03 | 2.24E-03 | 4.34E-04 | 1.68E-03 | 3.20E-03 |

The output of the comparative uncertainty analysis between product A and B show that for all impact categories except Climate change 100% of the Monte Carlo runs are favourable for product A, therefore the difference between the two products may be considered significant. For Climate change in the 65% of cases product A show less impact then product B, confirming that there is an equivalence between the two system products.

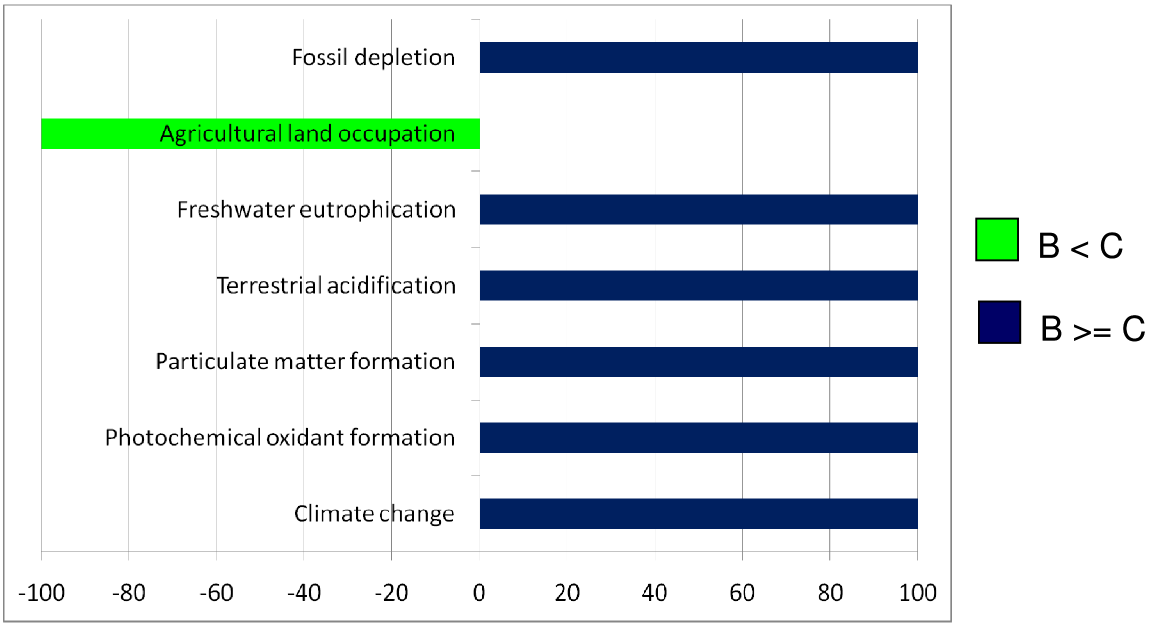


**Figure 3.9** *Results of the comparative Monte Carlo simulation between product B and product C.*

**Table 3.10** *Tabular representation of the difference of distribution between product B and C.*

| Impact category | Mean | Median | Standard deviation | 2.5% IC | 97.5% IC |
|---|---|---|---|---|---|
| Agricultural land occupation | -0.318 | -0.309 | 0.0779 | -0.502 | -0.191 |
| Climate change | 1.49 | 1.49 | 0.149 | 1.22 | 1.82 |
| Fossil depletion | 0.401 | 0.396 | 0.0632 | 0.294 | 0.549 |
| Freshwater eutrophication | 1.71E-04 | 1.41E-04 | 1.09E-04 | 5.33E-05 | 4.71E-04 |
| Particulate matter formation | 1.27E-03 | 1.26E-03 | 1.95E-04 | 9.37E-04 | 1.72E-03 |
| Photochemical oxidant formation | 3.36E-03 | 3.30E-03 | 5.01E-04 | 2.54E-03 | 4.53E-03 |
| Terrestrial acidification | 4.41E-03 | 4.37E-03 | 6.63E-04 | 3.27E-03 | 5.88E-03 |

The output of the comparative uncertainty analysis between product B and C shows that for all impact categories except Agricultural land occupation 100% of the Monte Carlo runs are favourable for product C, therefore the difference between the two products may be considered significant. The contrary is for Agricultural land occupation for which in 100% of cases product B show less impact then product C, confirming the results of the comparative LCA.

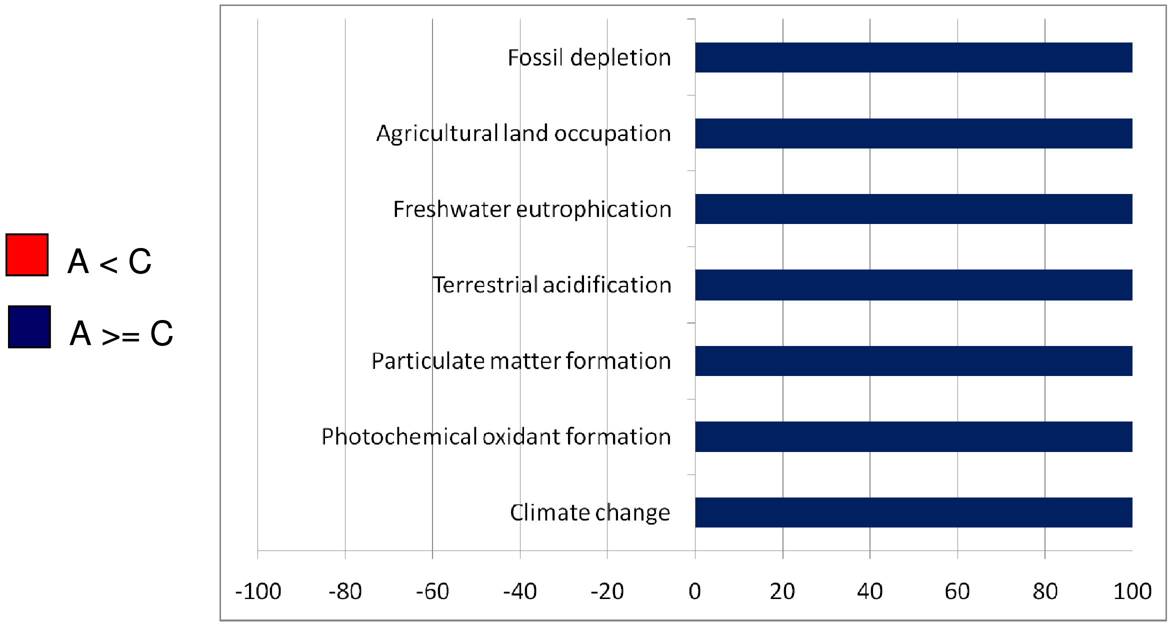


**Figure 3.10** *Results of the comparative Monte Carlo simulation between product A and product C.*

**Table 3.11** *Tabular representation of the difference of distribution between product A and C.*

| Impact category | Mean | Median | Standard deviation | 2.5% IC | 97.5% IC |
|---|---|---|---|---|---|
| Agricultural land occupation | 6.5 | 6.39 | 1.32 | 4.32 | 9.31 |
| Climate change | 1.46 | 1.46 | 0.137 | 1.21 | 1.75 |
| Fossil depletion | 0.451 | 0.448 | 0.0604 | 0.342 | 0.577 |
| Freshwater eutrophication | 2.73E-04 | 2.41E-04 | 1.47E-04 | 9.99E-05 | 6.19E-04 |
| Particulate matter formation | 2.30E-03 | 2.28E-03 | 2.64E-04 | 1.85E-03 | 2.88E-03 |
| Photochemical oxidant formation | 5.43E-03 | 5.39E-03 | 5.90E-04 | 4.37E-03 | 6.68E-03 |
| Terrestrial acidification | 6.70E-03 | 6.63E-03 | 7.10E-04 | 5.46E-03 | 8.32E-03 |

The output of the comparative uncertainty analysis between product A and C shows that for all impact categories 100% of the Monte Carlo runs are favourable for product C, therefore the difference between the two products may be considered significant. The results of the comparative LCA are therefore confirmed.

## 3.4 Conclusions

The influence of assumptions on the final results is very important when a company aims to compare the environmental performances of its products through Life Cycle Assessment. The ISO 14040-44 standards provide some techniques for enhance confidence in the results of an LCA study, such as sensitivity and uncertainty analysis. These techniques are not always taken into account within the interpretation phase, but if the goal of the LCA is to make an environmental claim regarding the superiority of one product versus another product with the same function, they should not be neglected, even if the products are manufactured by the same company.

LCA methodology was applied to compare the environmental impacts of three different types of tissue paper wipers manufactured with different raw materials: virgin pulp (product A), waste paper (product B) and recovered fibres from beverage carton containers (product C). A cradle to gate perspective was adopted, including all the processes from raw and auxiliary materials extraction, manufacturing, packaging of the final product, and waste management and transports. Therefore the focus was given to the paper manufacturing and a modelling process was developed in order to define specific resources consumption for the product system under study, as data were available at plant level.

The cut-off approach was taken into account with regard to the recycling of paper. ReCiPe method was used to perform impact assessment at midpoint level, considering the following impact categories: climate change, particulate matter formation, agricultural land occupation, fossil depletion, photochemical oxidant formation, terrestrial acidification, freshwater eutrophication

The results of impact assessment show that wipers made by virgin pulp have the greatest environmental impacts for all the impact categories considered, except for the climate change for which virgin fiber wipers and waste paper wipers have the same impact. The less impacting products are wipers made by recovered fibers from beverage carton. The main impacting life cycle stage is tissue production, which in the case of virgin pulp includes the chemical pulp production. The sensitivity analysis was performed changing the impact assessment methodology (Impact 2002+) and the results of the comparative LCA were confirmed.

The uncertainty analysis was performed using the 4-step procedure defined in Chapter 2, based on the Pedigree Matrix and the Monte Carlo algorithm, comparing two products at a time. The results of the uncertainty analysis showed that comparative LCA results are confirmed with a level of statistical significance of 100% for all impact categories and all combinations of comparison between products A, B and C. Only for climate change the level of statistical significance is lower, therefore confirming that for this impact category there is no actual difference between product A and B. The results of uncertainty analysis indicate that uncertainty intervals are useful in understanding the stability of results.

The results of both sensitivity and uncertainty analysis confirmed the LCA results, enhancing the confidence in the conclusions of the comparative LCA. Further work needs to be carried out in continuation of this study in order to improve data collection and test how the optimization of the production process can influence the environmental performances of the tissue paper products under study.

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# Chapter 4

# LCA of different technological solutions for wastewater treatment

This chapter[5] presents a case study, conducted during the internship at the Department of Environmental Science, Aahrus University. LCA methodology is applied to evaluate and compare the environmental performances of different technological solutions representative of Danish Waste Water Treatment Plants. Trade-off in the environmental performances are discussed in the context of expanded system boundaries to account for the avoided impacts achievable with: (1) sludge disposal through combustion with energy production and (2) agricultural sludge application. Uncertainty analysis with Monte Carlo simulation combined with the sensitivity analysis are used to improve the transparency and robustness of our findings.

## 4.1 Introduction

The main function of a Waste Water Treatment Plant (WWTP) is removing nutrients, metals and organic pollutants present in mixed household and industrial wastewater. The concentration of such components in wastewater effluents are reduced through biological degradation of organic pollutants, aqueous precipitation or sorption of metals and nutrients in the organic phase, and by preventing suspended solids and organic matter from being discharged to watercourses. The capacity of pollutants-removal and the environmental performance of WWTPs vary according to the influent wastewater characteristics, the technology, and the capacity of each WWTP. Apart from the main function, WWTPs provide also other secondary services and may thus be defined as multi-functional systems. The sludge generated as a co-product of the wastewater treatment process is increasingly seen as a resource rather than a waste material, and several end-of-life treatment options exist that allow recovering either energy or nutrients from sludge, depending on its quality. In particular, in recent times there has been a growing attention towards the recovery of phosphorus (P) from wastewater as P is an essential and non-substitutable nutrient mainly extracted from phosphate rock, a scarce resource (Cordell et al., 2011). In this context,

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WWTPs should be optimized for the provision of multiple services, but several trade-offs inevitably arise when choosing between different management options, as this leads to differences in the environmental performance of WWTPs.

The environmental performance of the wastewater treatment process can be determined by means of Life Cycle Assessment (LCA) (ISO 2006a, 2006b). LCA is a methodology for the assessment of the potential environmental impacts that a product/service generates over its entire life-cycle. There are many applications of LCA within the domain of wastewater treatment and some common issues can be identified. LCA has mainly been used to identify improvement alternatives for a single facility (Hospido et al., 2004; Pasqualino et al., 2009) as well as to compare different competing technology configurations (Coats et al., 2011, Gallego et al., 2008; Meneses et al., 2010). In some cases, the analysis is limited to the inventory level, without a quantification of potential environmental impacts (Lundin et al., 2000; Lundin et al., 2002; Foley et al., 2010). Previous studies, even if based on plant-specific data, rarely assess the influence of data variability on the final outcomes (Hospido et al., 2004). However, the environmental performance of WWTPs is usually not steady due to: changes in residual water flow and quality due to changes in the number of people served and industries connected to the collective sewer system, differences in population habits, and seasonal variations in the weather throughout the year (e.g. rainfall) (Meneses et al., 2010). Furthermore, previous studies tend to use secondary data to model effluent emissions (Foley et al., 2010) and an analysis at national scale is still lacking. As the performance of WWTPs highly depends on the number and types of industries connected to the sewer system even within the same plant type (technology and capacity), it is particularly important to conduct an analysis at national level. Another important issue in the LCA of WWTPs is that different methodological choices concerning the definition of system boundaries can largely influence the results (Lundin et al., 2000). Critical choices are the inclusion/exclusion of final sludge treatment and whether sewage sludge is regarded as a resource or a waste product. In this study we account for all the processes *"within and outside the life cycle that are affected by a change within the life cycle of the product under investigation"* (Ekvall and Weidema, 2004), i.e. we follow the principles of consequential LCA. Therefore, the end-of-life treatment of sludge is included within the system boundaries. Moreover, we apply system expansion, i.e. we account for the processes (and related impacts) outside the system boundaries that are avoided by the generation of co-products within the system.

The end-of-life treatment of sludge is one of the main contributors to improved environmental performance of WWTPs (Hospido et al., 2004; Tidåker et al., 2006). Many studies on sustainable sludge treatment focus on the recovery and useful reuse of the valuable components of sludge, i.e. nutrients (phosphorus and nitrogen) and carbon (Hospido et al., 2005; Lederer and Rechberger, 2010; Johansson et al., 2008; Linderholm et al., 2012; Suh and Rousseaux, 2002). Other studies discuss various options to recover energy from sewage sludge, such as anaerobic digestion and incineration with energy recovery (Rulkens, 2008). Thus, agricultural sludge application and incineration of dewatered sludge (or of residual digestate from biogas production) are two of the most debated solutions, both giving advantages and disadvantages. Agricultural sludge application substitutes for the production and use of mineral fertilizers reducing the depletion of virgin resources such as mineral phosphorus extracted from phosphate rock (Cordell et al., 2011). As such, when sludge is recycled back to agricultural soils, the substitution of mineral fertilizers according to the phosphorus content can be accounted as a benefit for the system. However, the application of sludge to agricultural soil results in a change in environmental performance compared to the equivalent application of conventional fertilizers. The difference between applying sludge rather than fertilizer to soil depends on the fertilizer quality, e.g. with respect to content of heavy metals. In Denmark, big centralized WWTPs are often equipped with anaerobic digesters for the production of biogas from sludge. The energy generated from biogas combustion can be used internally in the WWTPs, thus reducing their energy demand from external sources, e.g. from the electricity grid. For centralized plants, incineration is a mainstream option for the end-of-life treatment of the residual sludge, given the low quality of this product. The advantage of incineration is the production of energy, whereas the main drawback is the emission of toxic substances into the environment.

In this context, the objective of our study is to evaluate and compare the environmental performances of different technological solutions representative of Danish WWTPs. We apply system expansion and account for the avoided impacts of the end-of-life treatment of sludge. Moreover, we evaluate the influence of effluent wastewater variability, treatment capacity and technology on the final LCA outcomes.

## 4.2 Materials and methods

We performed a comparative LCA of four different technological solutions for Danish WWTPs, in accordance to the methodology defined by the international standard ISO 14040-44 (ISO 2006a, 2006b). The LCA consists of four steps: (1) goal and scope definition, (2) life cycle inventory (LCI), (3) life cycle impact assessment (LCIA) and (4) life cycle interpretation. In the first step the goal and scope, the product system, the functional unit, the system boundaries, and the level of detail of the LCA -that depend on the subject and the intended use of the study- are defined unambiguously. The LCI is an inventory of input/output data for the system being studied. The purpose of the LCIA is to understand the potential environmental impacts of the system, given the LCI results. Life cycle interpretation is the final phase of the LCA, in which the results of both LCI and LCIA are discussed as a basis for conclusions, recommendations and decision-making in accordance with the goal and scope of the study.

We grouped the existing Danish WWTPs into four plant types representative of an actual scenario, i.e. we consider state-of-the-art Danish WWT technologies. Two criteria were used: (1) plant-capacity measured in terms of PE (Person Equivalents); (2) production of biogas from sludge (anaerobic vs. aerobic sludge treatment). This last criterion was chosen based on the hypothesis that the final sludge quality varies between aerobic and anaerobic WWTPs; this is due to a reduced mass of organic carbon in digestate from biogas production and therefore increased concentration of heavy metals and persistent organic pollutants retained in the sludge. The four groups are summarized in Table 4.1.

**Table 4.1** *Categories of WWTP types included in the actual scenario*

| Plant type | Type 1 | Type 2 | Type 3 | Type 4 |
|---|---|---|---|---|
| PE | < 20,000 | > 20,000; <100,000 | > 20,000; <100,000 | > 100,000 |
| Biogas production | None, Aerobic sludge treatment | None, Aerobic sludge treatment | Yes, Anaerobic sludge treatment | Yes, Anaerobic sludge treatment |
| Final sludge disposal | Agriculture | Agriculture | Incineration | Incineration |

In Table 4.1, Type 1 represents small decentralized plants with no biogas production and agricultural sludge application as sludge end-of-life option. Type 2 is similar to type 1, except for size category. Type 3 and 4 represent WWTPs that perform anaerobic sludge treatment and therefore biogas production, both using sludge incineration as final sludge disposal. Type 4 represents big and centralized WWTPs.

### *4.2.1 Goal and scope definition*

The aim of the study is to assess and compare the potential environmental performance of the four different technological solutions for Danish WWTPs, as described above, which are representative of the main WWT technologies implemented in Denmark. The study should provide guidance in the management of Danish WWTPs, in particular by highlighting the trade-offs of different management options, taking into account the sludge and effluent wastewater quality for the different technology types.

Concerning scope definition, the product system under study is the Danish WWTP. The main function of the system is the treatment of organic rich and polluted inflow wastewater by production of purified effluent wastewater, complying with the quality criteria of the Water Framework Directive (EC, 2000). Therefore the functional unit (FU) is defined as the treatment of 1 $m^3$ of inlet wastewater.

The inputs/outputs, system boundaries, and system expansions for the multifunctional WWT system are shown in Figure 4.1. In accordance with previous LCA studies on WWT (Foley et al., 2010, Hospido et al., 2004, Lundin et al., 2000, Pasqualino et al., 2009), only the operational stage was considered. Because of the comparative nature of the study, the infrastructure or dismantling of buildings or equipment, as well as the water distribution system, were not considered as part of the system. In accordance with Lundin et al. (2000), the use of chemicals and energy were included within the system boundaries. Also, the type of sludge treatment (anaerobic vs. aerobic) was included, as it influences the energetic performance of the plant as well as the final quality of the sludge. The main co-products included in the system are biogas and sludge. Due to its P content, sludge is considered as resource that can be recycled to agricultural soils. Thus, the two end-of-life treatment options for sludge - included within the system boundaries - are incineration and agricultural application (Foley et al., 2010; Pasqualino et al., 2009). We excluded external biogas production in other treatment plants, as external biogas production constitute less than 2% of the total biogas production and to avoid double counting. Landfill constituted less than 1% in 2005 (DEPA, 2009) and for the plants selected for this study, landfill constitutes less than 0.3 % and only applies to two of the four plant types. The system expansion processes are avoided electricity and heat production, and production and use of fertilizer; these are described more in detail in the following sections.

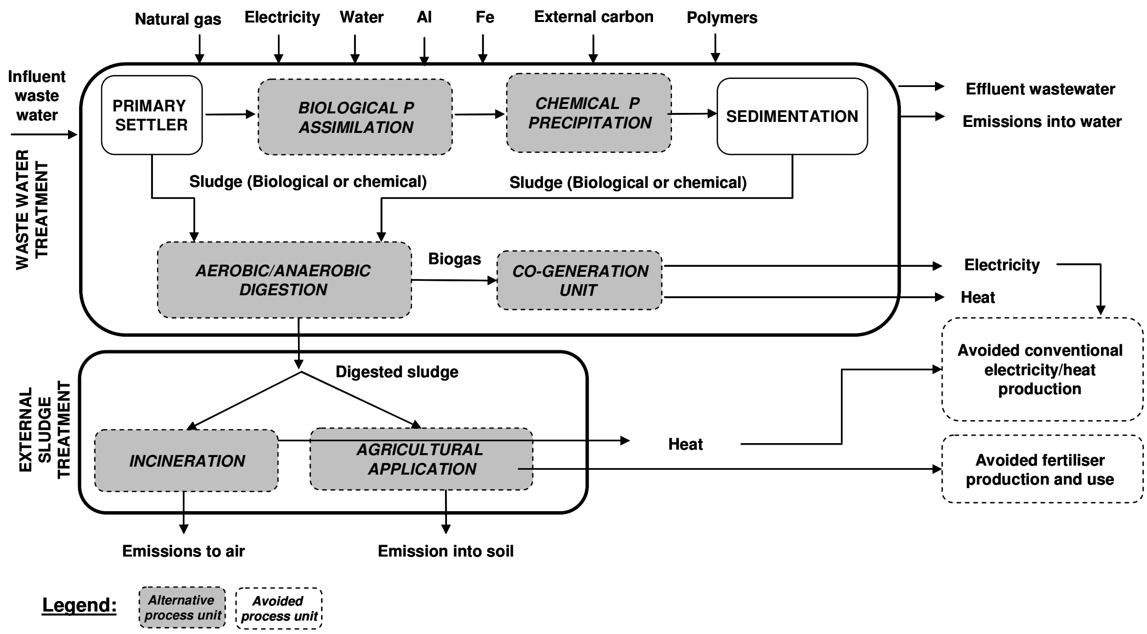


**Figure 4.1.** *Inputs, outputs, system boundaries, and system expansions of the multifunctional WWT system.. Al refers to aluminum chloride and Fe refers to iron chloride used as chemicals. The "Alternative process" units (gray-shaded boxes) refer to processes that are taking place or not depending on the kind of treatment performed by each plant type. The process "wastewater treatment" is modeled as a "black-box" process (input-output inventory data are aggregated and refer to the process as a whole and not to the individual sub-unit processes).*

## 4.2.2 Life Cycle Inventory

### 4.2.2.1 Energy balance and chemicals consumption

Energy and chemical consumption data refer to plant-specific operating data. We chose a single representative plant for each of the four types: Ulstrup for type 1, Solrød for type 2, Hillerød for type 3, Damhusåen for type 4. The input/output data on use of energy and chemicals for each type of WWTP are summarized in Table 4.2. These data refer to a three years interval taken from the most updated green accounts publicly available (Favrskov Forsyning A/S, 2011; Favrskov Forsyning A/S, 2012; Solrod Kommune, 2006; Lynettefællesskabett I/S, 2008; Lynettefællesskabett I/S, 2009; Lynettefællesskabett I/S, 2010; Hillerød Kommune, 2009). Green accounts are reports issued yearly by WWT companies with detailed information about their environmental performances, as well as a list of the main input and output of the plant. We calculated average value and standard deviation referred to the functional unit, 1 $m^3$ inlet wastewater (cf. Table 4.2).

Chemical consumption includes iron chloride (in Figure 4.1 denoted Fe), aluminum chloride (in Figure 4.1 denoted Al), polymers and molasses (in Figure 4.1 denoted as External carbon). Regarding the energy input, we included the electricity consumption from the grid as well as the consumption of natural gas burned at the WWTP. For those plants producing biogas from anaerobic digestion (namely type 3 and type 4), the amount of biogas produced and consumed internally as heat (including only impacts deriving from its use) has been included. Regarding system expansion we assumed that producing heat and electricity at the WWTP avoids the impacts of producing heat (for district heating) and electricity with the current Danish energy mix, which includes both electricity domestic net production and import shares from neighboring countries (Dones et al., 2007).

As may be observed from Table 4.2, mean values on energy consumption are similar for the four plant types, meanwhile there are some discrepancies in terms of chemicals consumption, e.g. standard deviation connected with ferric chloride consumption presents the highest value. The use of ferric chloride (Fe) and/or aluminum chloride (Al) is higher for plant type 1 compared to type 2, as type 2 has implemented an advanced technology for biological phosphorous removal (BioP technology) (Blackall et al., 2002). Plant type 3 has a well-functioning BioP technology implemented and less use of chemicals compared to plant type 4.

**Table 4.2** *Energy and chemicals consumption for the four plant types*

| Inventory data | Units | Type1 | | Type2 | | Type3 | | Type4 | |
|---|---|---|---|---|---|---|---|---|---|
| Reference years | / | 2009-2011 | | 2005-2007 | | 2006-2008 | | 2007-2009 | |
| Value | / | μ | σ | μ | σ | μ | σ | μ | σ |
| Natural gas consumption | kWh /m$^3$ | 0 | / | 0 | / | 1.00E-02 | 1.47E-03 | 2.29E-02 | 2.32E-03 |
| Electricity consumption | kWh/m$^3$ | 3.13E-01 | 8.82E-02 | 4.45E-01 | 6.57E-02 | 3.24E-01 | 2.84E-02 | 3.27E-01 | 6.19E-02 |
| Biogas | kWh/m$^3$ | 0 | / | 0 | / | 3.58E-01 | 2.30E-04 | 2.10E-01 | 5.71E-05 |
| Water | kg/m$^3$ | 6.63E-01 | 2.67E-01 | 4.40E-01 | 1.78E-01 | 35.78 | 4.18 | 1.05E-01 | 5.12E-02 |
| Al[1] | kg/m$^3$ | 0 | / | 7.26E-03 | 5.88E-03 | 1.69E-02 | 1.54E-02 | 2.23E-02 | 2.38E-02 |
| Fe | kg/m$^3$ | 6.06E-02 | 4.44E-02 | 0 | / | 1.52E-02 | 5.06E-03 | 4.62E-02 | 1.63E-02 |
| External carbon | kg/m$^3$ | 0 | / | 0 | / | 3.38E-03 | 7.77E-04 | 0 | / |
| Polymers | kg/m$^3$ | 1.39E-03 | 4.14E-04 | 4.28E-03 | 5.97E-04 | 3.30E-03 | 3.00E-04 | 2.20E-03 | 1.00E-03 |
| Electricity production | kWh/m$^3$ | 0 | / | 0 | / | 0 | / | 1.36E-01 | 3.70E-05 |
| Heat production | kWh/m$^3$ | 0 | / | 0 | / | 8.95E-02 | 1.33E-05 | 5.77E-02 | 1.87E-05 |

[1] modeled as aluminum sulphate, as the specific dataset was not available

### 4.2.2.2 Emissions into surface water/wastewater effluent quality

The data source is the National Danish water quality parameter database held by the Danish Environmental Protection Agency (www.miljoeportalen.dk). Data refer to the three years interval 2005-2007 for which a complete set of wastewater influent and effluent monitoring data was available. Corresponding sludge quality data were available from the sludge database. Once we collected plant-specific effluent water quality data from the database we grouped them into plant types defined in Table 4.1. Then we looked at the statistical distribution of wastewater effluent data and we found that they were log-normally distributed, finding a significant difference according to plant type. Therefore, we calculated the geometric mean and geometric standard deviation for each wastewater effluent parameter, and calculated the emissions into water with reference to the system's FU. Inventory data used in the comparative LCA of the four plant types are reported in Table 4.3.

**Table 4.3** *Emissions into surface water from wastewater (geometric average and geometric standard deviation)*

| Emission into surface water from wastewater (kg/m$^3$ inlet water) | Type 1 | | Type 2 | | Type 3 | | Type 4 | |
|---|---|---|---|---|---|---|---|---|
| | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ |
| Suspended solids | 7.49E-03 | 1.87 | 4.91E-03 | 1.63 | 6.37E-03 | 1.80 | 7.26E-03 | 2.17 |
| Non-Methane Volatile Organic Compounds | 0 | / | 0 | / | 1.03E-02 | / | 0 | / |
| BOD | 2.61E-03 | 1.51 | 2.66E-03 | 1.40 | 2.52E-03 | 2.11 | 3.79E-03 | 1.45 |
| COD | 2.54E-02 | 1.46 | 2.62E-02 | 1.32 | 3.04E-02 | 1.44 | 3.27E-02 | 1.41 |
| Ammonia. as N | 3.92E-04 | 3.35 | 2.97E-04 | 2.47 | 5.44E-04 | 3.61 | 6.29E-04 | 1.97 |
| Nitrate | 2.78E-03 | 3.26 | 1.57E-03 | 1.72 | 2.02E-03 | 2.09 | 2.43E-03 | 2.54 |
| Total-N | 6.20E-03 | 2.09 | 3.25E-03 | 1.46 | 4.07E-03 | 1.47 | 5.49E-03 | 1.34 |
| Total-P | 5.03E-04 | 2.18 | 3.11E-04 | 1.72 | 3.55E-04 | 1.92 | 5.09E-04 | 1.63 |
| Lead | 8.46E-07 | 1.35 | 1.27E-06 | 1.84 | 9.82E-07 | 1.94 | 1.51E-06 | 1.84 |
| Cadmium | 5.00E-08 | 1.00 | 1.01E-07 | 1.80 | 5.36E-08 | 1.02 | 6.51E-08 | 1.38 |
| Copper | 2.54E-06 | 1.70 | 4.06E-06 | 2.51 | 4.40E-06 | 1.63 | 4.73E-06 | 2.58 |
| Chrome | 1.36E-06 | 2.17 | 2.01E-06 | 3.20 | 9.76E-07 | 1.72 | 1.55E-06 | 2.24 |
| Mercury | 1.11E-07 | 2.52 | 8.35E-08 | 1.73 | 7.07E-08 | 1.49 | 6.69E-08 | 2.91 |
| Nickel | 5.70E-06 | 3.07 | 4.18E-06 | 2.02 | 3.08E-06 | 2.53 | 5.24E-06 | 2.08 |
| Zinc | 4.28E-05 | 1.67 | 5.54E-05 | 1.67 | 4.70E-05 | 2.23 | 4.99E-05 | 1.56 |

As shown in Table 4.3, there are no high discrepancies in terms of mean values among the plant types. The highest variability is connected with heavy metals emissions as well as ammonia emission which therefore are expected to have the highest influence the output of the comparative LCA study. Plant type 4 is the one with the overall highest emissions to surface water, except for suspended solids, nitrate and total N and mercury for which the highest emissions are for plant type 1 and cadmium, chrome and zinc for which the highest emissions are for plant type 2, respectively.

Only plants that have reported to the Danish sludge database as well as the water quality parameter database have been included (cf. section 4.2.2.3). Based on these restrictions, measurements of nonylphenols/-ethoxylates, phthalates or Linear Alkylbenzene Sulfonate (LAS) were incomplete; measurements on Di (2-ethylhexyl) phthalate (DEHP) were not covering for all four plants type, whereas measurements were missing for all plant types in case of the two detergents. Therefore, these pollutants were eliminated from the waste effluent inventory data.

#### 4.2.2.3 Final sludge disposal

Data on sludge quality are available for approximately one third of the WWTPs in Denmark in the three years interval 2005-2007. We calculated sludge quality data for the four WWTP types according to the same methodology described in section 4.2.2.2. For the sludge quality we did not differentiate between sludge disposal categories agricultural soil, forest, park, public and private gardens as the same sludge quality criteria apply to these disposal categories (EC, 1986).

We modeled the agricultural sludge application by considering the concentration of the pollutants As, Pb, Cd, K, Cu, Hg, Ni, Zi, Total N, Total P and N-nonylphenol in the sludge as measure of direct emissions to soil. Sludge quality is expressed in terms of gram pollutant per kg dry matter (DM) content in the sludge (Table 4.4).

**Table 4.4** *Emissions from sludge application into soil (geometric average and geometric standard deviation)*

| **Emission into soil from sludge (g /kg DM sludge)** | **Type1** | | **Type2** | | **Type3** | | **Type4** | |
|---|---|---|---|---|---|---|---|---|
| | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ | $\mu_g$ | $\sigma_g$ |
| Lead | 3.27E-02 | 1.70 | 2.92E-02 | 1.78 | 4.97E-02 | 3.33 | 4.49E-02 | 1.80 |
| Cadmium | 9.72E-04 | 1.66 | 1.09E-03 | 2.30 | 1.82E-03 | 2.70 | 1.36E-03 | 1.55 |
| Copper | 1.83E-01 | 1.63 | 1.97E-01 | 1.66 | 3.75E-01 | 1.59 | 2.38E-01 | 1.46 |
| Mercury | 5.87E-04 | 2.36 | 7.69E-04 | 2.28 | 1.16E-03 | 1.87 | 1.18E-03 | 1.79 |
| Nickel | 2.14E-02 | 1.54 | 2.09E-02 | 1.85 | 2.30E-02 | 2.31 | 2.80E-02 | 1.40 |
| Zinc | 6.20E-01 | 1.72 | 5.48E-01 | 1.52 | 8.23E-01 | 1.27 | 8.35E-01 | 1.52 |
| Total N | 4.04E+01 | 1.67 | 4.83E+01 | 1.46 | 3.45E+01 | 2.04 | 4.22E+01 | 1.18 |
| Total P | 2.59E+01 | 1.74 | 2.74E+01 | 1.52 | 2.84E+01 | 2.00 | 3.70E+01 | 1.18 |
| N-nonylphenol | 1.69E-03 | 2.30 | 1.85E-03 | 2.16 | 7.73E-03 | 2.50 | 1.04E-02 | 1.98 |

As may be observed from Table 4.4, measurements of N-nonylphenol concentration in sludge were available for all four plants types and were therefore included in the inventory even though data were missing for effluent wastewater.

Regarding system expansion, we assumed that the agricultural application of sludge replaces both the production and the use of conventional mineral fertilizer. We calculated the substitution rate, with reference to the P content, according to Equation 4.1:

$$\text{Substitution rate} = C_{p,sludge} / C_{p,\,ref} \tag{4.1}$$

Where $C_{p,sludge}$ is the concentration of phosphorus in sludge and $C_{p,ref}$ is the P content in mineral fertilizer. When modeling the avoided production of mineral fertilizer, we chose diammonium phosphate (DAP) as reference mineral fertilizer with 20,08% P content (46% $P_2O_5$ content) produced from ammonia and phosphoric acid (Nemecek and Kägi, 2007). As reliable data on the bioavailable fraction of P are not available, we assumed that 100% of the sludge enriched P is easily bioavailable and equal to unity. DAP production includes chemicals and energy consumption, as well as use of raw materials, transport, and emission into air and water, as reported by Nemecek and Kägi (2007). We modeled the avoided use of fertilizer as avoided emissions of heavy metals (As, Cd, Cr, Pb, Hg, Ni) based on the average heavy metals content of Danish mineral fertilizers, as reported by Petersen et al. (2009). The emission of pollutants into soil (Table 4.4) associated with the agricultural application of sludge were calculated according to the concentration of pollutants in sludge, per kg phosphorous. Furthermore, the emissions of total N and total P into soil were also considered as avoided emission into soil, based on the N content (18%) and P content (20,08%) of the reference mineral fertilizer.

No plant specific data were available for modeling sludge incineration, as no dedicated sludge incineration takes place in Denmark and different incineration plants use indeed a mix of feeding materials. Therefore we chose to use the “Disposal, digester sludge, to municipal incinerator” unit process from Ecoinvent database (Jungbluth et al., 2007) providing waste-specific air and water emissions from incineration, auxiliary material consumption for flue gas cleaning, as well as short-term emissions to river water and long-term emissions to ground water from slag compartment (from bottom slag) and residual material landfill (from solidified fly ashes and scrubber sludge). Process energy demands for Municipal Solid Waste Incineration (MSWI) are included. In accordance with the system expansion approach adopted in the study, we assumed that energy recovery from sludge incineration equals 0.74 MJ/kg wet sludge (Rambøll Danmark, 2008), which has been modeled as avoided production of heat for district heating.

### *4.2.3 Life Cycle Impact Assessment*

For LCIA we used the Recipe method (Goedkoop et al., 2009) at midpoint level. As wastewater treatment concerns mainly two types of environmental issues, namely climate mitigation and environmental quality issues, we focused on the following impact categories:

- climate change (kg CO2eq.);
- fossil depletion (kg oil eq.);
- human toxicity (kg 1,4-DB eq.);
- terrestrial, freshwater and marine ecotoxicity (kg 1,4-DB eq.);
- freshwater eutrophication (kg P eq.) and marine eutrophication (kg N eq.).

Climate change (CC), in kg $CO_2$ equivalents (eq.), accounts for the global warming potential of all greenhouse gases calculated within a time horizon of 100 years according to the method developed by the Intergovernmental Panel on Climate Change (IPCC).

Fossil depletion (FD), in kg oil eq., accounts for the extraction of fossil fuels. The characterization factor of fossil depletion is based on the upper calorific value of 42 MJ per kg for reference resource “Oil, crude, feedstock” (Hischier et al., 2009).

The characterization factors for human toxicity and ecotoxicity account for the environmental persistence (fate), exposure to humans and other organisms, and toxicity (effect) of a chemical measured in units of kg 1,4-dichlorobenzene equivalents (1,4-DB eq.). Fate and exposure factors can be calculated by means of ‘evaluative’ multimedia fate and exposure models, while effect factors can be derived from toxicity data on human beings and laboratory animals. In the Recipe method, the Uniform System for the Evaluation of Substances (USES-LCA 2.0) developed by Van Zelm et al. (2009) and adapted for LCA purposes (Huijbregts et al., 2000) is used to calculate default environmental fate and exposure factors. USES-LCA 2.0 differentiates between multiple environmental compartments and accounts for human intake via inhalation and ingestion on an infinite time horizon (Van Zelm et al., 2009). Impact categories covered by Recipe are: human toxicity (HT) including the following human exposure routes: inhalation, ingestion via root crops, leaf crops, meat products, dairy products, eggs, freshwater fish, marine fish and drinking water; terrestrial ecotoxicity (TET) for the soil ecosystem, freshwater ecotoxicity (FET) related to wastewater effluents and agricultural runoff and marine ecotoxicity (MET) related to air deposition as well as effluents and surface runoff transported to the marine environment. Aquatic eutrophication is the nutrient enrichment of the marine (therefore marine eutrophication, ME) and freshwater (therefore freshwater eutrophication, FE) aquatic environment. The characterization factor of ME is measured in units of kg N eq. and covers agricultural surface runoff, wastewater effluents as well as airborne emission of $NH_3$ and $NO_2$. The characterization factors of FE account for the environmental fate of phosphorous emissions to freshwater and the reference unit of this impact category is kg P eq..

### 4.2.4 Life Cycle Interpretation

We performed a sensitivity analysis consisting of two parts:

- We tested how alternative options for end-of-life treatment of sludge influence the environmental performance of the different plant types.
- We tested how the choice of the LCIA method influences results by applying alternative LCIA methods as recommended in the ILCD Handbook (EC-JRC, 2011; Hauschild et al., 2012).

In the first part (cf. section 4.3.1) we considered two alternative scenarios for plant types with anaerobic digestion, defined as type 3B and type 4B, respectively. For these types we assumed agricultural sludge application to be the final disposal option, instead of incineration (Table 4.5). The objective is to understand whether this solution is favorable in the case of big centralized plants performing anaerobic sludge treatment.

**Table 4.5** *Sludge production, substitution rate calculated with Eq. 4.1 and sludge end-of-life treatment for the actual and alternative scenarios.*

| | Units | Type1 | Type2 | Type3 | Type4 | Type3B | Type4B |
|---|---|---|---|---|---|---|---|
| Sludge production | ton/$m^3$ | 1.02E-04 | 2.27E-04 | 1.37E-04 | 1.60E-04 | 1.37E-04 | 1.60E-04 |
| Substitution rate | kg fertilizer/kg DM sludge | 1.29E-01 | 1.36E-01 | 0 | 0 | 1.41E-01 | 1.84E-01 |
| Agricultural application | % | 100 | 100 | 0 | 0 | 100 | 100 |
| Incineration | % | 0 | 0 | 100 | 100 | 0 | 0 |

In the second part (cf. section 4.3.2) we performed an alternative LCIA by using the LCIA methods (at midpoint level) that are recommended in the International Reference Life Cycle Data System Handbook (EC-JRC, 2011) published by the Joint Research Centre (JRC) of the European Commission. For the CC and aquatic eutrophication impact categories, the models recommended by the handbook are those implemented by Recipe, and that we already used as default. These are the baseline model of 100 years of the IPCC (Forster et al. 2007) for climate change and the EUTREND model (Struijs et al., 2009) for aquatic eutrophication which includes both FE and ME.
However, for HT, FET and abiotic resources depletion the ILCD handbook does not recommends using Recipe. Instead, the USEtox nested multimedia model (Rosenbaum et al., 2011) is recommended for HT, which differentiates between the contribution of cancer and non-cancer effect, therefore HT – cancer and HT – non cancer and FET impact categories. For abiotic resource depletion, the ILCD Handbook recommends the CML 2002 (Guinee et al., 2002) method. This impact category is related to extraction of minerals and use of fossil fuels in the system.
In addition to the sensitivity analysis, we performed an uncertainty analysis (cf. section 4.3.3) for the model outputs according to variability of inventory data provided in Table 4.2, 4.3 and 4.4. We applied Monte Carlo sampling technique in order to assess the robustness of LCIA according to variability of inventory data. Uncertainty analysis was conducted using a 1000-run Monte Carlo analysis in the Simapro software (Frischknecht et al., 2007) for all WWTP scenarios. We assumed normal distribution for energy and chemicals use data, calculating the arithmetic mean and standard deviation, as reported in Table 4.2, while data on emissions into water and soil were lognormal and we calculated geometric mean and geometric standard deviation as reported in Table 4.3 and 4.4, respectively.

## 4.3 Results and discussion

### *4.3.1 Life Cycle Impact Assessment (Recipe method)*

Table 4.6 reports the ranking in the environmental performance of the different plant types according to each LCIA category. We highlight that in the context of consequential LCA positive values of performance represent environmental impacts (thus, a potential environmental burden or damage) and negative values represent avoided impacts (thus, a sort of potential environmental "benefit"). No univocal recommendation is obtained as results change according to the different impact categories.

**Table 4.6.** *Life Cycle Impact Assessment results per plant type for the actual and alternative scenarios*

| Impact category | Units | Type1 | Type2 | Type3 | Type4 | Type 3B | Type 4B |
|---|---|---|---|---|---|---|---|
| Climate change (CC) | kg $CO_2$ eq. | 1.95E-01 | 2.13E-01 | 2.01E-01 | 1.34E-01 | 1.80E-01 | 9.98E-02 |
| Fossil depletion (FD) | kg oil eq. | 5.91E-02 | 6.64E-02 | 5.68E-02 | 3.81E-02 | 5.15E-02 | 2.93E-02 |
| Freshwater eutrophication (FE) | kg P eq. | 3.56E-04 | -1.63E-04 | 4.40E-04 | 5.82E-04 | 8.88E-05 | 5.09E-05 |
| Marine eutrophication (ME) | kg N eq. | 7.27E-03 | 3.93E-03 | 5.15E-03 | 6.75E-03 | 5.10E-03 | 6.70E-03 |
| Human toxicity (HT) | kg 1.4-DB eq. | 7.59E-02 | 4.67E-02 | 7.05E-02 | 6.80E-02 | 5.68E-02 | 4.86E-02 |
| Terrestrial ecotoxicity (TET) | kg 1.4-DB eq. | 7.19E-04 | 1.54E-03 | 5.04E-05 | 3.19E-05 | 1.57E-03 | 1.44E-03 |
| Freshwater ecotoxicity (FET) | kg 1.4-DB eq. | 1.46E-03 | 1.41E-03 | 1.85E-03 | 1.62E-03 | 1.74E-03 | 1.38E-03 |
| Marine ecotoxicity (MET) | kg 1.4-DB eq. | 1.12E-03 | 6.42E-04 | 1.71E-03 | 1.54E-03 | 1.14E-03 | 6.50E-04 |

According to the actual scenario big plants equipped with anaerobic sludge digestion (type 3 and type 4) perform better than small aerobic plants for the CC, FD, and TET impact categories, whereas for eutrophication (FE and ME) and other toxicity-related categories (HT, FET, MET) they show higher environmental impacts. These results are disaggregated and explained in detail in following sub-sections through a contribution analysis at impact category level.

#### 4.3.1.1 Climate mitigation

For the climate mitigation impact categories (CC and FD) the main contribution to the impact for all plant types is internal electricity consumption as visualized in Figure 4.2 and 4.3.

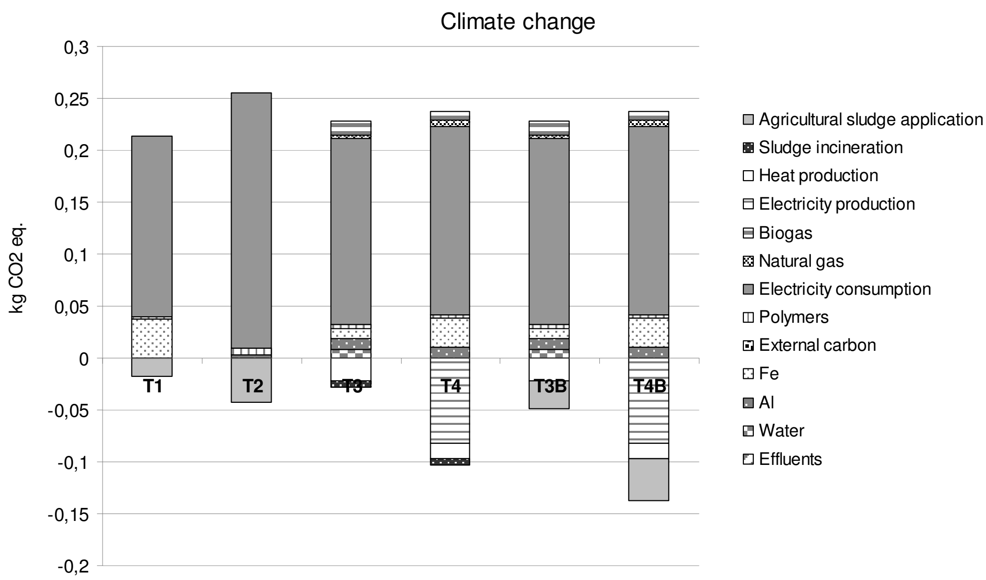


**Figure 4.2.** *Visualisation of process-specific contributions to the impact category climate change (CC), measured in kg CO2-eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

It should be mentioned that contribution to climate change, visualized as positive emissions, does not differentiate between fossil and bio-based energy consumption. In other words, green energy production and consumption occurring inside the plant count as a positive value while export of green energy, and i.e. external consumption, counts as a negative emission assumed to replace and reduce fossil energy consumption.

Plant type 1 and 2 have negative emissions corresponding to the avoided fertilizer production due to recirculation of N and P to agricultural soil. Similar negative emissions would occur upon realization of alternative scenarios for plant type 3 and 4 (T3B and T4B in Figure 4.2). Plant type 4 has both negative heat and electricity production while type 3 export green energy in terms of heat production. Both plant type 3 and 4 have a negative contribution to $CO_2$ emission from waste to energy production by sludge incineration.

In summary, plant type 4 has the overall lower impact due to: (1) the negative contribution of avoiding fossil-based energy production when assuming that exported electricity and heat produced by the plant substitutes fossil fuel consumption; (2) the negative contribution of energy recovery from sludge sent to incineration by WWTP types 3 and 4 again assuming that an equal amount of fossil energy depletion is avoided.

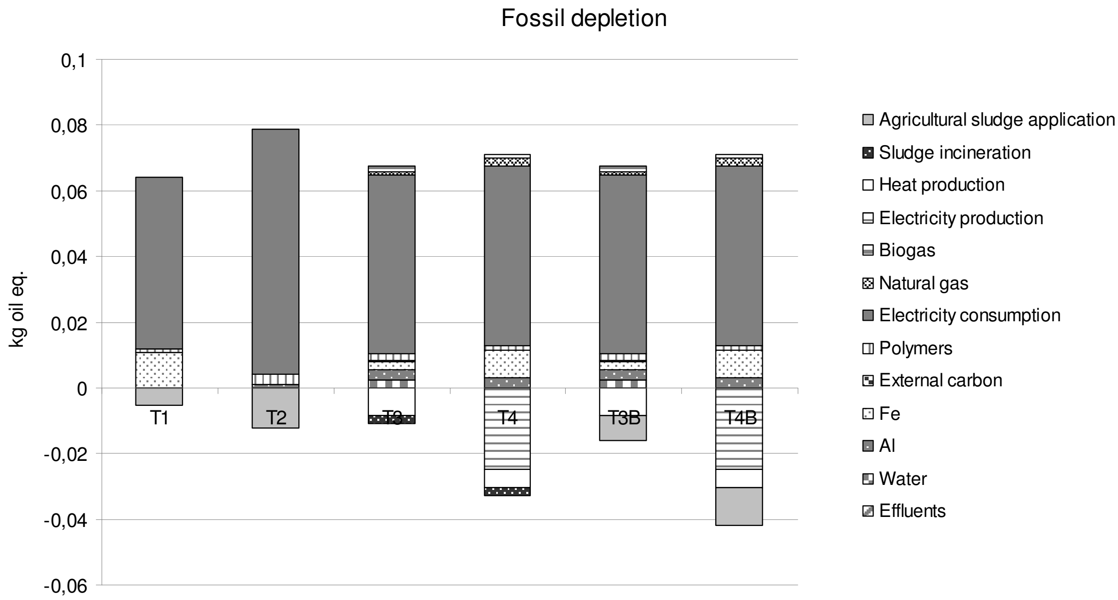


**Figure 4.3.** *Visualisation of process-specific contributions to the fossil depletion (FD) impact category, measured in kg oil-eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

Figure 4.3 shows a positive contribution of internal electricity consumption (for all plant types) and ferric chloride for plant type 1, 2 and 4. A negative contribution for plant type 1 and 2, and also for the alternative scenarios (T3B and T4B, Figure 4.3), is due to the avoided emissions from fossil fuels used during fertilizer production. Type 3 includes avoided emissions from heat production, meanwhile type 4 includes also a negative contribution from

internal electricity production. For both type 3 and 4 there is a negative contribution to fossil depletion due to the recovered energy from sludge incineration.
The production of chemicals, particularly ferric chloride, used by plant types 1, 3 and 4, shows a not negligible positive contribution to fossil depletion. In term of avoided resource depletion by increased self-supply, the alternative scenarios (T3B and T4B, figure 4.3) show the highest energy and fertilizer recycling efficiency compared to plant type 1 and 2. Furthermore, by comparing the actual versus alternative scenarios (T3 with T3B, and T4 with T4B, Figure 4.3) we may conclude that for the system under analysis final disposal on agricultural soils contributes less to FD.

#### 4.3.1.2 Eutrophication

Plant types adopting agricultural application as final sludge disposal option (T1, T2, T3B, T4B, Figure 4.4) show a negative contribution on FE, which is due to the avoided P emission from avoided production of conventional fertilizer. All plant types have a positive contribution to FE from effluents emission, namely total P emission into water. A change in the end-of-life treatment of sludge (T3 and T4 versus T3B and T4B) lowers the impact significantly, thanks to the avoided P emission to freshwater occurring during the production of P-fertilizer. It should be mentioned that the surface runoff from agricultural application is not included in the LCIA model for the impact category freshwater eutrophication.

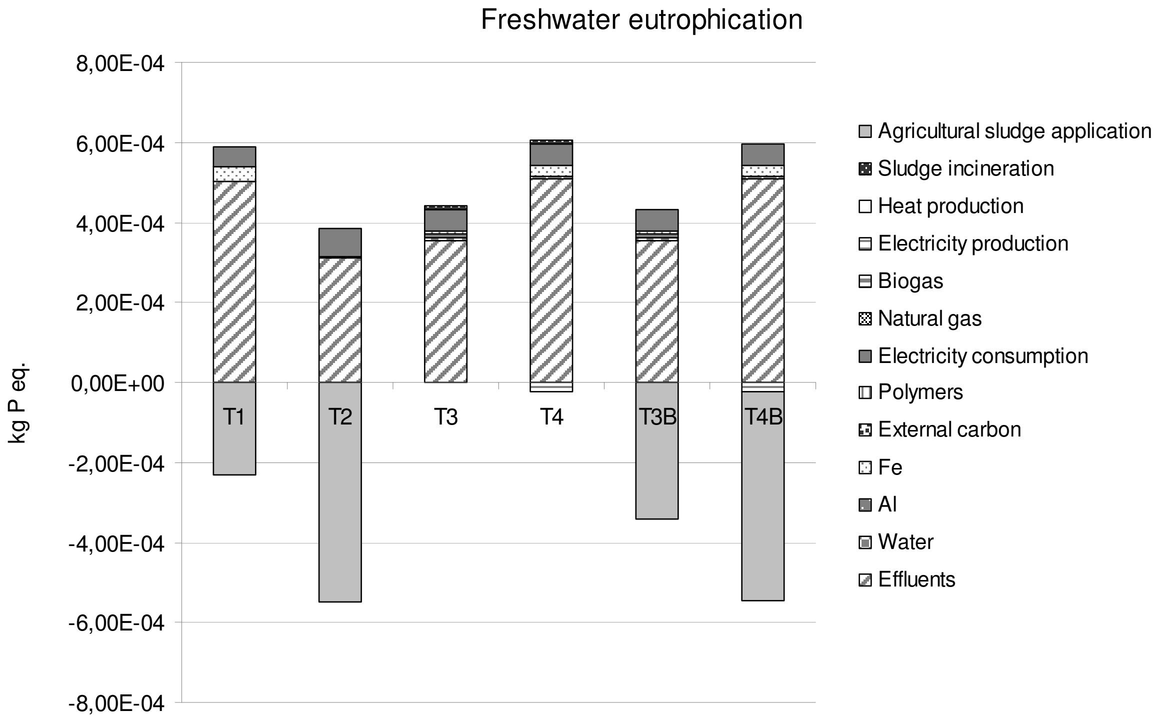


**Figure 4.4** *Visualisation of process-specific contributions to the freshwater eutrophication (FE) impact category, measured in kg P eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

As shown in Figure 4.5, for ME results depend almost entirely on the effluent emissions into water, in particular total N emissions into water and, to a lesser extent, ammonia and nitrate (see Table 4.3). Plant type 2 has the lowest emissions and therefore the best environmental performance. Type 1 and 4 have the largest positive impact, because of the highest total nitrogen and ammonia emissions into water, respectively.

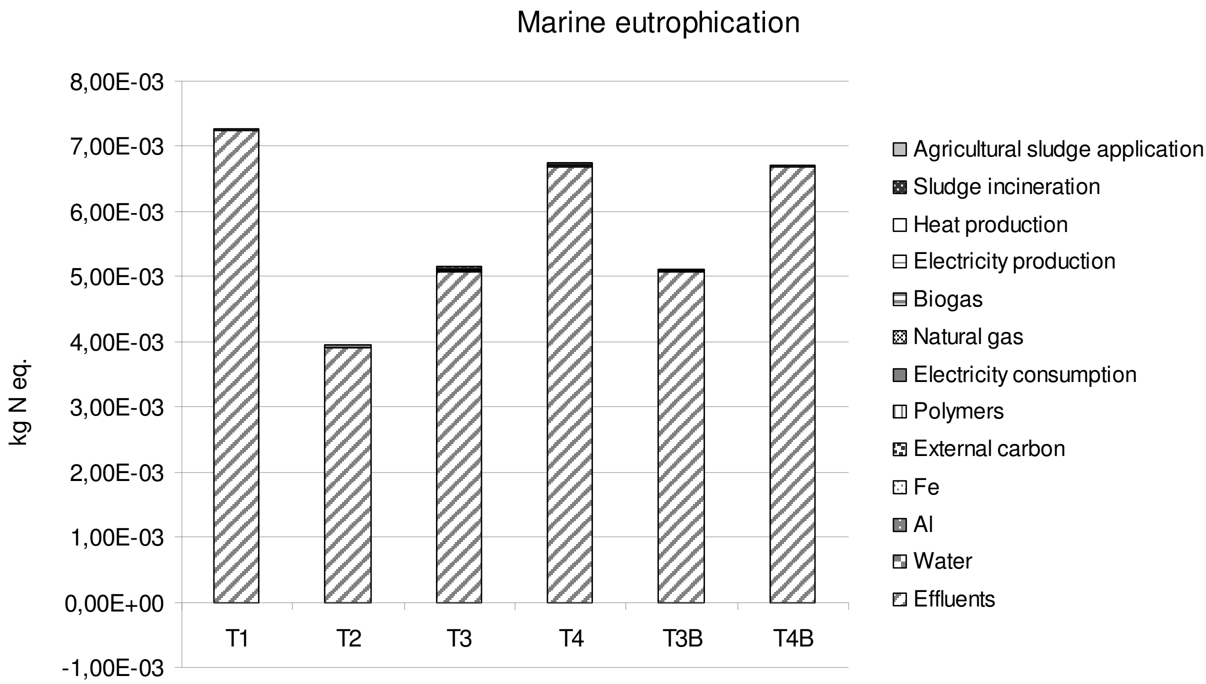


**Figure 4.5.** *Visualisation of process-specific contributions to the marine eutrophication (ME) impact category, measured in kg N eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

## 4.3.1.3 Toxicity impacts

As shown in Figure 4.6, for all plant types the most relevant contribution is connected with fossil energy produced outside the plant. For plant type 1, 3 and 4 a relevant contribution originates also from ferric chloride production. Ferric chloride consumption for type 1 is higher than for the other plant types, therefore this plant type has the highest impact for HT.

For plant type 4 there is a negative value from electricity produced inside the plant, which contributes to lower the overall HT impact. Concerning end of life of sludge, for plant type 3 and 4 we see a positive contribution to HT due to sludge incineration, while for plant type 1 and 2 we observe a negative impact associated with avoided air emissions from avoided fertilizer production. Similar negative emissions would occur in case of alternative scenarios for plant type 3 and 4 (T3B and T4B in Figure 4.6).

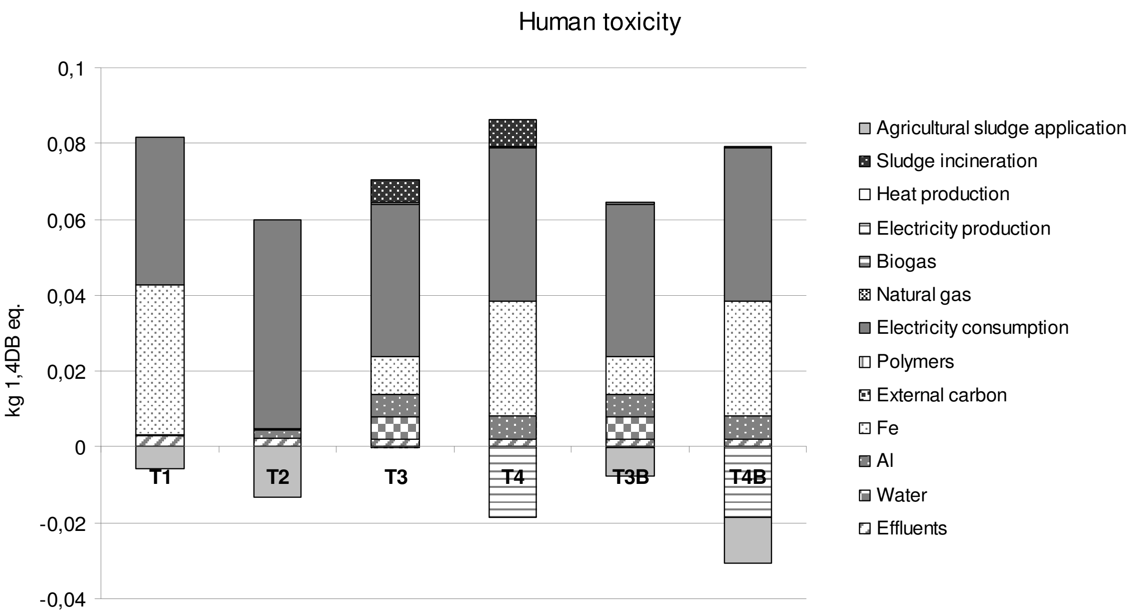


**Figure 4.6.** *Visualisation of process-specific contributions to the human toxicity (HM) impact category, measured in kg 1.4 DB eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

From Figure 4.7, type 1 and type 2, which include agricultural sludge application, have much higher TET impacts than type 3 and 4, whose final sludge treatment is 100% incineration. TET impacts are due to the emission of heavy metals (mostly copper and zinc) into agricultural soil (see Table 4.4). In particular, type 2 has the highest impact due to the highest sludge production per FU. It should be mentioned that the unit process "agricultural sludge application" includes both positive and negative emissions. Positive emissions derive from sludge application into agricultural soil, whereas negative emissions originate from the avoided fertilizer use. Since the emissions into soil from agricultural sludge application are higher than the emissions into soil from fertilizer (and the substitution rate is not 1:1, as described in section 4.2.2.3), only the (positive) net impact is shown in Figure 4.7 as both contributions are nested within this unit process.

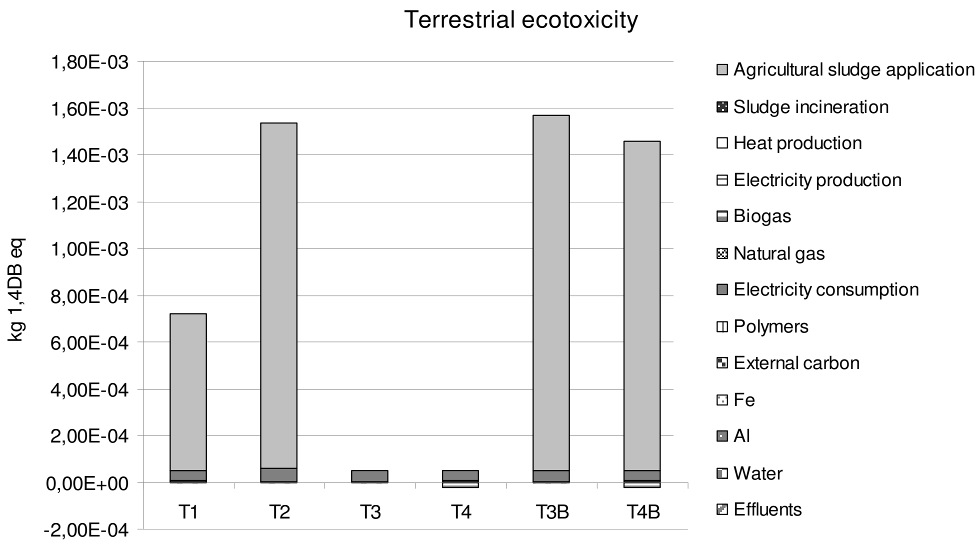


**Figure 4.7.** *Visualisation of process-specific contributions to the terrestrial ecotoxicity (TET) impact category, measured in kg 1.4 DB eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

As visualized in Figure 4.8, the FET impact is mainly due to three processes: electricity consumption from the grid, ferric chloride production (for all plant types except type 2) and aluminium chloride (for all plant types except type 1). Within these processes, the substances responsible for the impact are emissions of nickel and manganese into water from coal-based energy production. Moreover, a relevant contribution is due to the copper concentration levels in the effluent wastewater, which is higher for type 2. Furthermore, for type 4 there is an avoided contribution from the electricity produced internally from biogas, meanwhile for type 3 there is also a positive contribution from natural gas used by the WWTP.

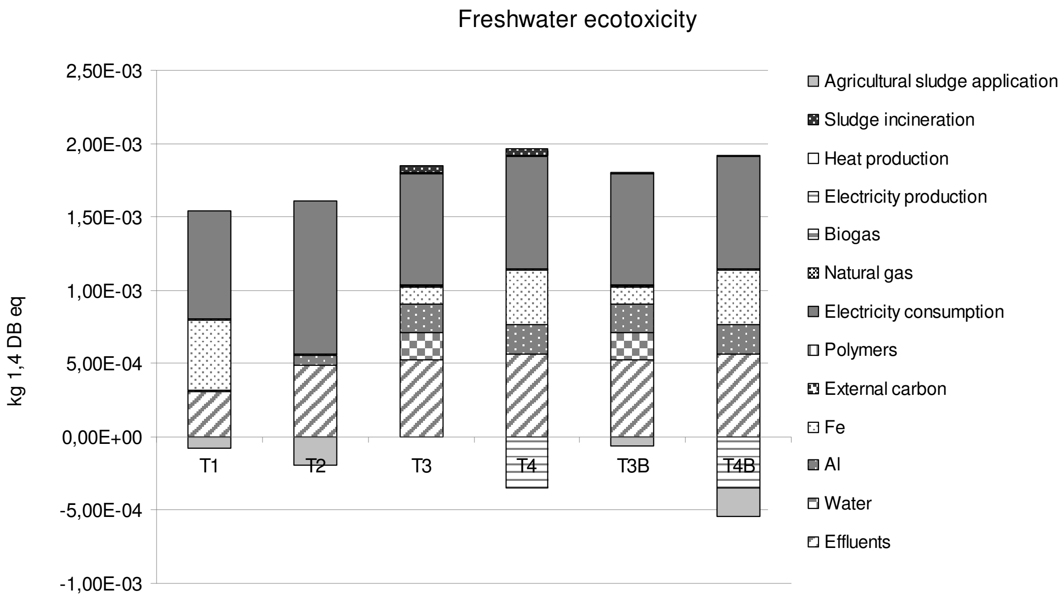


**Figure 4.8.** *Visualisation of process-specific contributions to the freshwater ecotoxicity (FET) impact category, measured in kg 1.4 DB eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

From Figure 4.9, for MET the largest positive contribution is due to effluent emission, namely copper emission into water, which is higher for plant type 3 and 4. Furthermore, a positive contribution is given by the fossil energy production outside the plant. Regarding agricultural sludge application (T1, T2, T3B, T4B, Figure 4.9), there is an avoided MET impact from fertilizer production. Its contribution is directly dependent on the sludge production per functional unit, therefore it is larger for type 2 rather than for type 1.

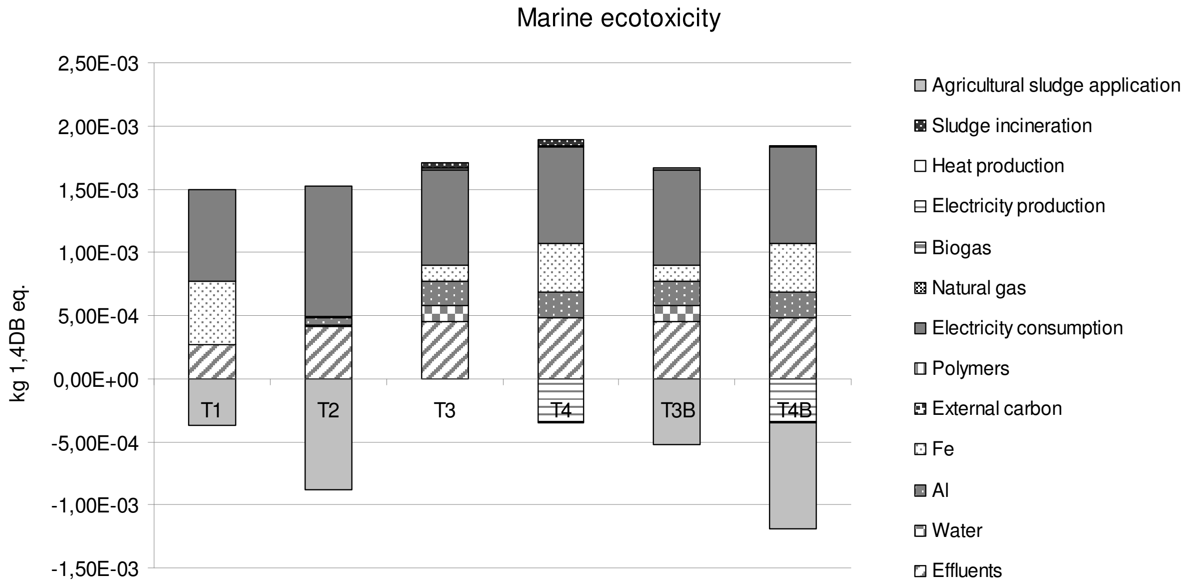


**Figure 4.9.** *Visualisation of process-specific contributions to the marine ecotoxicity (MET) impact category, measured in kg 1.4 DB eq., for the four plant types of the actual scenarios (T1, T2, T3, T4) and for the alternative scenarios (T3B, T4B).*

## *4.3.2 Choice of the LCIA method and its influence on the final results*

The LCIA for the actual WWTP scenario with USEtox is shown in Figure 4.10. We reported as a reference the LCIA results with Recipe, with regard to HT and FET. In this section the contribution of the main substances responsible for the impact is detailed, for both LCIA methods.

From Figure 4.10, for HT calculated with Recipe, which includes cancer as well as and non-cancer impacts, plant type 1 has the highest impact. The substance mainly responsible of the impact to HT is manganese emission into water. This emission originates from fossil energy production outside the plant and is connected with the unit processes “electricity consumption” and “Fe”, i.e. ferric chloride production (see Figure 4.6). When we unfolded this unit processes, we found that the emission of manganese into water derives from a downstream process, i.e. landfill disposal of residual lignite mine waste rock and coal mine waste rock, as reported by the Ecoinvent database (Doka, 2009). A smaller contribution to the HT impact from internally produced electricity is also due to lead and mercury emissions into air. Ferric chloride production also includes mercury emission into air.

For HT calculated with USEtox (cancer impacts), plant type 3 and 4 have the highest impact. This is mainly due to 2,3,7,8-tetrachlorodibenzo-p dioxins emissions into air from sludge incineration, which is the sludge treatment option for these two plant types in the actual scenario. Cancer impacts for type 1 and 2 are due to the emission of tetrachloro methane and CFC-10 into air from ferric chloride production and to the emission of formaldehyde into air from electricity consumption from the grid, respectively.

For HT calculated with USEtox (non-cancer impacts) impacts are due to the emission of tetrachloro methane into air from ferric chloride consumption (type 1, 3 and 4) and to the emission of hexane into air from electricity from the grid production. The ranking among the plant types is proportional to the chemicals consumption use (Table 4.2).

The discordance between the two LCIA methods may be explained by differences in the calculation of the fate factors, like e.g. the inclusion/exclusion of specific exposure pathway and substances (Rosenbaum et al., 2008). The results provided by USEtox seem to be more complete regarding transport routes leading to human exposure. The exposure pathways considered are not only inhalation through air but also ingestion through drinking water, agricultural produce, meat and milk, and fish (Rosenbaum et al, 2011). With regard to ingestion through agricultural produce, it should be mentioned that only organic pollutants for which plant uptake and human health characterization factors exist are included in the HT impact category (Rosenbaum et al., 2008). As all the metals seem to be excluded, the contribution of carcinogenic metals such as Cd and Ni, neurotoxic metals such as Hg and Pb as well as endocrine disrupting chemicals such as nonyl phenol/-ethoxylates, emitted to soil through agricultural sludge application, is not included in the HT impact category. Therefore the contribution of the two sludge end of life treatment options is not equally addressed within this impact category and the impact assessment is biased. Moreover, it should be underlined that electricity consumption and sludge incineration were modeled using secondary datasets from the ecoinvent database, choosing the “unit process” option, which details the contribution of all the subunits included in the life cycle of the modeled process or system (Frischknecht et al., 2007).

For the freshwater ecotoxicity impact category, Figure 4.10 shows that according to the USEtox method type 3 has the highest impact. This is explained by the use of chlorotanonil (fungicide) in molasses production (used as external carbon). Molasses production was modeled using secondary data from the Ecoinvent database (Jungbluth et al., 2007). Furthermore, type 1 and 2 present avoided impacts due to the avoided phosphate rock extraction for fertilizer production. Differently from FET evaluated with Recipe, with USEtox the negative contribution from conventional fertilizer production exceeds the positive contribution from electricity production.

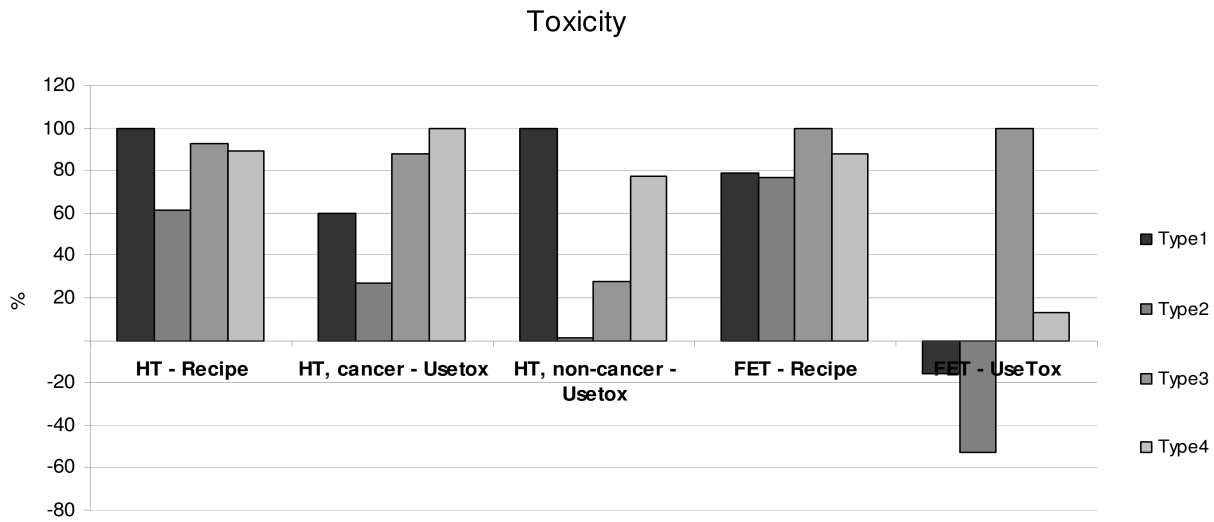


**Figure 4.10**. *Comparison of different LCIA methods for the impact categories HT and FET (Recipe Vs USEtox).*

Figure 4.11 shows the LCIA for the abiotic resource depletion impact category with CML, including also the results provided by Recipe for fossil depletion as a reference. In this case the results are confirmed: Type 1 and type 2 present the highest impact due to the highest fossil fuels consumption from the electricity from the grid.

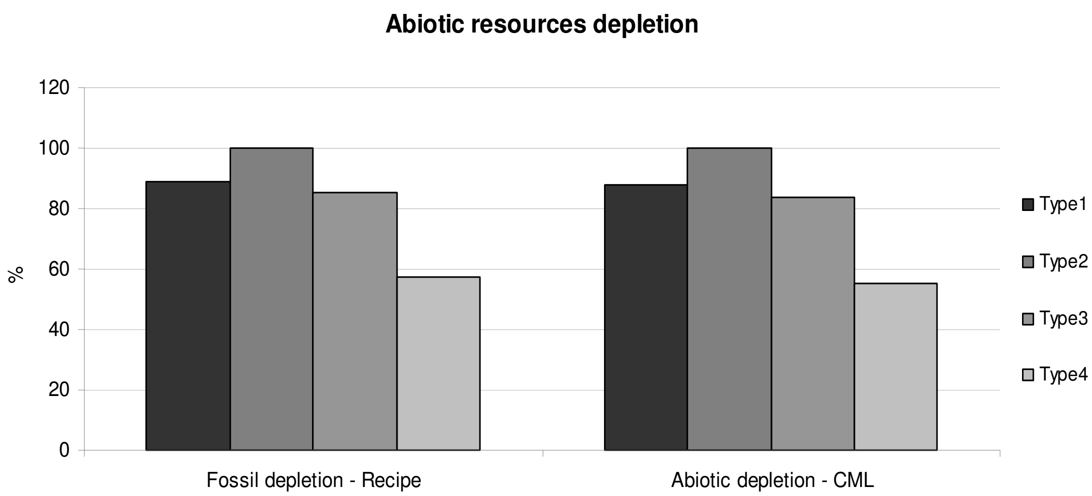


**Figure 4.11.** *Comparison of different LCIA methods (Recipe vs CML) for the assessment of abiotic resources depletion.*

In summary, the sensitivity analysis performed with the methods recommended by the ILCD Handbook showed that LCIA results calculated with USEtox and Recipe for the HT and FET categories (Figure 4.10) are different. This is due to the different importance given to the substances responsible of the impact. Previous studies (Pizzol et al., 2010a, 2010b) demonstrated that this discordance is explained by the large differences existing between the characterization factors calculated for specific chemical substances, e.g. metals, in the two LCIA methods. The results provided by USEtox suggest that chemicals consumption has an influence in the environmental performance of the different WWTP technologies; therefore even though more advanced technological solutions provide improved levels of waste water treatment and nutrient removal, they determine higher impacts for the HT and FET impact categories. The comparison of results obtained with CML and Recipe for the abiotic resource depletion impact category shows the same trend and ranking among the plant types, even though the Recipe method accounts only for impacts from use of fossil fuels, excluding minerals extraction.

These results are consistent with those reported in the LCA study of Renou et al. (2008), where the influence of the choice of LCIA method on the results is determined for the case of a urban WWTP. Renou et al. (2008) concluded that for impact categories such as climate change and resource depletion the results provided by different LCIA are similar, whereas, large discrepancies exist for the human toxicity impact category.

### 4.3.3 Uncertainty analysis results

Based on the probability distribution of the input data (Tables 4.2, 4.3 and 4.4) the Monte Carlo simulation provided the distribution and confidence intervals of the results of each impact category (Figure 4.12).

Figure 4.12c shows that the FE impact scores of types 1, 2, 3B and 4B have the largest uncertainty, whereas scores for CC (Figure 4.12a), FD (Figure 4.12b), and TET for type 3 and 4 (Figure 4.12f) show the lowest deviation from the mean. As the main contributor to FE is the emission of P into water, there is a direct link between the dispersion of this emission (Table 4.3) and the uncertainty associated with the final results.

For ME (Figure 4.12d), we observe larger uncertainty for results of type 1 than for other types. This can be explained by the large uncertainty associated with the estimate of N emitted into water (Table 4.3). This emission is the main contributor to ME and for type 1 the standard deviation of N emission into water is higher than the value of other plant types.

Among the toxicity-related impact categories (Figures 4.12e,f,g,h), MET shows the highest deviation from the mean (Figure 4.12h). This is due to the uncertainty associated with the estimate of copper emitted into water and the standard deviation is higher for type 2 and 4.

The high confidence in LCA results for the climate mitigation impact categories indicates that the environmental profiles of the WWTP types are accurate for these impact categories, and also provide a better basis for the LCA comparisons between the different plant types. For the remaining impact categories the LCA results don’t allow drawing robust conclusions due to the large uncertainties of the analysis.

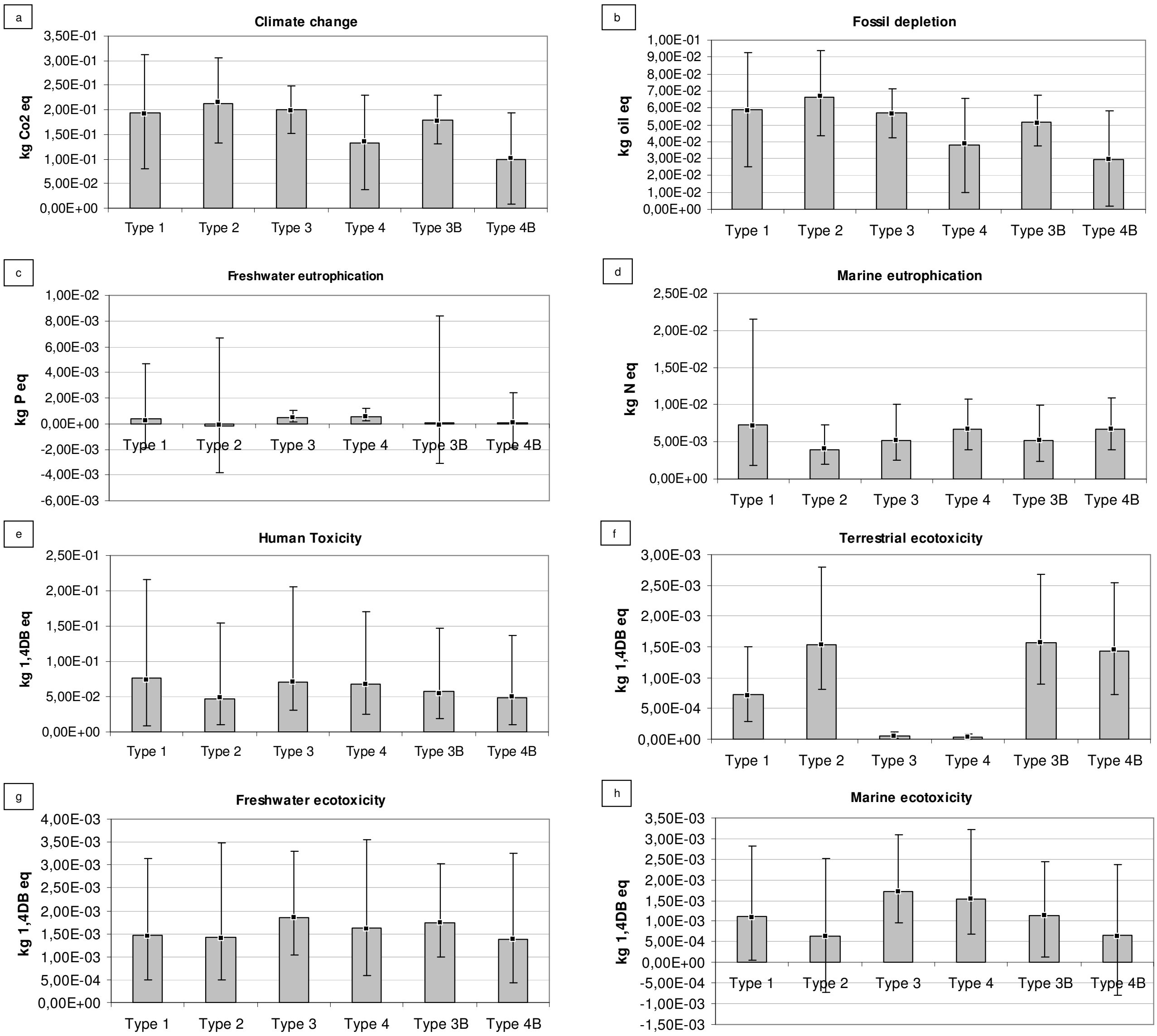


**Figure 4.12.** *Uncertainty analysis showing LCIA results calculated with Recipe for the actual and alternative scenario (grey bars), mean coming from Monte Carlo simulation (black dot), and 95% confidence interval (error bars). The error bars indicate that in 95% of the cases the characterized LCIA would fall within the range.*

## 4.4 Conclusions

We applied the LCA methodology to evaluate the environmental performances of four different technological solutions representative of Danish WWTPs, taking into account different end of life options for sludge disposal, given the importance of recycling phosphorus from sludge. The ranking between the different plant types changed according to the impact categories considered. According to the actual scenario big plants equipped with anaerobic sludge digestion performed better than small aerobic plants for climate mitigation and TET impact categories. Instead, for both freshwater and marine eutrophication and other toxicity-related categories (HT, FET, MET) they show higher environmental impacts. An important contribution to the reduced impact of small plants applying aerobic sludge treatment is due to the sludge application on agricultural soil. According to the LCIA results presented, recycling of phosphorus to agricultural soils represents an alternative sustainable sludge management practice in Denmark compared to sludge incineration regarding the CC and FD impact categories. This is confirmed by the results of the sensitivity analysis on final sludge treatment for plants equipped with anaerobic digestion. Agricultural sludge application is an option that decreases the potential environmental impacts for all impact categories (except TET), thanks to the avoided impacts from conventional fertilizer production. However, as mentioned, the potential impacts from ingestion of agricultural produce are excluded from the impact assessment. With these limitations in mind, agricultural sludge application appears to be the best end-of-life treatment option for big centralized WWTP. Regarding the end-of life of sludge, the sensitivity analysis showed that both sludge quality and quantity are the most influential parameters. In particular, concerning with the quality aspect, P content determines the substitution rate and the avoided impacts, whereas heavy metals drive the TET results. Moreover, given the same sludge quality, the higher is the amount of sludge produced per FU, the higher are the impacts, both positive and negative.

The results of the sensitivity analysis on the LCIA method proved the validity of the LCIA results for climate mitigation, meanwhile toxicity-related scores confirmed to be sensitive to the choice of the impact assessment method. We thus believe that the sensitivity analysis on the LCIA phase – performed by applying different LCIA methods and by determining whether the choice of the method influences the final outcomes- improved the understanding and the transparency of our results.

Furthermore, the influence of input data variability tested with the uncertainty analysis via Monte Carlo simulation showed that the impact categories with highest uncertainty (FE and MET) are those highly dependent on the effluent emissions. Therefore the variability of water and sludge quality parameters greatly affects the LCIA results. We believe that the use of primary (measured) data represents the best option for modeling effluent and sludge quality in the context of LCA for WWT management. We used monitoring data and grouped them into four technology groups representative of Danish WWTPs and this resulted in high uncertainties in the impact assessment phase, due to the natural variability in influent wastewater characteristics between the plants. The probability distribution of grouped data was used to derive uncertainty estimates for the Monte Carlo analysis. If empirical data are not available for calculating the uncertainty distribution, we recommend performing the uncertainty assessments of the LCA by using expert judgment to make qualified estimates.

Further improvement for the study would be to address the influence of chemical and biological P removal technologies in terms of P recycling efficiency, bioavailability as well as the influence of using chemical precipitation on the quality of the sludge. Considering the relevance of system boundaries definition, the study could also be deepened by investigating the influence of transport of sludge to the final disposal as well as the operation connected with sludge storage. Finally, the selection of the sustainable WWT technology and sludge practice in Denmark, should be based not only on efficient use of resources, resource preservation and emission reductions, but also on improved quantification of soil and human health impact categories. Moreover, the assessment of the economic and social repercussions of the different options may be included.

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# Chapter 5

# Use of proxy LCI indicator in the beverage packaging sector

This chapter[6] discusses the significance of the use of non renewable fossil CED as proxy indicator in the beverage packaging sector, in order to detect those situations in which companies can benefit from the use of proxy indicators before a full LCA application. Starting from a case study of two milk containers, the objectives are to assess if the use of this inventory indicator can be a suitable proxy indicator both (i) to decide which is the packaging alternative with the lowest environmental impact and (ii) to identify the most impacting process units of the two products under study.

## 5.1 Introduction

The choice of the Life Cycle Impact Assessment (LCIA) methods is important in an LCA, particularly with reference to the use of the results obtained from the application of LCA to a particular product or process. Depending on whether an evaluation of a specific impact category is required (Pant et al. 2004) or whether performance of a product or process from several viewpoints is needed (Mizsey et al. 2009), the assessment of the results from different LCIA methods could be critical for decision making. In fact, depending on the motivations and objectives, a particular impact assessment method or category may be more suitable (Dreyer et al. 2003, Bovea et al. 2006).

It is known that differences between impact assessment methods are huge (Hauschild et al. 2008) and the effect of the impact assessment method on the final outcome of an LCA has previously been evaluated in many industrial sectors: plastic materials (Bovea et al. 2006), chemicals (Dreyer et al. 2003, Pant et al., 2004), PV technologies (Manzardo et al.2011). A comparison of results obtained by applying different LCIA methods to the same product has not yet been presented in the packaging literature, particularly in the beverage packaging sector.

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There are two recent bodies of work that relates to the selection of relevant LCIA categories for packaging (GreenBlue 2009, Consumer Goods Forum 2011). The first study aims to provide a comprehensive set of indicators and metrics focused on packaging-level measurement in order to define which attributes and impacts of packaging should be measured in terms of sustainability performance and why. With regard to the environmental aspect particular emphasis is given to the measurement of energy use, as there is a strict connection between this aspect and other environmental impacts. The burning of fossil fuel, indeed, releases many emissions, such as GHG emissions which contribute to climate change, sulphur dioxide which contributes to the creation of acid rain and particulate matter, which can influence human health (GreenBlue 2009). The study by Consumer Goods Forum (2011) distinguishes two types of life cycle indicators in the environmental area: inventory indicators and impact categories indicators. There are some indicators which are advised if energy from fossil fuels is used, such as global warming potential, particulate respiratory effects, acidification potential, non-renewable resources depletion. This is because the extraction and use of resources for energy generation is acknowledged as a major contributor to a wide range of environmental impact categories. For this reason the inventory indicator fossil CED (Cumulative Energy Demand) (Frischknecht and Jungbluth 2007) has historically been used as a proxy indicator for other environmental impact categories in LCA screening studies and its appropriateness as an indicator for the environmental performance has been tested for the following product categories: energy production, material production, transport and waste treatment (Huijbregts et al. 2006).

Furthermore, the calculation of the direct and indirect use of energy associated with the life-cycle of products has been widely discussed in the development of the LCA methodology. The very first studies of life-cycle energy analysis refer to the calculation of the cumulative energy associated with the production of beverage containers (Hannon, 1972), and the importance of net energy analysis for technologies assessment (Brendt, 1982, Hannon, 1980). In the paper of Hannon (Hannon, 2010) the objective of life cycle energy analysis was to compare the system of refillable containers and the system of throwaway containers (Hannon, 2010). The interest on energy savings has been always relevant in the packaging sector. In fact the first LCA applications were focused on the energy aspects relevant to different types of beverage containers (e.g. plastic, glass, and aluminum) (Hunt et al. 1996).

Within the packaging sector, the beverage packaging industry play an important role in the protection from environmental influences, such as heat, light, moisture, oxygen, pressure, enzymes, microorganisms, dust particles, which can cause deterioration of the beverages, but it can generates potential adverse impacts to the environment over its life cycle. In recent years, many companies in the beverage packaging sector (Von Falkenstein et al. 2009) have been utilizing LCA as a tool to analyze the environmental performance of their packaging systems and the number of potential applicants is increasing.

As LCA can be used both for product improvement (internal use) and product comparison (external use), different situations need to be discussed within this sector. For instance, when the aim of a company is to compare the environmental performances of its products with competing products for comparative assertions intended to be disclosed to the public, the same data quality for both products has to be provided, according to the ISO 14040-44 standards. Data collection is a time and resource demanding task, which requires a close cooperation between the companies involved in the study.

Another typical situation refers to the identification of opportunities to improve the environmental performance of products at various stages of their life cycle and the definition of priority measures according to the identified most impacting life cycle stages. This situation requires to collect many life cycle data, too.

Therefore, companies would benefit from the use of simplified analysis before starting a full LCA application. Though the usefulness of fossil CED has previously been tested in other sectors (Huijbregts et al. 2006) there is no evidence in literature that its use can effectively support a preliminary LCA in the beverage packaging sector.

This research intends to fulfill this gap by discussing to what extent companies in the beverage packaging sector can benefit from the use of the inventory indicator non-renewable fossil CED for screening life cycle analysis.

The objectives of this work are to assess if the use of non-renewable fossil CED can be a suitable proxy indicator both (i) to decide which is the packaging alternative with the lowest environmental impact and (ii) to identify the most impacting process units of the two products under study, by the means of a comparison with a selection of other impact categories (climate change, fossil depletion, particulate matter formation, photochemical oxidant formation and terrestrial acidification). Finally some insights for the selection of the most relevant categories in the context of screening LCAs in the packaging sector are provided.

Starting from these preliminary remarks, a single case study in the beverage packaging sector is considered with regard to an Italian company, where LCA methodology was used to: (1) compare two different packaging alternatives and (2) identify the environmental hot-spots in the packaging value chain of the two products.

## 5.2 Materials and methods

With regard to the first objective, the goal of the comparative LCA was to evaluate and compare the potential environmental impacts from cradle to grave of two packaging alternatives used for containing long-life milk (UHT), a laminated carton container consisting of six alternating layers of polyethylene, paper, and aluminum and a triple-layer HDPE bottle. The LCA was made according to ISO 14040 standards. The functional unit chosen was the capacity of the packaging to contain a liter of milk. The process units to be included within the system boundaries were selected to compare the two types of packaging, according to the process units usually considered in the packaging field: raw and auxiliary material extraction and processing, production, transport, use, and end-of-life (Mourad et al. 2008, Humbert et al. 2009). Figures 5.1 and 5.2 show the product system and the system boundaries, respectively, of the laminated carton container and the HDPE bottle. As the LCA study is of a comparative nature, the environmental and energy loads of the use phase have been considered comparable for both types of packaging and some stages, marked in Figures 5.1 and 5.2, were not included in the system boundaries (Humbert et al. 2009). The data provided for the LCI phase, for both containers, consist of primary and secondary data collected during previous LCA studies conducted for similar products, i.e. for the end of life scenario, some assumptions were made regarding the percentage of waste to landfill, incineration with energy recovery and recycling, according to Italian statistics (Corepla, 2004), and some secondary data referencing databases in SimaPro© software (PRé Consultants, 2008) used for LCA application.

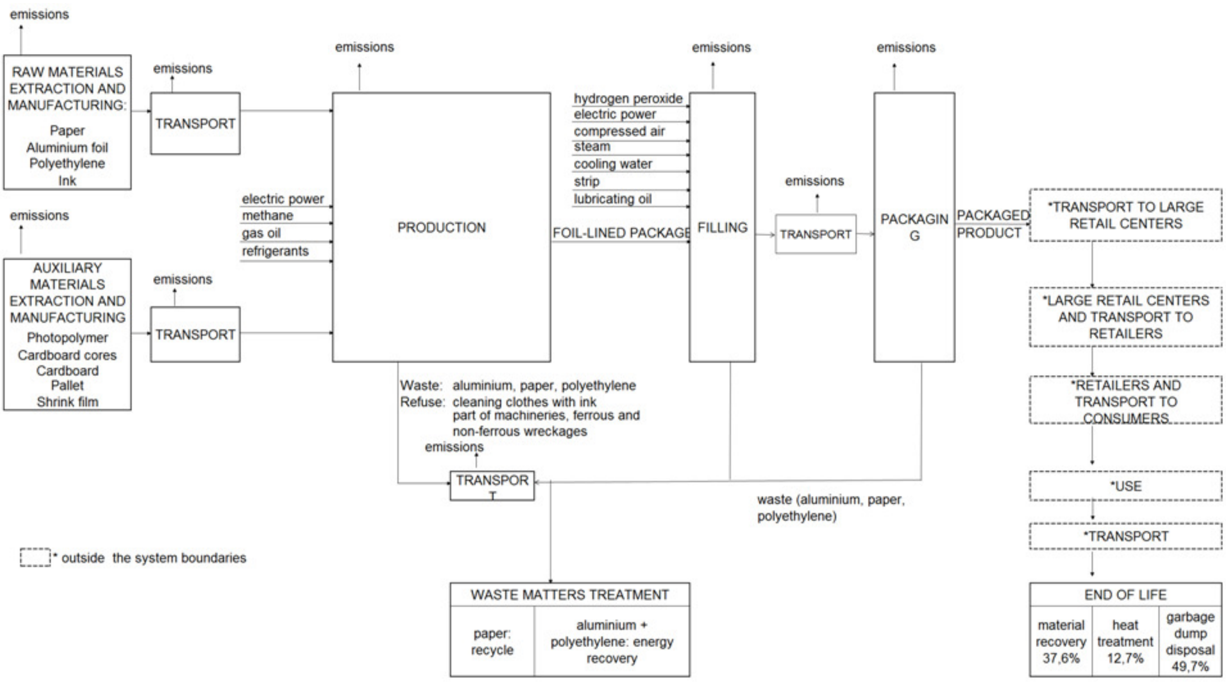


**Figure 5.1** *Product system for the laminated carton container and definition of the processes included in the system boundaries*

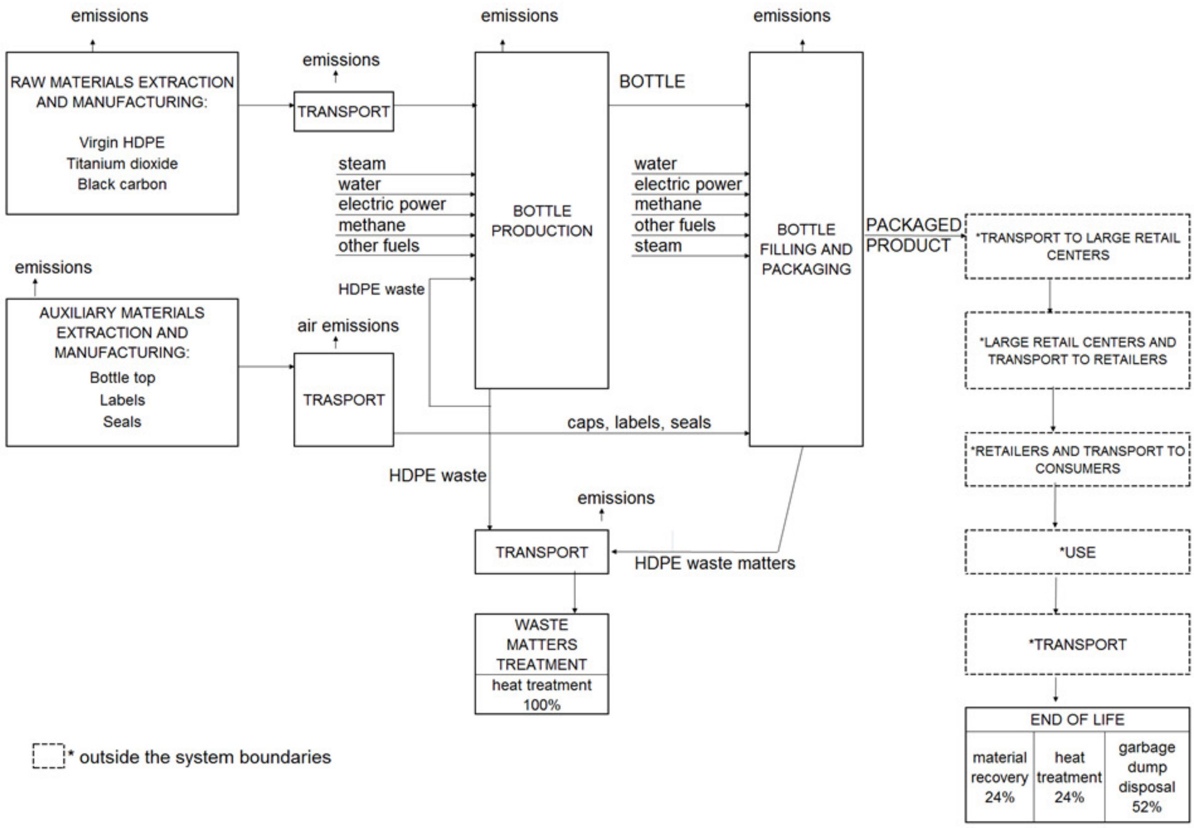


**Figure 5.2** *Product system for the laminated carton container and definition of the processes included in the system boundaries*

The quantification of the environmental impacts is conducted with one of the most recommended LCIA methods in the context of the packaging sector (Consumer Goods Forum 2011): ReCiPe 2008 (Goedkoop et al. 2009) with a selection of impact categories recommended at the endpoint level by the ILCD Handbook (EC-JRC, 2011), i.e. particulate matter formation and photochemical oxidant formation. Furthermore, the results of the comparative LCA obtained with the non-renewable fossil CED indicator, which was selected as the most significant issue for the study, are compared with the results of the impact categories which are connected with the use of fossil fuels: fossil depletion, climate change, particulate matter and acidification (GreenBlue 2009). As a consequence of this selection, in addition to the set of relevant impact categories provided by ReCiPe 2008, in the LCIA step the midpoint LCIA method IPCC 2007 (with the 100-year timeframe used, being the timeframe basis for the Kyoto protocol) (Frischknecht and Jungbluth 2007) was considered. Furthermore, energy from fossil fuels and climate change are recognized as of primary interest in the beverage packaging sector (von Falkenstein et al., 2010).

In order to test the robustness of the comparative LCA results, within the life cycle interpretation step, both an uncertainty analysis of the inventory using Monte Carlo statistical technique (Humbert et al. 2009) and a sensitivity analysis on the end-of-life assumptions (Pasqualino et al., 2011) were performed.

With reference to the second objective, each of the product systems in Figure 5.1 and Figure 5.2 was analyzed separately in order to determine which are the environmental hot spots in terms of life cycle stages. Again, the LCIA results for all the above mentioned impact categories were compared with the results coming from the life cycle inventory flow non-renewable fossil CED. In order to compare the results provided by the different methods a contribution analysis was performed (Dreyer et al., 2003, Pizzol et al., 2011a,b).

## 5.3 Results

In Figure 5.3 the results of the comparative LCA obtained using the selected impact categories by ReCiPe 2008, the climate change category by IPCC and the inventory indicator non renewable fossil CED are displayed as percentage, in order to determine whether different results are provided by the different life cycle categories.

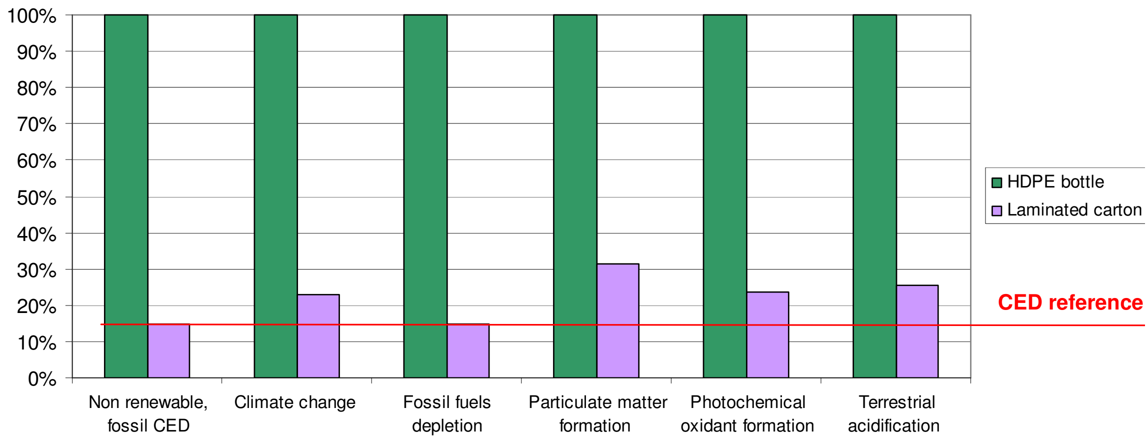


**Figure 5.3** *Inventory and impact categories results from the comparative LCA between a laminated carton and HDPE bottle*

The results given by the non renewable fossil CED indicator are confirmed by all the impact categories considered: the laminated carton container is the packaging solution for containing long-life milk with the best performances from an environmental point of view. As the contribution of fossil fuels was identified as the most significant issue of the analysis, the value provided by non renewable fossil CED was considered as reference value. Some differences in relative terms can be outlined: there is a minimum deviation from the reference value for fossil fuels depletion and a maximum deviation for particulate matter formation. According to these results, it seems that non renewable fossil CED undervalues the LCIA results.

Within the life cycle interpretation step, an uncertainty analysis of the inventory for the comparative LCA was performed using Monte Carlo statistical techniques. Table 5.1 presents the results of the comparison in terms of mean value and standard deviation.

**Table 5.1** *Results of the uncertainty analysis in terms of mean value and standard deviation for the life cycle of the laminated carton container and the HDPE bottle*

| **Container** | | **Laminated carton container** | | **HDPE bottle** | |
|---|---|---|---|---|---|
| Life cycle category | Unit of measurement | Mean | Standard Deviation | Mean | Standard Deviation |
| Non renewable fossil CED | MJ | 0.735 | 0.0177 | 4.99 | 0.401 |
| Climate change | kg CO2 eq | 0.0609 | 0.00338 | 0.262 | 0.0209 |
| Fossil depletion | $ | 0.278 | 0.00763 | 1.9 | 0.167 |
| Particulate matter formation | DALY | 1.90E-08 | 4.02E-10 | 7.8E-08 | 8.49E-09 |
| Photochemical oxidant formation | DALY | 7.99E-12 | 1.77E-13 | 2.5E-11 | 3.57E-12 |
| Terrestrial acidification | species.yr | 1.48E-12 | 2.96E-14 | 5.72E-12 | 6.26E-13 |

Figure 5.4 shows the graphical results of the uncertainty analysis for the comparison between the HDPE bottle and the laminated carton container for the selected categories.

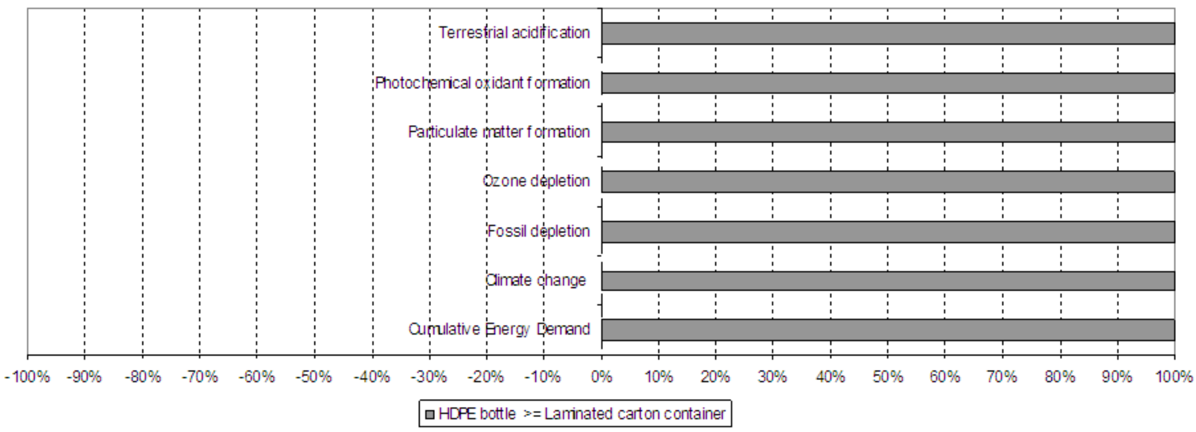


**Figure 5.4** *Graphical results of the uncertainty analysis for the comparison between the HDPE bottle and the laminated carton container for the life cycle categories considered in the comparative LCA*

It confirms that the impacts of the HDPE bottle are higher than those of the laminated carton container for all impact categories. The level of statistical significance is equal to 100%. Furthermore, a sensitivity analysis was conducted in order to test the influence of the assumptions on the end-of-life stage to the overall results. Three scenario variants considering different end-of-life parameter were compared, assuming that 100% of the packaging was disposed through recycling, incineration with energy recovery and landfill, respectively. Table 5.2 shows, for each selected category, the percentage results between the two packaging alternatives. It confirms that, regardless of the disposal options, the laminated carton container has lower impact than the HDPE bottle. Again, the value provided by non renewable fossil CED are lower than the results of the other life cycle impact categories.

**Table 5.2** *Inventory and impact categories indicators considered in the sensitivity analysis focusing on the end of life assumption*

| **End of life option** | **Recycling** | | **Incineration with energy recovery** | | **Landfill** | |
|---|---|---|---|---|---|---|
| Life cycle category | Laminated carton | HDPE bottle | Laminated carton | HDPE bottle | Laminated carton | HDPE bottle |
| Non renewable, fossil CED | 18.6% | 100% | 14.5% | 100% | 14.5% | 100% |
| Climate change | 19.2% | 100% | 16.1% | 100% | 22.8% | 100% |
| Fossil fuels depletion | 18.5% | 100% | 14.5% | 100% | 14.5% | 100% |
| Particulate matter formation | 90.9% | 100% | 27.0% | 100% | 27.4% | 100% |
| Photochemical oxidant formation | 24.7% | 100% | 23.8% | 100% | 23.8% | 100% |
| Terrestrial acidification | 25.5% | 100% | 25.9% | 100% | 25.9% | 100% |

A further analysis on the LCA results for each single container was conducted with regard to all the selected impact categories in order to define if they are in accordance with the results of the non-renewable fossil CED. Based on the analysis of the laminated carton container, for all impact categories the largest contributing unit process was raw materials extraction and manufacturing, but in the case of non renewable fossil CED, as well as all the other impact categories except climate change, the relative contribution is greater than the value emerging from the climate change analysis, as shown in Figure 5.5.

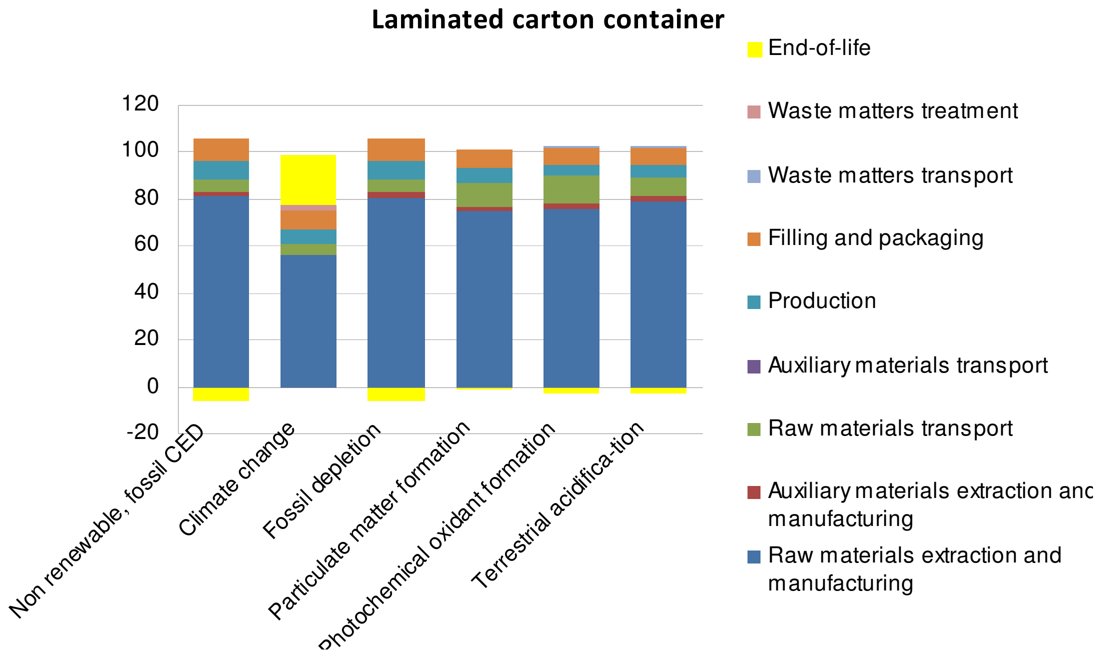


**Figure 5.4** *Results of the contribution analysis of the laminated carton container*

Furthermore, the analysis of climate change aspect shows that an important contribution is given by the end-of-life phase, which is, instead, contributing positively to the reduction of the overall impact in the other categories, as it contributes as an avoided impact. A minor difference can be outlined for the impact categories particulate matter formation, terrestrial acidification, photochemical oxidant formation, whose second largest contributing unit is raw materials transport, instead of filling and packaging, which is the second largest contributing unit for non renewable fossil CED and fossil fuels depletion.

In the case of the HDPE bottle, as shown in Figure 5.5, some differences can be found in the definition of the most impacting process units. They are the production phase for particulate matter formation, terrestrial acidification, photochemical oxidant formation, climate change, and raw material extraction and manufacturing for non renewable fossil CED and fossil fuels depletion, respectively. Furthermore the contribution from raw material extraction and manufacturing was twice for the energy related categories. Again, for all categories except climate change, the end-of-life stage had a negative contribution, which is not taken into account by the analysis on the climate change aspect. As a consequence, if one considered only the non renewable fossil CED, especially in the case of the HDPE bottle, the contribution of the production stage would be underestimated. Furthermore, if one considered only the climate change, for both containers the analysis of the most contributing process units would be distorted; i.e. the possible positive contribution to the overall impact given by the end-of-life stage would be neglected.

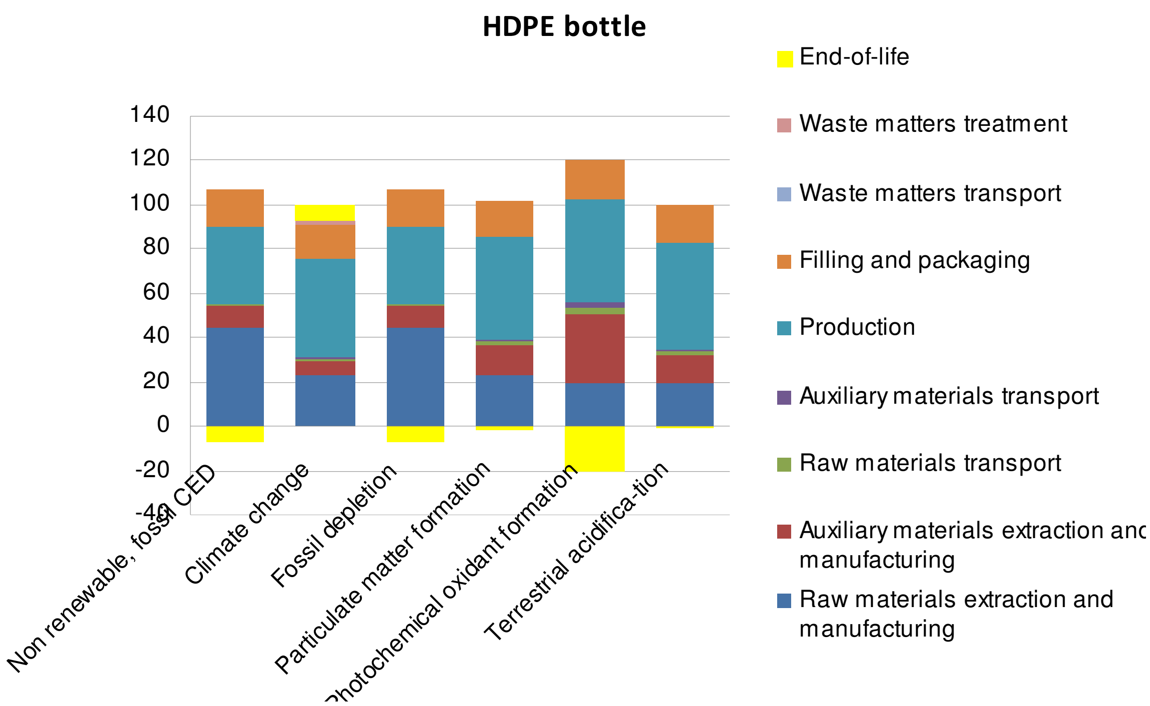


**Figure 5.5** *Results of the contribution analysis of the laminated carton container*

## 5.4. Discussion

From the analysis of the results of the comparative LCA and contribution analysis of the packaging under study some trends can be highlighted, if these results are compared with other findings in the beverage packaging literature. From the comparative LCA the use of non-renewable fossil CED revealed to be useful for a screening, as the results are confirmed by other impact categories which have a connection with the non-renewable resources depletion, i.e. climate change, fossil fuels depletion, particulate matter, terrestrial acidification, (GreenBlue, 2009) and photochemical oxidant formation. If the aim of the LCA study was to define which is the packaging solution with a lower environmental impact, the choice of an inventory indicator such as non renewable fossil CED could have led to the same decision as if a comprehensive LCIA method was used.

This result is consistent with the results reported in the meta-analysis conducted by IFEU (von Falkenstein et al. 2009) which provided an overview of LCA applications on beverage cartons (mainly laminated carton container) and other packaging systems. In the case of fresh milk indeed, the results provided by non renewable fossil CED are confirmed by other impact categories such as acidification, climate change and summer smog (which corresponds to photochemical oxidant formation).

As a consequence, in the milk packaging sector a screening LCA using non renewable fossil CED as proxy could be useful in the decision making process. If the performance of the packaging of a company are better than the performance of the competing products, this company can decide to go on with a full LCA.

With regard to the hot spot analysis, as for both containers the main energy-impacting stage of the life cycle is raw material extraction and manufacturing, the choice of the relevant LCIA category favour a category that is able to quantify the impact of the raw material production processes, which are energy-intensive. But the raw material extraction is not the only one relevant life cycle stage in the packaging sector. Our analysis, indeed, revealed that there are other important impacting process, besides raw material extraction and manufacturing , i.e. the end of life stage in relation to climate change or production for the HDPE bottle in relation to particulate matter formation, terrestrial acidification, photochemical oxidant formation.

This result is confirmed by other studies involving laminated carton container, i.e. in the milk packaging (Xie et al., 2011) where raw material extraction was the highest of the total environmental impacts contributor in the packaging life cycle except for the disposal stage, as well as in the juice and water packaging (Pasqualino et al., 2011). Considering the results of the sensibility analysis, if we assumed that 100% of the packaging was disposed through recycling, also the results of the hot spot by climate change would confirm the results provided by the other impact categories, differently from the base scenario. But the contribution of the recycling stage (representing the end of life stage) is underestimated by climate change, if compared with the contribution analysis provided by the non renewable fossil CED, as shown in Tables 5.3 and 5.4.

**Table 5.3** *Results of the contribution analysis of the laminated carton container—sensitivity analysis (end of life 100 % recycling)*

| | **Container** | **LAMINATED CARTON** | | | | | |
|---|---|---|---|---|---|---|---|
| | Life cycle category | *Non renewable, fossil CED* | *Climate change* | *Fossil depletion* | *Particulate matter formation* | *Photochemical oxidant formation* | *Terrestrial acidification* |
| Unit process | Unit of measurement | % | % | % | % | % | % |
| | Raw materials extraction and manufacturing | 81.8 | 74.8 | 81.8 | 76.9 | 80.8 | 80.6 |
| | Auxiliary materials extraction and manufacturing | 1.8 | 1.5 | 1.8 | 2.2 | 2.3 | 2.0 |
| | Raw materials transport | 5.5 | 6.8 | 5.5 | 10.2 | 12.2 | 8.2 |
| | Auxiliary materials transport | 0 | 0 | 0 | 0.1 | 0.1 | 0 |
| | Production | 7.9 | 8.9 | 7.9 | 6.0 | 5.1 | 5.9 |
| | Filling and packaging | 9.7 | 9.9 | 9.7 | 8.4 | 7.9 | 7.5 |
| | Waste matters transport | 0.1 | 0.1 | 0.1 | 0.2 | 0.4 | 0.2 |
| | Waste matters treatment | 0 | 2.8 | 0 | 0.1 | 0.1 | 0.1 |
| | End-of-life | -6.9 | -4.8 | -6.9 | -4.0 | -8.9 | -4.5 |
| | Total impact | 100 | 100 | 100 | 100 | 100 | 100 |

**Table 5.4** *Results of the contribution analysis of the HDPE bottle—sensitivity analysis (end of life 100 % recycling)*

| | Container | HDPE BOTTLE | | | | | |
|---|---|---|---|---|---|---|---|
| | Life cycle category | *Non renewable, fossil CED* | *Climate change* | *Fossil depletion* | *Particulate matter formation* | *Photochemical oxidant formation* | *Terrestrial acidification* |
| Unit process | Unit of measurement | % | % | % | % | % | % |
| | Raw materials extraction and manufacturing | 44.9 | 25.2 | 44.8 | 23.2 | 19.9 | 19.7 |
| | Auxiliary materials extraction and manufacturing | 10 | 7.3 | 10 | 13.7 | 31.5 | 13 |
| | Raw materials transport | 0.6 | 1 | 0.6 | 1.9 | 3.1 | 1.6 |
| | Auxiliary materials transport | 0.3 | 0.4 | 0.3 | 1 | 2.1 | 0.7 |
| | Production | 34.1 | 48.9 | 34.2 | 45.8 | 47.1 | 48.2 |
| | Filling and packaging | 17.1 | 16.5 | 17.1 | 16.3 | 17.4 | 17.4 |
| | Waste matters transport | 0 | 0 | 0 | 0 | 0 | 0 |
| | Waste matters treatment | 0 | 1.6 | 0 | 0 | 0.1 | 0 |
| | End-of-life | -6.9 | -1.0 | -6.9 | -2.0 | -21.2 | -0.8 |
| | Total impact | 100 | 100 | 100 | 100 | 100 | 100 |

This result is consistent with those reported in the paper of Pasqualino et al. (2011), which compares the different stages of the beverages' life cycles considering recycling as disposal option (aseptic carton for juice, aluminum can for beer and PET bottle for water). From the environmental profile for the different stages of three beverages' life cycle, subtracting the contribution of the beverage to the overall impact, the contribution of the recycling stage (representing the end of life option) is underestimated by climate change, if compared with the contribution analysis provided by non renewable fossil CED.
Even if there is a correlation between climate change and non renewable fossil CED, as most of the impact on climate change is attributable to energy consumption, the hot spot analysis provided contrasting results. If the aim of the LCA is to have a rough reliable description of the most impacting process units of the beverage packaging, the suggestion is therefore to consider not only non renewable fossil CED as indicator but climate change as well.

## 5.5 Conclusions

LCA methodology is time-consuming and requires many efforts for data collection. What could be useful for a company in the packaging field is the conduction of a preliminary screening analysis which needs less data to be collected.
The present study focused on the significance of the use of non renewable fossil CED as proxy indicator in the beverage packaging sector, in order to detect those situations in which companies can benefit from the use of proxy indicators before starting a full LCA application.
The aim was to test if the results provided by the LCI indicator non-renewable fossil CED are consistent with those of a full LCA using a comprehensive LCIA method such as ReCiPe for a selection of impact categories and a midpoint method such as IPCC 2007 for climate change with regard to two typical situation within LCA, namely product comparison and product improvement.
A case study within an Italian company in the beverage packaging sector was considered, where LCA methodology was used to: (1) compare two different packaging alternatives and (2) identify the environmental hot-spots in the packaging value chain of the two products.
In the first application, a comparative LCA study was applied to two containers for milk, a laminated carton container and an HDPE bottle, and the results of each study were compared. Three different types of life cycle categories were selected, an inventory indicator (non renewable fossil CED), the midpoint LCIA method IPCC 2007 (climate change) and a selection of energy-related impact categories from one comprehensive LCIA methodology (ReCiPe 2008): photochemical oxidant formation, particulate matter formation, terrestrial acidification, fossil depletion. Regardless of the impact method chosen and considering the outcomes of both uncertainty and sensibility analysis, the less impacting packaging is the

laminated carton container. This application confirms the predictability of an LCIA result out of a proxy indicator in the milk packaging sector, where energy data are the most available.
The second application, focusing on the identification of environmental hot-spots in the packaging value chain, revealed that the choice of an inventory indicator as non renewable fossil CED can lead to contrasting results, if compared with another impact categories. Indeed, there could be other unit processes responsible for the environmental impact, not necessarily the raw materials and extraction process being the most impacting, as demonstrated by the analysis at climate change level, which gives more relevance to the end of life stage.
Ultimately, this study leads to two conclusions: in the beverage packaging sector, the use of the screening LCI indicator non-renewable fossil CED: (i) can be useful to determine which is the alternative with the lowest environmental impact, but (ii) can lead to misleading decisions when assessing the contribution of life cycle stages to the overall impact. Therefore companies in the beverage packaging sector can benefit for the use of non renewable fossil CED if their aim is to obtain a preliminary estimation of the life cycle environmental impacts of two or more competing products.
Finally, some insights can be formulated to decide which impact categories need to be considered within a screening analysis in the beverage packaging sector. The importance of raw material extraction and manufacturing stages to the overall impact and the importance of fossil fuel contributions to the total resource extraction cannot be neglected. At the same time the importance of the end of life stage can not be ignored, because the environmental impacts in the beverage packaging sector could not be fully explained by fossil energy use. As a consequence, non renewable fossil CED and climate change can be used together to provide a rough estimation of the environmental profile in the beverage packaging sector. It could be interesting to test if this correlation is valid also within other sectors, apart from milk packaging.
As in the future development of beverage packaging system LCA will be necessarily integrated in the design process, it is important to define some ways of simplifying its application and spread its use among companies. However companies need to be aware that the results of a screening LCA should not be used for comparative assertions to be disclosed to the public. Additional research is required to define which unit process could be safely omitted while analyzing or comparing different beverage packaging alternatives without greatly affecting the results. Further investigation is also needed in order to define specific guidelines that can help in the selection of the most suitable LCIA categories for screening LCAs according not only to the objective and purpose of the study, but also to the industrial sector.

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# Chapter 6

# Parametric model for the LCI of wooden pallet

This chapter discusses the use of parametric Life Cycle Inventory models as a support in the design phase of new products. A LCI parametric model is set to define the life cycle of a series of wooden pallets with similar characteristics. The model is tested with one reference product, namely a 4-blocks non reversible pallet and the most influent parameters are identified. The model is then applied to further 12 products and correlation between the most influent parameters and the environmental impacts are defined.

## 6.1 Introduction

Parameterization refers to the practice of presenting LCA data using raw data and formulas instead of computed numbers in unit process datasets within databases (Cooper et al., 2012) The use of parametric model is widely recognized as a useful tool for optimizing data collection within the Life Cycle Inventory step of an LCA study (Müller et al., 2004). This technique can indeed be used to define the Life Cycle Inventory of a range of products, when they present similar characteristics. Furthermore, when focusing at the a company level, a distinction should be made between operations which are common for all the products and a those operations which are specific for some products. As a consequence it can be helpful to develop a model based on a defined set of parameters to describe the life cycle inventory and use this model to assess the environmental impacts of products having similar characteristics.
Parametric estimation techniques, coming from production and/or Life Cycle Cost estimation Practice, have been proposed as a simplified LCA technique for estimating the environmental impacts of similar technical products based on a limited number of LCA studies. The aim of these environmental parametric estimation techniques is to establish a coupling between functional requirements or design parameters that product developers have at hand in early design phases and the environmental impact of the product (Dick et al., 2004).
The main applications of parametric LCI models refer to the design phase (Dick et al., 2004; Müller et al., 2004; Geyer R., 2008; Early et al., 2009; Collado-Ruiz and Ostad-Ahmad-Ghorabi, 2009; Ostad-Ahmad-Ghorabi and Collado-Ruiz, 2011), as the early design phase is

where life cycle assessment is most effective and where it is easiest to carry out changes and to reduce environmental impact.
One of the most effective way of using parametric models within a Life Cycle Assessment is to determine a relationship between the environmental impacts and parameters relevant for the product under study.
Starting from these premises a case study in the wooden pallet sector was analyzed, with the aim of developing a parametric model to describe the Life Cycle Inventory of a range of wooden pallet used as tertiary packaging.
The parametric model was tested with a reference product, namely a non reversible pallet with 4-way blocks, represented in Figure 6.1, for which the environmental impacts using the LCA methodology as defined by ISO 14040 (2006a) was used, in order to define the most impacting life cycle stages.

**Figure 6.1** *Representation of the reference product: a non reversible pallet with 4-way blocks*

The technical characteristics of the reference pallet are the following:
- 5 upper boards 1200x70x16 mm (lenght x width x thickness);
- 3 lower boards 1200x70x16 mm
- 3 axles 800x70x16 mm;
- 9 blocks 70x70x75 mm (lenght x width x height);
- 18 helicoidal nails;
- 18 helicoidal pointless nails;
- 24 smooth nails.

Finally, the parametric LCI model was used to determine correlations between the environmental impacts and the most significant inventory parameters.

### *6.1.1 Definition of the LCI parameters*

The aim of the definition of the LCI parametric model is to quantify inventory data starting from the definition of the life cycle parameters.
The life cycle of the wooden pallet is described in Figure 6.2, which includes all the steps from raw materials acquisition to the end of life of the product. The raw materials considered are wood (softwood such as pinewood or fir and hardwood, such as poplar and alder) which is used for the elements which constitute the pallet and nails (smooth, helical and pointless

helical). Auxiliary materials are all the materials used for the packaging of raw materials and final product (plastic and metallic straps, cardboard, labels). After the storage they enter the manufacturing process, which includes some operations which are common to all the wooden pallets (cutting and assembly) and some other operations which are optional:

- revelling: cutting of the edges of the pallet;
- milling: superficial treatment;
- high temperature treatment: the wood must be heated to achieve a minimum core temperature of 56 °C at least 30 minutes in order to conduct a phytosanitary treatment;
- stamping: conducted in conjunction with the high temperature treatment.

The use phase is excluded from the system boundaries. Meanwhile the transport of the final product and waste, as well as the waste treatment and final end of life of the product is included.

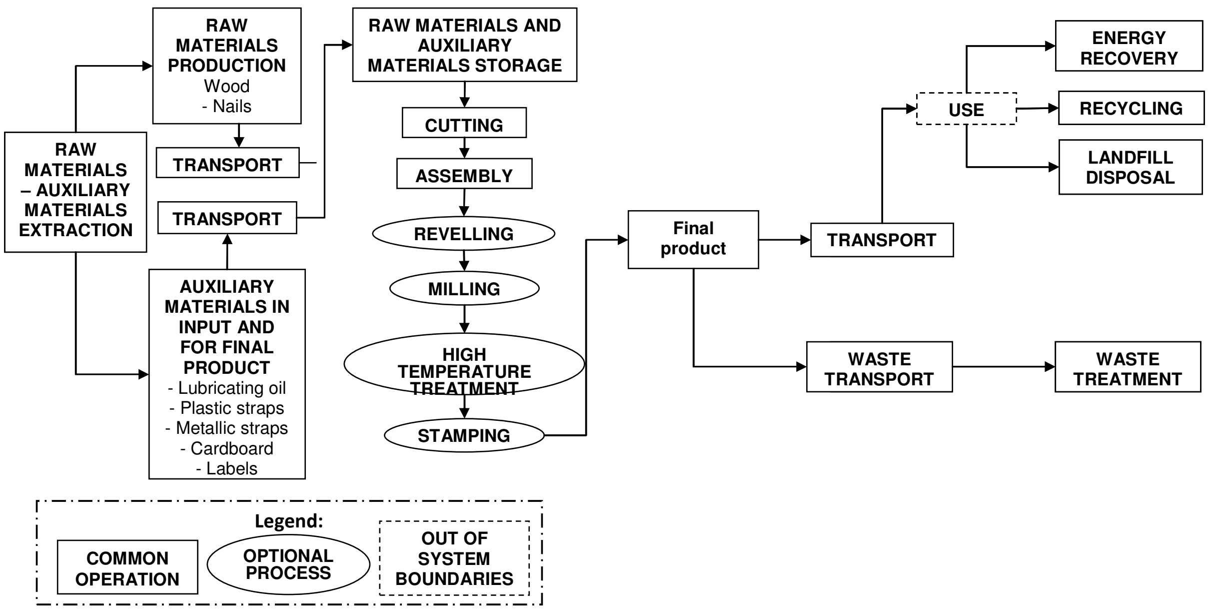


**Figure 6.2** *Definition of the general life cycle of the wooden pallets manufactured by the company with inclusion of the process units which are common to all the products and those which are optional.*

Once defined the process units which describe the life cycle of a generic pallet manufactured by the company, it was possible to quantify the relationships among the different life cycles by the means of parameters, which can be divided into two categories: independent parameters and dependent parameters.

Independent parameters refer to characteristics inherent in the product and can be divided into some sub-categories:

- Number of elements: boards *(n_tav),* blocks (*n_tap*), axles (*n_tr*), nails (*n_c*);
- Dimension of elements: lenght (*l_i*), width (*a_i*), thickness (*s_i*) with *i = tav, tap, tr*
- Distance for transport of the elements/final product: *dist_i* with *i = tav, tap, tr;*

- Switch (optional proc): revelling (*fres*), milling(*smus*), high temperature (*HT*), stamping (*timb*);
- Others: nails mass (*mass_c*), waste fraction (*tw*), hardwood fraction (*hard_i*), softwood fraction (*1-hard_i*) with i = tav, tap, tr.

A full list and description of the 30 independent parameters is provided in Table 6.1.

**Table 6.1** *List and description of the independent parameters*

| Parameter | Description | Parameter | Description |
|---|---|---|---|
| *n_tr (-)* | number of axles | *rho (kg/m³)* | nominal density of wood = 500 |
| *n_tap (-)* | number of blocks | *pw (-)* | waste during cutting = 0.03 |
| *n_tav_sup (-)* | number of upper boards | *hard_tr* (-) | fraction of hardwood in axles |
| *n_tav_inf (-)* | mass of lower boards | *dist_tr (km)* | weighted distance for the transport of axles |
| *l_tav (m)* | length of boards | *nc_el (-)* | number of helical nails |
| *a_tap (m)* | width of blocks | *mass_c_el (g)* | mass of 1 helical nail |
| *a_tav_sup (m)* | width of upper boards | *nc_l (-)* | number of smooth nails |
| *s_tav_sup (m* | thickness of upper boards | *mass_c_el (-)* | mass of 1 smooth nail |
| *a_tav_inf (m)* | width of lower boards | *mass_c:el_sp* | mass of 1 pointless helical nail |
| *s_tav_inf (m* | thickness of lower boards | *dist_c* | weighted distance for the transport of nails |
| *l_tr (m)* | length of axles | *fres* | switch for milling phase |
| *a_tr (m)* | width of axles | *smus* | switch for board reveling phase |
| *s_tr (m)* | thickness of axles | *HT* | switch for the High Temperature treatment phase |
| *h_tap (m)* | height of blocks | *timb* | switch for the stamping phase |
| *l_tap (m)* | length of blocks | *dist_client (km)* | weighted distance for the transport of the final product |

Dependent parameters are defined through correlations between independent parameters and are listed in Table 6.2 with their calculation formula.

**Table 6.2** *List and description of the dependent parameters*

| Parameter | Description | Formula |
|---|---|---|
| *a_tap (m)* | width of blocks | *= a_tav_inf* |
| *n_tap (-)* | number of blocks | *= n_tr * n_tav_inf* |
| *vol_tav_sup (m³)* | volume upper boards | *= a_tav_sup * l_tav_sup * k_tav_sup* |
| *vol_tav_inf (m³)* | volume lower boards | *= a_tav_inf * l_tav_inf * k_tav_inf* |
| *mass_tav_sup (kg)* | mass of upper boards | *= vol_tav_sup * rho* |
| *mass_tav_inf (kg)* | mass of lower boards | *= vol_tav_inf * rho* |
| *mass_tav_tot (kg)* | total mass of boards | *= n_tav_sup * mass_tav_sup + n_tav_inf * mass_tav_inf* |
| *vol_tap (m³)* | volume of blocks | *= a_tap * l_tap * k_tap* |
| *mass_tap (kg)* | mass of blocks | *= vol_tap * rho* |
| *vol_tr (m³)* | volume of axles | *= a_tr * l_tr * k_tr* |

| Parameter | Description | Formula |
|---|---|---|
| *mass_tr (kg)* | mass of axles | *= vol_tr * rho* |
| *mass_tr_tot (kg)* | total mass of axles | *= n_tr * mass_tr* |
| *nc_el_sp (-)* | number of pointless helical nails | *= n_el* |
| *mass_c (kg)* | mass of nails | *= nc_l * mass_c_l + nc_el * mass_c_el + nc_ el_sp *mass_c_el_sp* |
| *mass_wood (kg)* | mass of wood | *= mass_tap + mass_tr+ mass_tav* |
| *mass_pallet (kg)* | mass of pallet | *= mass_wood + mass_c* |
| *dist_pond (km)* | average distance weight for wood | *= (dist_tap*mass_tap_tot + dist_tav*mass_tav_tot + dist_tr*mass_tr)/ mass_wood* |
| *fw (-)* | waste during milling | *= 0,005 * fres* |
| *sw (-)* | waste during revelling | *= 0,002 * smus* |
| *tw (-)* | total waste | *= fw + sw + pw* |

The life cycle stages considered in the model are listed in Table 6.3. They define the main process units of the product system under study, as they were considered in the Simapro software (PRé Consultants, 2008). Furthermore they reflect the structure of the questionnaires used of data collection.

**Table 6.3** *Life cycle stages of the wooden pallet as modelled in the software*

| Life cycle stage | Description | Parameter | Unit |
|---|---|---|---|
| 1. Wood | Includes the production and transport connected with the different wooden components (blocks, axles, upper boards, lower boards), considering the number of elements included in the product as well as the EURO class and average load of the truck used for the transport. | mass_wood | kg |
| 2. Nails | Includes the production and manufacturing of steel for the different types of nails, as well as the transport connected with nails supply | mass_c | kg |
| 3. Auxiliary materials in input | Includes the production and transport of input auxiliary materials (cardboard boxes, metallic and plastic straps) | mass_wood | kg |
| 4. Auxiliary materials in production | Includes the production and transport of auxiliary materials used in the manufacturing phase (lubricating oil, labels) | mass_wood | kg |
| 5. Pallet manufacturing – common | Includes the consumption of diesel for fork lifts, diesel combusted in the boiler for the drying phase and electricity consumption (hydropower source) for cutting and assembly phases | mass_wood | kg |
| 6. Pallet manufacturing – optional | Includes the consumption of GPL for the oven during the HT treatment, as well as electricity consumption (hydropower source) for reveling, milling, high temperature treatment and stamping | mass_wood | kg |

| Life cycle stage | Description | Parameter | Unit |
|---|---|---|---|
| 7. Waste | Includes the transport and recycling treatment of wood waste (from cutting, reveling and milling) as well as the transport and recycling of labels and metallic straps | mass_wood | kg |
| 8. Emission into air | Includes the emissions of particulates during the manufacturing phase | mass_wood | kg |
| 9. Transport to clients | Includes the transport to the final clients by truck, considering the average composition of | mass_pallet | kg |
| 10. End of life | Includes the end of life of wood pallet, including the final disposal of wood and nails | mass_pallet | kg |
| 11. Pallet – final product | Unit process assembling all the previous life cycle stages | 1 | piece |

For the recycling of waste and of the final product the system expansion option was adopted, considering the selection efficiency and substitution rates for metals in accordance with Rigamonti et al. (2009), Rigamonti and Grosso (2010) and Lazarevic et al. (2010).
For the definition of the end of life of the wood pallet, the scenario has been defined in accordance with Rilegno (2010): 37% in landfill, 60% at recycling, 3% at incineration with energy recovery.

## 6.2 Life Cycle Assessment of the reference product

### *6.2.1 Goal and scope definition*

The goal of the LCA study was to define the environmental potential impacts connected with the life cycle of one non reversible pallet with 4-way blocks by the application of the parametric LCI previously developed.
As the pallet is the structural foundation of a unit load which allows handling and storage efficiencies, the functional unit was considered as one single product unit.
The unit processes included in the system boundaries are defined in Figure 6.2, concerning with the optional step only the high temperature treatment and stamping phases are included.
Input considered inside the system boundaries are:

- raw materials consumption and production (wood, nails);
- auxiliary materials consumption and production (cardboard boxes, metallic and plastic straps, lubricating oil, labels);
- electricity and fossil fuels consumption.

Manufacturing, maintenance and dismantling of infrastructure (buildings and machineries) were excluded from the system boundaries, as well as the use of industrial soil, under the assumption that their contribution to the overall environmental impact can be neglected. As output emissions into air, water and soil deriving from the product system under study were quantified, including waste from the manufacturing phase.

Cut-off criteria based on mass, primary energy, and environmental significance are used to decide whether processes shall be included in the product system and data gathered. A cut-off level of 2% is applied (the process is neglected if it reaches less than 2% of the total known mass, primary energy, and impact, respectively). All processes where data are available are taken into account, even if their contribution is less that 2%. Therefore, the cut-off rule is used to avoid gathering unknown data, but not to neglect known (Humbert et al., 2009).

The LCIA is performed using the Recipe 2008 methodology (Goedkoop et al.,2009) at midpoint level, focusing on a selection of impact categories:

- Climate Change (CC),
- Human Toxicity (HT),
- Particulate Matter Formation (PMF),
- Agricultural Land Occupation (ALO),
- Fossil Depletion (FD).

Primary data and information were obtained directly from the company who manufactures the wooden pallet, by the means of specific questionnaires. Secondary data are obtained from the scientific literature and databases recognized at international level: Ecoinvent (Frischknecht et al., 2005), US Life Cycle Inventory (U.S. Life Cycle Inventory Database, 2012) and ELCD (http://lca.jrc.ec.europa.eu/lcainfohub).

As far as data quality requirement is concerned, the criteria used are described in Table 6.4.

**Table 6.4** *Data quality requirement set in the LCA study.*

| Parameter | Description |
|---|---|
| Time-related coverage | 2010, if secondary data are used they should be at the latest 15 years old |
| Geographical coverage | Data refer to the Italian production, but if data are not available at National level, they refer to average central Europe situation |
| Technology coverage | State of the art of wooden pallet manufacturing |
| Precision | Data refer to average values at annual level mass of lower boards |
| Completeness | The percentage of mass inflow measured or estimated is equal to 95% |
| Representativeness | The level of representativeness of data is high, as data are collected directly by the production site |
| Consistency | The method used for data collection, such as allocation and cut-off criteria, is consistent with the overall method |
| Reproducibility | The data is very specific to this study and can not be reproduced by an independent practitioner |
| Uncertainty about information | The uncertainty about data and hypothesis is coherent width of lower boards |

The application of the parametric model to calculate the environmental impacts of the reference product was critically reviewed by an external expert.

### 6.2.2 Life Cycle Inventory

The definition of the independent and dependent parameters with regard to the reference product is given in Tables 6.5 and 6.6, respectively.

**Table 6.5** *Independent parameters for the reference product*

| Parameter | Value | Parameter | Value |
|---|---|---|---|
| *n_tr (-)* | 3 | *rho (kg/m³)* | 500 |
| *n_tap (-)* | 9 | *pw (-)* | 0.03 |
| *n_tav_sup (-)* | 5 | *hard_tr* (-) | 1 |
| *n_tav_inf (-)* | 3 | *dist_tr (km)* | 1681 |
| *l_tav (m)* | 1.2 | *nc_el (-)* | 18 |
| *a_tap (m)* | 5 | *mass_c_el (g)* | 1.7 |
| *a_tav_sup (m)* | 0.07 | *nc_l (-)* | 24 |
| *s_tav_sup (m* | 0.016 | *mass_c_l (g)* | 1.7 |
| *a_tav_inf (m)* | 0.07 | *mass_c:el_sp (g)* | 1.7 |
| *s_tav_inf (m* | 0.016 | *dist_c* | 72.4 |
| *l_tr (m)* | 0.8 | *fres* | 0 |
| *a_tr (m)* | 0.07 | *smus* | 0 |
| *s_tr (m)* | 0.016 | *HT* | 1 |
| *h_tap (m)* | 0.075 | *timb* | 1 |
| *l_tap (m)* | 0.07 | *dist_client (km)* | 31.8 |

**Table 6.6** *Dependent parameters for the reference product*

| Parameter | Value | Parameter | Value |
|---|---|---|---|
| *a_tap (m)* | 0.07 | *mass_tr (kg)* | 0.488 |
| *n_tap (-)* | 9 | *mass_tr_tot (kg)* | 1.344 |
| *vol_tav_sup (m³)* | 1.344E-03 | *nc_el_sp (-)* | 18 |
| *vol_tav_inf (m³)* | 1.344E-03 | *mass_c (kg)* | 0.1758 |
| *mass_tav_sup (kg)* | 0.672 | *mass_wood (kg)* | 8.37 |
| *mass_tav_inf (kg)* | 0.672 | *mass_pallet (kg)* | 8.55 |
| *mass_tav_tot (kg)* | 5.376 | *dist_pond (km)* | 785.8 |
| *vol_tap (m³)* | 3.675E--04 | *fw (-)* | 0 |
| *mass_tap (kg)* | 0,18375 | *sw (-)* | 0 |
| *vol_tr (m3)* | 8.96E-04 | *tw (-)* | 0.03 |

The application of the parametric model to the reference product allowed to determine the input and output flows describing the life cycle of the pallet, according to the 11 steps of the life cycle, as reported in the following Tables. Both for wood and nails, the contribution of the extraction of raw materials as well as the contribution of the manufacturing stage and transport by truck were considered.

**Table 6.7** *Life Cycle Inventory of life cycle stage "1.Wood" for the reference product*

| Life cycle sub-stage | Input | Amount | Unit | Data set |
|---|---|---|---|---|
| Upper boards | Softwood | 1.04E-03 | $m^3$ | Sawn timber, softwood, planed, air dried, at plant/RER U |
| | Hardwood | 3.46E-04 | $m^3$ | Sawn timber, hardwood, planed, air/kiln dried, u=10%, at plant/RER U |
| | Transport by truck | 0.457 | tkm | Transport, lorry>16t, fleet average/RER U |
| Lower boards | Softwood | 1.04E-03 | $m^3$ | Sawn timber, softwood, planed, air dried, at plant/RER U |
| | Hardwood | 3.46E-04 | $m^3$ | Sawn timber, hardwood, planed, air/kiln dried, u=10%, at plant/RER U |
| | Transport by truck | 0.457 | tkm | Transport, lorry>16t, fleet average/RER U |
| Blocks | Softwood | 0.000379 | $m^3$ | Sawn timber, softwood, planed, air dried, at plant/RER U |
| | Transport by truck | 0.0878 | tkm | Transport, lorry>16t, fleet average/RER U |
| Axles | Hardwood | 9.23E-04 | $m^3$ | Sawn timber, hardwood, planed, air/kiln dried, u=10%, at plant/RER U |
| | Transport by truck | 0.776 | tkm | Transport, lorry>16t, fleet average/RER U |

**Table 6.8** *Life Cycle Inventory of life cycle stage "2.Nails" for the reference product*

| Life cycle sub-stage | Input | Amount | Unit | Data set |
|---|---|---|---|---|
| Smooth nails | Steel | 1.7 | g | Steel, low-alloyed, at plant/RER U |
| | Steel manufacturing | 1.7 | g | Steel product manufacturing, average metal working/RER U |
| Helical nails | Steel | 5.0 | g | Steel, low-alloyed, at plant/RER U |
| | Steel manufacturing | 5.0 | g | Steel product manufacturing, average metal working/RER U |
| Helical pointless nails | Steel | 2.5 | g | Steel, low-alloyed, at plant/RER U |
| | Steel manufacturing | 2.5 | g | Steel product manufacturing, average metal working/RER U |
| Transport (all nails) | Transport by truck | 0.0127 | tkm | Transport, lorry 16-32t, EURO5/RER U |

For auxiliary materials used as input and in the production stage, raw materials extraction and manufacturing are included, as well as transport by truck.

**Table 6.9** *Life Cycle Inventory of life cycle stage "3. Auxiliary materials in input" for the reference product*

| Input | Amount | Unit | Data set |
|---|---|---|---|
| Plastic straps | 3.55E-03 | kg | Nylon 66, at plant/RER U |
| Plastic straps manufacturing | 3.76E-03 | kg | Injection molding/RER U |
| Corrugated board | 1.76E-04 | kg | Packaging, corrugated board, mixed fibre, single wall, at plant/RER U |
| Metallic straps | 4.94E-03 | kg | Steel, low-alloyed, at plant/RER U |
| Metallic straps manufacturing | 4.94E-03 | kg | Steel product manufacturing, average metal working/RER U |
| Plastic straps transport by truck | 2.79E-03 | tkm | Transport, lorry>16t, fleet average/RER U |
| Cardboard transport by truck | 1.27E-05 | tkm | Transport, lorry 16-32t, EURO5/RER U |
| Metallic straps transport by truck | 3.33E-03 | tkm | Transport, lorry>16t, fleet average/RER U |

**Table 6.10** *Life Cycle Inventory of life cycle stage "4. Auxiliary materials in production" for the reference product*

| Input | Amount | Unit | Data set |
|---|---|---|---|
| Lubricating oil | 1.92E-03 | kg | Nylon 66, at plant/RER U |
| Label | 2.0E-01 | g | Injection molding/RER U |
| Label manufacturing | 2.05E-01 | g | Packaging, corrugated board, mixed fibre, single wall, at plant/RER U |
| Lubricating oil transport by truck | 2.81E-04 | tkm | Steel, low-alloyed, at plant/RER U |
| Label transport by truck | 1.7E-06 | tkm | Steel product manufacturing, average metal working/RER U |

The manufacturing operations common to all products include the consumption of diesel both for fork lifts and the boiler and electricity for the cutting and assembly stages.

**Table 6.11** *Life Cycle Inventory of life cycle stage "5. Pallet manufacturing – common" for the reference product*

| Input | Amount | Unit | Data set |
|---|---|---|---|
| Diesel for fork lifts | 3.35E-03 | kg | Diesel, at regional storage/RER U |
| Diesel used in the boiler | 1.14E-02 | l | Diesel, combusted in industrial boiler/US |
| Electricity (cutting phase) | 8.88E-03 | kWh | Electricity, hydropower, at power plant/IT U |
| Electricity (assembly phase) | 1.10E-01 | kWh | Electricity, hydropower, at power plant/IT U |

The optional manufacturing operations for the reference product include the consumption of resources (GPL and electricity) for the High temperature treatment and electricity for the stamping phase.

**Table 6.12** *Life Cycle Inventory of life cycle stage "6. Pallet manufacturing – optional" for the reference product*

| Input | Amount | Unit | Data set |
|---|---|---|---|
| GPL used in the HT oven | 8.82E-02 | l | Liquefied petroleum gas, combusted in industrial boiler/US |
| Electricity (HT treatment phase) | 3.55E-02 | kWh | Electricity, hydropower, at power plant/IT U |
| Electricity (stamping phase) | 1.01E-02 | kWh | Electricity, hydropower, at power plant/IT U |

The operations considered in the waste section are transport to recycling as well as the recycling of the different materials: wood waste, cardboard, plastic and metallic straps and labels. Meanwhile as far as emissions into air are concerned only particulate emissions are considered.

**Table 6.13** *Life Cycle Inventory of life cycle stage "7. Waste " for the reference product*

| Input | Amount | Unit | Data set |
|---|---|---|---|
| Transport wood waste cutting phase to recycling | 5.53E-03 | tkm | Transport, lorry>16t, fleet average/RER U |
| Transport cardboard to recycling | 2.64E-07 | tkm | Transport, lorry>16t, fleet average/RER U |
| Transport plastic straps to recycling | 3.20E-07 | tkm | Transport, lorry>16t, fleet average/RER U |
| Transport labels to recycling | 3.0E-07 | tkm | Transport, lorry>16t, fleet average/RER U |
| Transport metallic straps to recycling | 4.45E-05 | tkm | Transport, lorry>16t, fleet average/RER U |
| **Output** | **Amount** | **Unit** | **Data set** |
| Recycling wood waste from cutting phase to pellet | 2.51E-01 | kg | Recycling wood – Pellet/RER U |
| Cardboard to recycling | 1.76E-04 | kg | Recycling cardboard/RER U |
| Plastic straps to recycling | 3.55E-03 | kg | Recycling PA66/RER U |
| Labels to recycling | 2.0E-01 | g | Recycling PP/RER U |
| Metallic straps to recycling | 4.94E-03 | kg | Recycling steel and iron/RER U |

**Table 6.14** *Life Cycle Inventory of life cycle stage "8. Emission into air" for the reference product*

| Output | Amount | Unit | Data set |
|---|---|---|---|
| Particulate | 4.9E-05 | mg | Particulates emission into air |

The transport to clients consider the average situation for the reference year, meanwhile the end of life of the final product includes the treatment of the different fractions, wood and nails.

**Table 6.15** *Life Cycle Inventory of life cycle stage "9. Transport to clients" for the reference product*

| Output | Amount | Unit | Data set |
|---|---|---|---|
| Transport of the final product by truck | 2.72E-01 | tkm | Transport, lorry>16t, fleet average/RER U |

**Table 6.16** *Life Cycle Inventory of life cycle stage "10. End of life" for the reference product*

| Output | Amount | Unit | Data set |
|---|---|---|---|
| Recycling of wood in the pallet | 5.02 | kg | Recycling wood – system expansion |
| Landfill disposal of wood in the pallet | 3.10 | kg | Disposal, wood untreated, 20% water, to sanitary landfill/CH U |
| Incineration of wood in the pallet | 2.50E-01 | kg | Waste incineration of untreated wood (10.7% water content), EU-27 S |
| Recycling of nails in the pallet | 1.05E-01 | kg | Recycling steel and iron/RER U |
| Landfill disposal of nails in the pallet | 6.51E-02 | kg | Disposal, steel, 0% water, to inert material landfill/CH U |
| Incineration of nails in the pallet | 5.27E-03 | kg | Waste incineration of ferro metals, EU-27 S |

Concerning with the waste and final product disposal, the recycling process is modelled according to the "system expansion" approach (Frischknecht R, 2010), therefore the avoided impacts from primary production are included, considering the following avoided product from the Ecoinvent database:

- wood pellets from wood waste from cutting phase;
- core board for cardboard;
- pig iron for metallic straps;
- nylon 6,6 for plastic straps;
- polypropylene granulate for labels;
- particle board for residual wood (final product).

### *6.2.3 Life Cycle Impact Assessment*

The results of the LCIA step are reported in Figure 6.3, with reference to the selected impact categories.

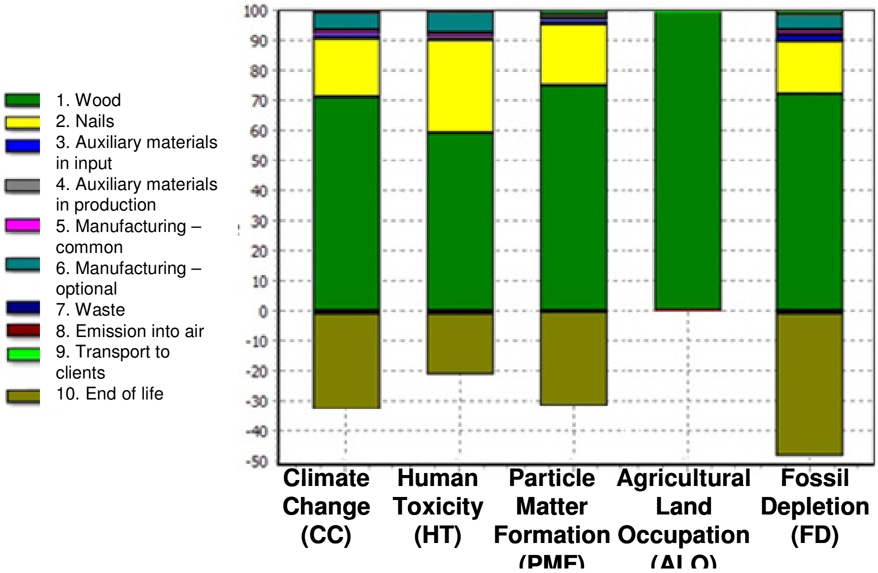


**Figure 6.3** *Life Cycle Impact assessment results at midpoint level*

The numerical results are reported in Table 6.17, according to their unit of measurement.

**Table 6.17** Life Cycle Impact Assessment results for the reference product

| *Impact category* | *Unit* | *Value* |
|---|---|---|
| Climate change | kg $CO_2$ eq | 2.27 |
| Human toxicity | kg 1,4-DB eq | 1.07 |
| Particulate matter formation | kg $PM_{10}$ eq | 5.58E-03 |
| Agricultural land occupation | m2a | 68.9 |
| Fossil depletion | kg oil eq | 6.12E-01 |

From the analysis of the LCIA results, the most relevant life cycle stages can be identified:

- raw materials extraction and transport (wood and nails);
- end of life.

These life cycle stages refer to operations which are conducted outside the company, as they include the extraction of wood and steel used for nails manufacturing. Furthermore also the contribution of transport is included, which has a relevant impact as the major part of suppliers are located in eastern Europe.

None of the operations under the direct control of the company impact more than 2% on the overall impact for all impact categories considered, except for high temperature treatment, whose impact is due to GPL consumption in the oven.

A positive contribution is due to the recycling of wood, which is responsible of about 30% of the impact. These results are confirmed by the results by similar studies on wooden pallet: Gasol et al. (2008) e Anil et al. (2010).

### 6.2.4 Life Cycle Interpretation

The first step of this phase is the identification of the most relevant issues, namely the life cycle stages which are mainly responsible of the impact within each impact categories.

The results of the contribution analysis show that:

- for Climate change the impact is mainly due to fossil $CO_2$ emissions into air coming from the transport of the wood components to the production site; furthermore a contribution is also given by nails manufacturing and to a lesser extent from the combustion of GPL in the oven during the high temperature treatment;
- for Human toxicity the impact is mainly due to manganese emissions into water and mercury emissions into air from electricity production during wood and nails production;
- for Particulate matter formation the impact is mainly due to $NO_X$ and particulates (< 2.5 μm) emissions into air from transport and cutting of the wood;
- for Agricultural land occupation the impact is mainly due to occupation of forestry during wood extraction;
- for Fossil depletion the impact is due to fossil fuel used for the transport and energy production.

## 6.3 Definition of correlations between input parameters and environmental impacts

### 6.3.1 Application of the LCI parametric model to a range of products

The parametric LCI model was applied to calculate the environmental impacts of a range of further 12 products manufactured by the company which can be categorized into 4 different types according to the size and type of treatment:

- products with High temperature treatment (apart from the reference product): B and H;
- corresponding products without High temperature treatment: A, C, I;
- products with a lower mass if compared with the reference product: E, M, N.
- products with a higher mass if compared with the reference product: D, F, G, L.

The list of the independent and dependent parameters for the further 12 products is presented in Tables 6.18 and 6.19.

**Table 6.18** *Summary of the independent parameters for the further 12 products analyzed*

| **Parameter** | **Unit** | A | B | C | D | E | F | G | H | I | L | M | N |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| l_tav | m | 1.200 | 1.200 | 1.200 | 1.200 | 1.200 | 1.000 | 1.200 | 1.200 | 1.200 | 0.900 | 1.200 | 2.130 |
| a_tav_sup | m | 0.070 | 0.070 | 0.070 | 0.070 | 0.070 | 0.090 | 0.070 | 0.090 | 0.090 | 0.070 | 0.070 | 0.023 |
| s_tav_sup | m | 0.016 | 0.016 | 0.016 | 0.016 | 0.013 | 0.016 | 0.013 | 0.016 | 0.016 | 0.016 | 0.013 | 0.021 |
| a_tav_inf | m | 0.070 | 0.070 | 0.070 | 0.090 | 0.070 | 0.090 | 0.070 | 0.090 | 0.090 | 0.070 | 0.070 | - |
| s_tav_inf | m | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.020 | 0.016 | 0.016 | 0.016 | 0.016 | 0.013 | - |
| n_tav_sup | - | 5 | 7 | 7 | 4 | 5 | 7 | 7 | 5 | 5 | 7 | 5 | 15 |
| n_tav_inf | - | 3 | 3 | 3 | 6 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 0 |
| pw | - | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |
| hard_tav | - | 0.25 | 0.23 | 0.23 | 0.15 | 0.12 | 0 | 0 | 0.19 | 0.19 | 0 | 0 | 0 |
| dist_tav | km | 660.9 | 643.3 | 643.3 | 790.4 | 637.3 | 681.8 | 574.6 | 1029.7 | 1029.7 | 493.2 | 601.5 | 538.6 |
| l_tap | m | 0.070 | 0.070 | 0.070 | 0.090 | 0.075 | 0.090 | 0.070 | 0.090 | 0.090 | 0.075 | 0.072* | - |
| h_tap | m | 0.075 | 0.075 | 0.075 | 0.075 | 0.090 | 0.075 | 0.075 | 0.075 | 0.075 | 0.075 | 0.075 | - |
| dist_tap | km | 463.3 | 464.1 | 464.1 | 401.9 | 396.6 | 401.9 | 464.1 | 401.9 | 401.9 | 387.0 | 411.0 | - |
| l_tr | m | 0.800 | 0.800 | 0.800 | 1.000 | 0.800 | 1.000 | 1.000 | 0.800 | 0.800 | 1.000 | 0.800 | - |
| a_tr | m | 0.070 | 0.070 | 0.070 | 0.090 | 0.075 | 0.090 | 0.070 | 0.090 | 0.090 | 0.075 | 0.070 | - |
| s_tr | m | 0.016 | 0.016 | 0.016 | 0.020 | 0.014 | 0.020 | 0.016 | 0.016 | 0.016 | 0.017 | 0.013 | - |
| n_tr | - | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 0 |
| hard_tr | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.02 | 0 | 0 | 0 |
| dist_tr | km | 486.4 | 483.0 | 483.0 | 801.0 | 693.6 | 801.0 | 486.4 | 618.1 | 658.5 | 516.3 | 601.5 | - |
| nc_el | - | 18 | 18 | 18 | 27 | 18 | 27 | 18 | 27 | 27 | 18 | 18 | 0 |
| nc_l | - | 24 | 36 | 36 | 36 | 24 | 36 | 36 | 24 | 24 | 36 | 24 | 0 |
| mass_c_el | g | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | - |
| mass_c_l | g | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.5 | 1.7 | 1.7 | 1.7 | 1.5 | - |
| mass_c_el_sp | g | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | - |
| dist_c | km | 72.4 | 71.1 | 71.1 | 72.4 | 61.7 | 72.4 | 56.6 | 73.4 | 73.4 | 71.1 | 61.7 | - |
| n_et | - | 26 | 29 | 29 | 29 | 27 | 26 | 27 | 29 | 29 | 29 | 23 | 89 |
| smus | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 |
| fres | - | 0 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 |
| HT | - | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 |
| timb | - | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 1 | 0 |
| dist_clienti | km | 29.4 | 23.9 | 19.0 | 71.5 | 37.6 | 78.8 | 9.5 | 48.9 | 17.1 | 23.0 | 13.2 | 2.0 |

**Table 6.19** *Summary of the dependent parameters for the further 12 products analyzed*

| **Parameter** | **Unit** | **A** | **B** | **C** | **D** | **E** | **F** |
|---|---|---|---|---|---|---|---|
| a_tap | m | 0.070 | 0.070 | 0.070 | 0.090 | 0.070 | 0.090 |
| n_tap | - | 9 | 9 | 9 | 9 | 9 | 9 |
| ass | - | 1 | 1 | 1 | 1 | 1 | 1 |
| fw | | 0 | 0.005 | 0.005 | 0.005 | 0.005 | 0.005 |
| sw | | 0 | 0 | 0 | 0 | 0 | 0 |
| tw | | 0.03 | 0.035 | 0.035 | 0.035 | 0.035 | 0.035 |
| vol_tav_sup | $m^3$ | 0.00134 | 0.00134 | 0.00134 | 0.00134 | 0.00109 | 0.00144 |
| vol_tav_inf | $m^3$ | 0.00134 | 0.00134 | 0.00134 | 0.00173 | 0.00134 | 0.00180 |
| mass_tav_sup | kg | 0.672 | 0.672 | 0.672 | 0.672 | 0.546 | 0.720 |
| mass_tav_inf | kg | 0.672 | 0.672 | 0.672 | 0.864 | 0.672 | 0.900 |
| mass_tav_tot | kg | 5.376 | 6.720 | 6.720 | 7.872 | 4.746 | 7.740 |

| Parameter | Unit | A | B | C | D | E | F |
|---|---|---|---|---|---|---|---|
| vol_tap | $m^3$ | 0.000368 | 0.000368 | 0.000368 | 0.000608 | 0.000473 | 0.000608 |
| mass_tap | kg | 0.184 | 0.184 | 0.184 | 0.304 | 0.236 | 0.304 |
| mass_tap_tot | kg | 1.654 | 1.654 | 1.654 | 2.734 | 2.126 | 2.734 |
| vol_tr | $m^3$ | 0.000896 | 0.000896 | 0.000896 | 0.001800 | 0.000840 | 0.001800 |
| mass_tr | kg | 0.448 | 0.448 | 0.448 | 0.900 | 0.420 | 0.900 |
| mass_tr_tot | kg | 1.344 | 1.344 | 1.344 | 2.700 | 1.260 | 2.700 |
| nc_el_sp | - | 18 | 18 | 18 | 27 | 18 | 27 |
| mass_c | kg | 0.176 | 0.196 | 0.196 | 0.264 | 0.176 | 0.264 |
| mass_wood | kg | 8.37 | 9.72 | 9.72 | 13.31 | 8.13 | 13.17 |
| mass_pallet | kg | 8.55 | 9.91 | 9.91 | 13.57 | 8.31 | 13.44 |
| dist_pond | km | 593.9 | 590.6 | 590.6 | 712.7 | 583.1 | 648.1 |
| **Parameter** | **Unit** | **G** | **H** | **I** | **L** | **M** | **N** |
| a_tap | m | 0.070 | 0.090 | 0.090 | 0.070 | 0.070 | - |
| n_tap | - | 9 | 9 | 9 | 9 | 9 | 0 |
| ass | - | 1 | 1 | 1 | 1 | 1 | 0 |
| fw | | 0.005 | 0.005 | 0.005 | 0.005 | 0.005 | 0 |
| sw | | 0 | 0.002 | 0.002 | 0 | 0 | 0 |
| tw | | 0.035 | 0.037 | 0.037 | 0.035 | 0.035 | 0.03 |
| vol_tav_sup | m3 | 0.00109 | 0.00173 | 0.00173 | 0.00101 | 0.00109 | 0.00103 |
| vol_tav_inf | m3 | 0.00134 | 0.00173 | 0.00173 | 0.00101 | 0.00109 | - |
| mass_tav_sup | kg | 0.546 | 0.864 | 0.864 | 0.504 | 0.546 | 0.514 |
| mass_tav_inf | kg | 0.672 | 0.864 | 0.864 | 0.504 | 0.546 | - |
| mass_tav_tot | kg | 5.838 | 6.912 | 6.912 | 5.040 | 4.368 | 7.716 |
| vol_tap | m3 | 0.000368 | 0.000608 | 0.000608 | 0.000394 | 0.000380 | - |
| mass_tap | kg | 0.184 | 0.304 | 0.304 | 0.197 | 0.190 | - |
| mass_tap_tot | kg | 1.654 | 2.734 | 2.734 | 1.772 | 1.710 | 0 |
| vol_tr | m3 | 0.00112 | 0.00115 | 0.00115 | 0.00128 | 0.000728 | - |
| mass_tr | kg | 0.560 | 0.576 | 0.576 | 0.638 | 0.364 | - |
| mass_tr_tot | kg | 1.680 | 1.728 | 1.728 | 1.913 | 1.09 | 0 |
| nc_el_sp | - | 18 | 27 | 27 | 18 | 18 | 0 |
| mass_c | kg | 0.189 | 0.243 | 0.243 | 0.196 | 0.171 | 0 |
| mass_wood | kg | 9.17 | 11.37 | 11.37 | 8.72 | 7.17 | 7.72 |
| mass_pallet | kg | 9.36 | 11.62 | 11.62 | 8.92 | 7.34 | 7.72 |
| dist_pond | km | 538.5 | 816.3 | 822.4 | 476.7 | 556.1 | 538.6 |

The LCIA results at midpoint level are shown in Table 6.20 for the 12 products analyzed.

**Table 6.20** *LCIA results for the further 12 products*

| Parameter | CC (kg $CO_2$ eq) | PMF (kg $PM_{10}$ eq) | HT (kg 1,4-DB eq) | FD (kg oil eq) | ALO ($m^2a$) |
|---|---|---|---|---|---|
| Reference product | 2.27 | 0.00558 | 1.070 | 0.612 | 68.9 |
| A | 1.84 | 0.00478 | 0.898 | 0.462 | 57.3 |
| B | 2.29 | 0.00564 | 1.120 | 0.587 | 66.6 |
| C | 2.11 | 0.00548 | 1.020 | 0.524 | 66.6 |
| D | 3.17 | 0.00822 | 1.390 | 0.826 | 83.1 |
| E | 1.80 | 0.00464 | 0.855 | 0.451 | 49.5 |
| F | 3.02 | 0.00781 | 1.330 | 0.776 | 72.1 |
| G | 1.90 | 0.00491 | 0.908 | 0.463 | 50.2 |
| H | 2.27 | 0.00558 | 1.070 | 0.612 | 68.9 |
| I | 2.86 | 0.00739 | 1.260 | 0.759 | 74.1 |
| L | 1.79 | 0.00462 | 0.891 | 0.433 | 47.8 |
| M | 1.57 | 0.00403 | 0.762 | 0.391 | 39.2 |
| N | 0.07 | 0.000211 | 0.0287 | 0.0169 | 2.8 |

## 6.3.2 Definition of linear correlations

From the analysis of the contribution analysis on the reference product it was possible to determine which are the input parameters most influencing each impact category.
Furthermore the application of the LCI parametric model to 13 products allowed to determine a correlation between the input parameter and the impact category, according to the definition of the most relevant parameters, which differ from impact category to impact category.
A quantitative linear regression of potential environmental impacts with regard to the input parameters was conducted.
For *Climate Change* the main influencing input parameters are *mass_wood* and *dist_pond*.

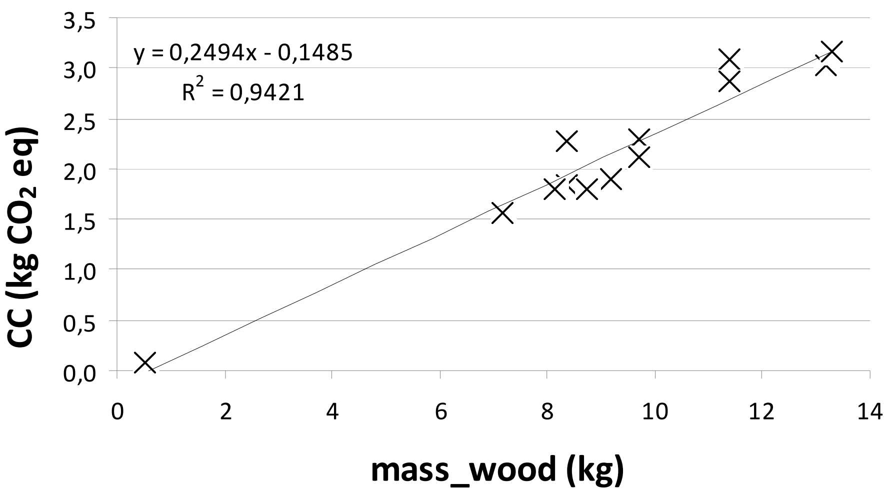


**Figure 6.3** *Linear regression for Climate change VS the mass of wood.*

The correlation with the mass of wood shows a coefficient of determination $R^2$ with a high value (0.942), therefore the regressor (*mass_wood*) defines with a good approximation the value of this environmental impact (Figure 6.3).
The same calculation was done considering the regressor *dist_pond* (Figure 6.4), but in this case the value of the coefficient of determination $R^2$ is lower (0.4559).

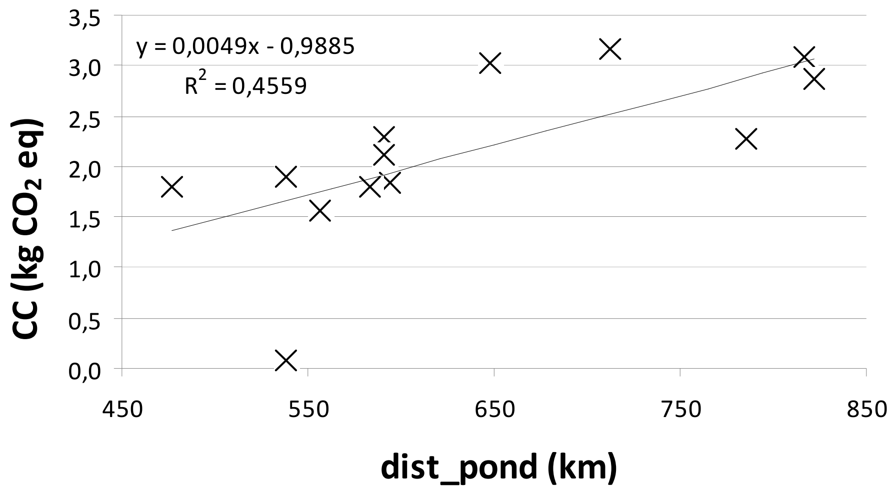


**Figure 6.4** *Linear regression for Climate change VS the average distance for wood transport.*

A new regressor is defined in Eq. 6.1 considering the product of the previous regressors:

$$trasp\_wood = mass\_wood \cdot dist\_pond\,, \tag{6.1}$$

where *trasp_wood* (tkm) is the amount connected with the transport of wood.
The new correlation is presented in Figure 6.5.

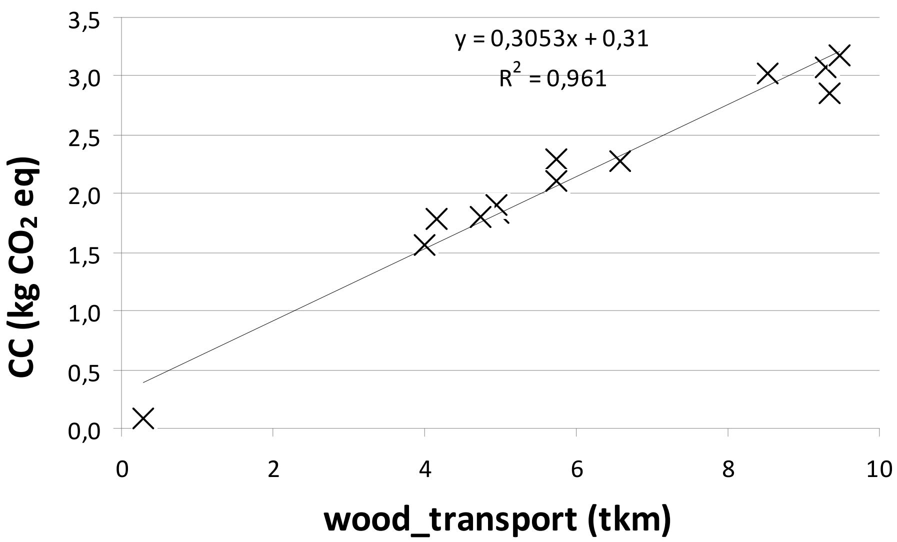


**Figure 6.5** *Linear regression for Climate change VS wood transport in tkm.*

The value of the coefficient of determination $R^2$ (0.961) has a higher value if compared with the one obtained with the correlation with *mass_wood*.
A similar correlation can be defined for *Particulate Matter Formation* and *Fossil Depletion*., as shown in Figures 6.6 and 6.7, respectively.

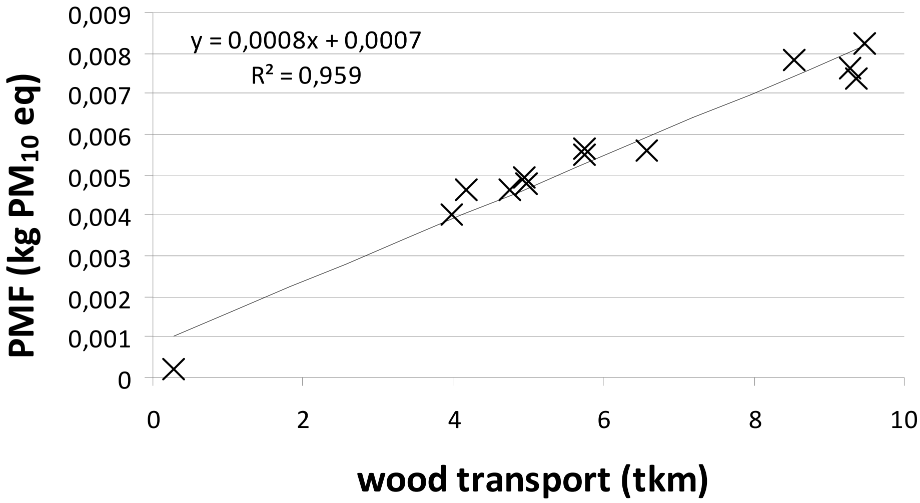


**Figure 6.6** *Linear regression for Particulate Matter Formation VS wood transport in tkm.*

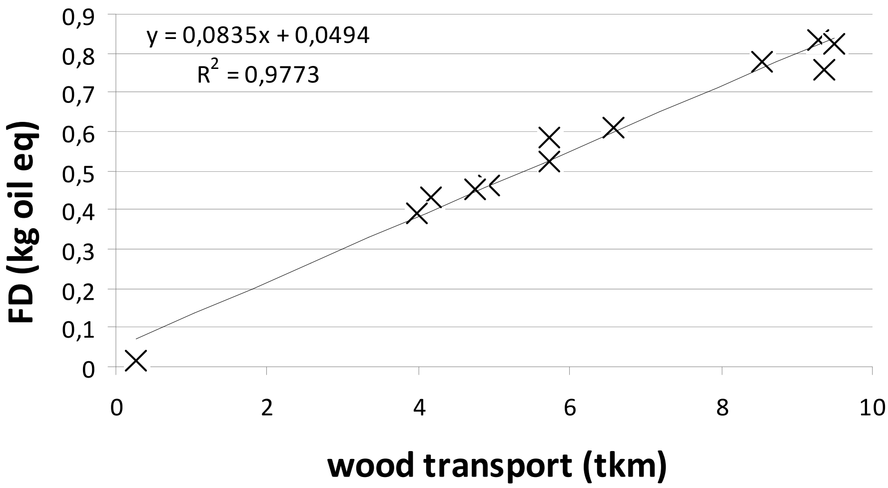


**Figure 6.7** *Linear regression for Fossil Depletion VS wood transport in tkm*

With regard to *Human Toxicity* the most relevant input parameter is *mass_wood* (Figure 6.8), as the contribution of impacts is much lower than the previous impact categories.

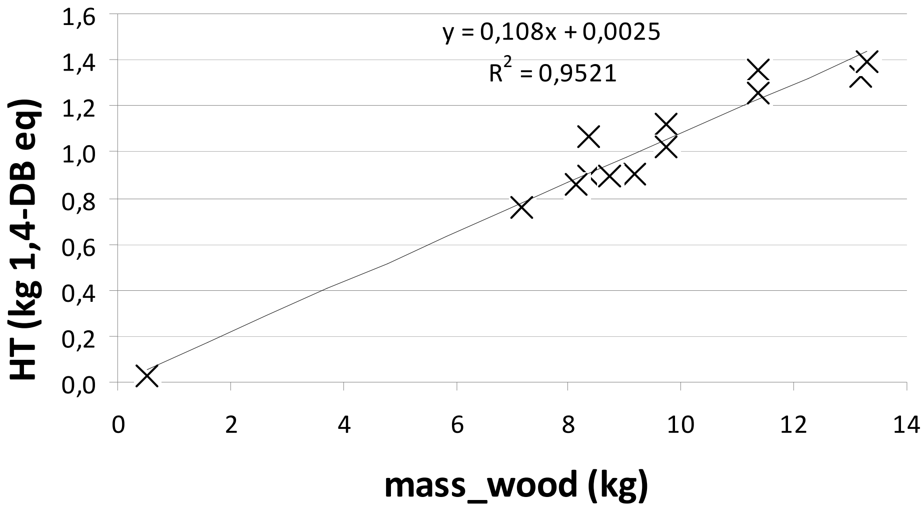


**Figure 6.8** *Linear regression for Human Toxicity VS mass wood.*

For the category *Agricultural Land Occupation,* the environmental impact is mainly connected with the production of wood, therefore the most influencing parameter is *mass_wood* (Figura 6.9).

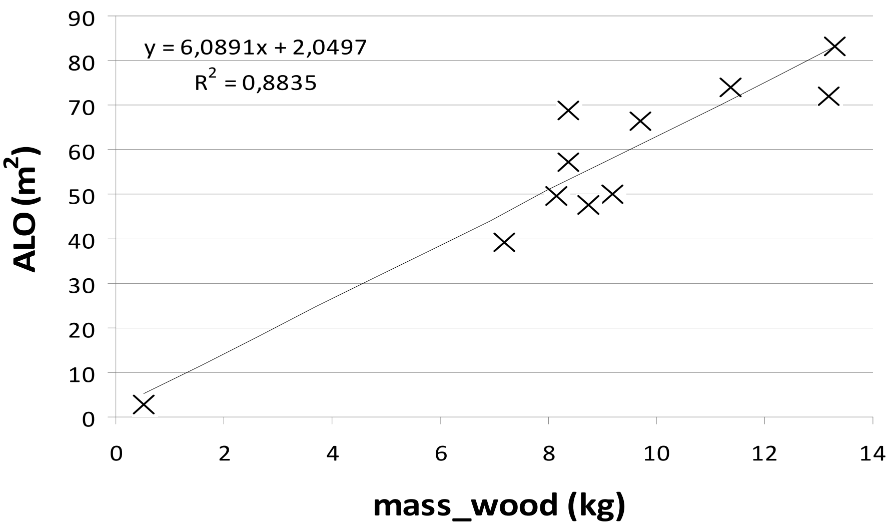


**Figure 6.9** *Linear regression for Agricultural Land Occupation VS mass wood.*

The regression presents a coefficient of determination much lower if compared with the values for the other impact categories, as the effect of the two types of wood (hardwood and softwood) has a different contribution.

In Figure 6.10 we considered the correlation between those products which use only softwood as raw materials (type F, G, L, M, N). In this case the value of the coefficient of determination is much better ($R^2$ = 0.999).

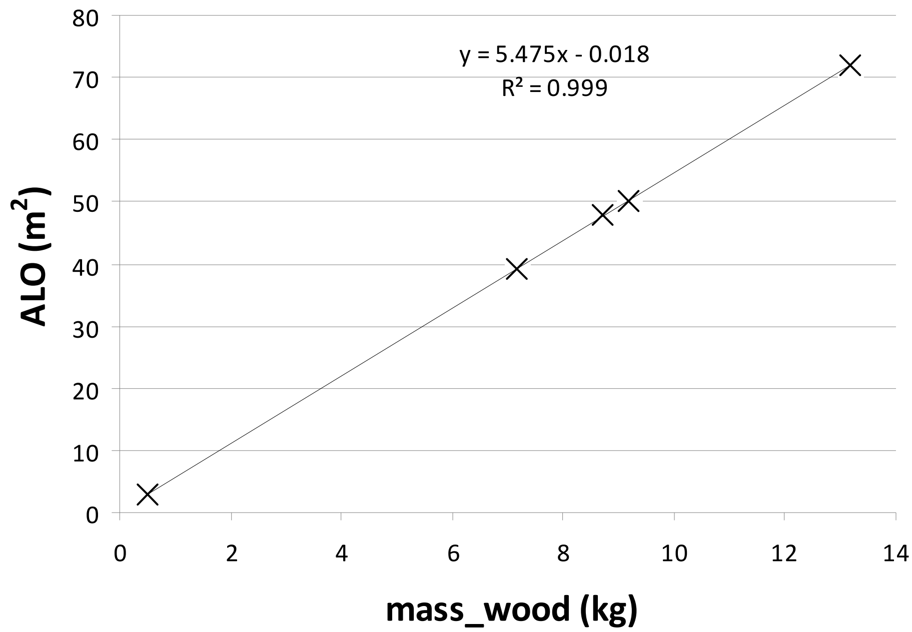


**Figure 6.10** *Linear regression for Agricultural Land occupation VS mass softwood.*

### *6.3.3 Definition of multiple non linear correlations*

The definition of correlations between input parameters and impact category indicator was done considering also multiple non linear relationship in order to consider the simultaneous influence of different parameters.

Three types of formula have been tested to get the fitting equation for quantifying the relationship of environment impact and corresponding parameters, as shown in Eq.6.2, Eq. 6.3 and Eq. 6.4:

$$y = a x_1^c x_2^d \tag{6.2}$$

$$y = a + bx_1 + cx_2 + dx_1x_2 + ex_1^2 + fx_2^2 \tag{6.3}$$

$$y = a + bx_1 + cx_2 + dx_1x_2 + ex_1^2 + fx_2^2 + gx_1^2x_2 + hx_1x_2^2 + ix_1^3 + jx_2^3 \tag{6.4}$$

The best fitting equation revealed to be Eq. 6.4, which can be used to predict the impact category indicator according to the variation of the two most influencing parameters, namely *mass_wood* and *dist_pond.*

The definition of the fitting equation was done with a Matlab code, reported in Appendix A, based on a multiple non linear regression (Petráš et al., 2010).

The relative error, as defined in Eq. 6.5, was calculated in order to test the effectiveness of the fitting equation if compared with the actual value given by the implementation of the LCI parametric model.

$$\text{Relative Error} = \frac{|\text{Actual Value} - \text{Calculated Value}|}{\text{Actual Value}} \quad (6.5)$$

In Table 6.21 are reported the value of the coefficient of the fitting equation according to the different impact category indicators considered.

**Table 6.21** *Coefficient of the fitting equation for the impact category indicators*

| | CC | PMF | HT | FD | ALO |
|---|---|---|---|---|---|
| a | 93.5412 | 0.1061 | 61.6127 | 30.6272 | 2631.6 |
| b | 11.8064 | -0.0110 | -7.7428 | -3.8059 | -194.3 |
| c | -0.2645 | -0.0003 | -0.1753 | -0.0881 | -10.2 |
| d | 0.0320 | 0 | 0.0212 | 0.0102 | 0.5 |
| e | 0.1857 | 0.0003 | 0.1158 | 0.0640 | 5.5 |
| f | 0.0002 | 0 | 0.0001 | 0.0001 | 0 |
| g | -0.0004 | 0 | -0.0002 | -0.0001 | 0 |
| h | 0 | 0 | 0 | 0 | 0 |
| i | 0 | 0 | -0.0006 | 0.0001 | -0.0002 |
| j | 0 | 0 | 0 | 0 | 0 |

In the following Tables the calculation of the impact category indicators according to the fitting equation and the calculated values, as well as the relative errors are reported.

**Table 6.22** *Calculation of impact category indicators Climate change and Particulate Matter formation with the fitting equation*

| | Actual value CC (kg $CO_2$ eq) | Calculated value CC (kg $CO_2$ eq) | Relative error CC (%) | Actual value PMF (kg $PM_{10}$ eq) | Calculated value PMF (kg $PM_{10}$ eq) | Relative error PMF (%) |
|---|---|---|---|---|---|---|
| Reference product | 2.27 | 2.2689 | 0.05 | 5.58E-03 | 5.6E-03 | 0.02 |
| A | 1.84 | 1.8600 | 1.09 | 4.78E-03 | 4.8E-03 | 0.60 |
| B | 2.29 | 2.2132 | 3.35 | 5.64E-03 | 5.6E-03 | 5.6 |
| C | 2.11 | 2.2132 | 4.89 | 5.48E-03 | 5.6E-03 | 1.71 |
| D | 3.17 | 3.1728 | 0.09 | 8.22E-03 | 8.2E-03 | 0.04 |
| E | 1.80 | 1.7841 | 0.89 | 4.64E-03 | 4.6E-03 | 0.63 |
| F | 3.02 | 3.0180 | 2.85 | 7.81E-03 | 7.8E-03 | 0.03 |
| G | 1.90 | 1.8623 | 1.98 | 4.91E-03 | 4.9E-03 | 0.75 |
| H | 2.27 | 2.9921 | 0.06 | 5.58E-03 | 5.6E-03 | |
| I | 2.86 | 2.9423 | 2.88 | 7.39E-03 | 7.5E-03 | 1.17 |
| L | 1.79 | 1.8023 | 0.68 | 4.62E-03 | 4.6E-03 | 0.26 |
| M | 1.57 | 1.5708 | 0.05 | 4.03E-03 | 4.0E-03 | 0.11 |
| N | 1.15 | 1.151 | 0.04 | 3.16E-03 | 3.1E-03 | 0 |

**Table 6.23** *Calculation of impact category indicators Human* Toxicity and Fossil depletion with the fitting equation

| | Actual value HT (kg 1,4DBeq) | Calculated value HT (kg 1,4DBeq) | Relative error HT (%) | Actual value FD (kg oil eq) | Calculated value FD (kg oil eq)) | Relative error FD (%) |
|---|---|---|---|---|---|---|
| Reference product | 1.070 | 1.0696 | 0.04 | 0.612 | 0.6116 | 0.06 |
| A | 0.898 | 0.8974 | 0.07 | 0.462 | 0.4685 | 1.40 |
| B | 1.120 | 1.0746 | 4.05 | 0.587 | 0.5601 | 4.59 |
| C | 1.020 | 1.0746 | 5.36 | 0.524 | 0.5601 | 6.88 |
| D | 1.390 | 1.3908 | 0.06 | 0.826 | 0.8270 | 0.12 |
| E | 0.855 | 0.8614 | 0.75 | 0.451 | 0.4462 | 1.06 |
| F | 1.330 | 1.3296 | 0.03 | 0.776 | 0.7753 | 0.09 |
| G | 0.908 | 0.8930 | 1.65 | 0.463 | 0.4498 | 2.85 |
| H | 1.070 | 1.068 | 2.32 | 0.612 | 0.608 | 3.66 |
| I | 1.260 | 1.2896 | 2.35 | 0.759 | 0.7876 | 3.77 |
| L | 0.891 | 0.8958 | 0.53 | 0.433 | 0.4373 | 0.99 |
| M | 0.762 | 0.7591 | 0.38 | 0.391 | 0.3911 | 0.02 |
| N | 0.0287 | 0.0287 | 0.16 | 0.0169 | 0.0169 | 0.07 |

**Table 6.24** *Calculation of impact category indicator Agricultural Land Occupation*

| | Actual value ALO ($m^2a$) | Calculated value ALO ($m^2a$) | Relative error ALO(%) |
|---|---|---|---|
| Reference product | 68.9 | 68.9259 | 0.04 |
| A | 57.3 | 55.8750 | 2.49 |
| B | 66.6 | 66.2530 | 0.52 |
| C | 66.6 | 66.2530 | 0.52 |
| D | 83.1 | 83.0060 | 0.11 |
| E | 49.5 | 51.3334 | 3.70 |
| F | 72.1 | 72.1886 | 0.12 |
| G | 50.2 | 50.9802 | 1.55 |
| H | 68.9 | 72.012 | 3.02 |
| I | 74.1 | 72.0196 | 2.81 |
| L | 47.8 | 47.5324 | 0.56 |
| M | 39.2 | 38.8031 | 1.01 |
| N | 2.8 | 2.8035 | 0.12 |

## 6.4 Conclusions

The present study focused on the definition of a parametric LCI model in order to calculate inventory data from the quantification of parameters describing the life cycle of a range of wooden pallet. The identified parameters refer to technical characteristics of the product system, i.e. number and dimension of elements constituting the wooden pallet, and were divided into two categories: independent and dependent parameters. The definition of quantitative relations between inventory data can save time during the conduction of multiple LCA studies, as the parametric LCI model can be directly applied to calculate the potential environmental impact of any other wooden pallets manufactured by the company.

The parametric LCI model was tested with one reference product, namely one non reversible pallet with 4-way blocks manufactured by the company, and the input and output flows were quantified according to eleven life cycle stages. The LCIA step included a selection of impact categories: climate change, particulate matter formation, human toxicity, agricultural land occupation and fossil depletion. From the analysis of the LCIA results, the most relevant life cycle stages were identified: raw materials extraction and transport (wood and nails) and the end of life. Furthermore the parameter influencing each impact categories were identified as follows:

- *mass_wood* (the total mass of wood in the pallet) for Human Toxicity, Agricultural Land Occupation and Climate Change;
- *wood_transport* (the averaged transport of the wood, given by the product of *mass_wood* and *dist_pond*, which is the weighted distance for the transport of the wood components) for Particulate Matter Formation and Fossil Depletion.

Once detected the most influencing parameters it was possible to apply the parametric LCI model to further 12 products. This allowed to define correlation between the most influencing LCI parameter and the corresponding impact category score. The correlation was defined both with linear regression and multiple non linear regression.

The definition of mathematical correlation between inventory data and environmental impacts can be used as a support in product design and development. Indeed, the impact of a new product with a defined amount of mass wood and/or a specific supply of wood can be estimated by the correlations, providing a rough quantification of the environmental impact before starting a full LCA application.

## 6.5 References

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# Chapter 7

# Conclusions and future perspectives

The research activity was concerned with the application of the Life Cycle Assessment methodology in order to quantify the potential environmental impacts of different products and processes at industrial level.
LCA methodology was applied to different case studies with the aim to define solutions for improving the reliability of LCA studies and the applicability of the methodology with regard to its typical applications at industrial level: product improvement, benchmarking between competing products, processes or technological solutions providing the same function and product design and conceptualization.
The quantification of the uncertainty of LCA final results is a crucial issue for the reliability of the methodology and an important step toward reliable and transparent decision support. With regard to this aspect, three case studies were investigated, where LCA methodology was used for comparing different farming techniques, tissue paper wipers and wastewater treatment options.
When different options are compared it is important to quantify the influence of data variability on the final results, therefore the contribution of uncertainty analysis and sensitivity analysis revealed to be essential for the interpretation phase.
A 4-step procedure for the quantification of parameter uncertainty using a combined qualitative and stochastic method was defined and applied in the first two cases. The qualitative analysis with the use of the Pedigree matrix transformed the data quality indicators to probability distributions by representing the data quality indicator value by a "default" lognormal distribution, therefore providing a qualitative judgement on inventory data. Secondly, the quantitative uncertainty analysis (geometric standard deviation quantification and Monte Carlo simulation) quantified how data quality translated into uncertainty level on the Life Cycle Impact Assessment results. The implementation of the procedure allowed to test the robustness of the results.
In the case of the comparative LCA between conventional and organic farming of soybean and barley in a three year cycle the uncertainty analysis confirmed the LCIA results at endpoint level: no unambiguous comparative assertion between the two farming systems can be stated. The conventional cycle revealed better performances for ecosystems, due to the higher yield per hectare, meanwhile organic farming showed better performance for

resources, due to the lesser fossil fuels consumption. For human health there is no significant difference. Furthermore the implementation of a sensitivity analysis to assess the influence of the spring rainfall index as an important geo-climatic factor in LCIA results allowed to define a variable range of LCIA, with the inclusion of two borderline scenarios, namely high rainfall and drought conditions. Ultimately, this application confirmed that the annual variation in agricultural production should be considered when comparing organic and conventional farming.

For the comparative LCA between tissue paper wipers manufactured with different raw materials the results of impact assessment showed that wipers made by virgin pulp have the greatest environmental impacts for all the impact categories considered, except for climate change for which virgin fiber wipers and waste paper wipers have the same impact. The results of the uncertainty analysis confirmed these results with a level of statistical significance of 100% for all impact categories and all combinations of comparison between the three products. Only for climate change the level of statistical significance is lower, therefore confirming that for this impact category there is no actual difference between virgin fibres and waste paper. The sensitivity analysis, changing the LCIA method (Impact 2002+ instead of Recipe 2008) confirmed the results. As a general conclusion, if the goal of the LCA is to make an environmental claim regarding the superiority of one product versus another product with the same function, the role of both sensitivity and uncertainty analysis should not be neglected, even if the products are manufactured by the same company.

The 4-step procedure for uncertainty analysis can be easily applied to other sectors, as it does not require further data collection. Therefore it is fully transferable to any other industrial sector.

Moreover, we applied the LCA methodology at process level, namely to evaluate the environmental performances of four different technological solutions representative of Danish WWTPs, by taking into account different end of life options for sludge disposal. The ranking between the different plant types changed according to the impact categories considered. According to the actual scenario big plants equipped with anaerobic sludge digestion performed better than small aerobic plants for climate mitigation (climate change and fossil depletion) and terrestrial ecotoxicity impact categories. Instead, for both freshwater and marine eutrophication and other toxicity-related categories (human toxicity, freshwater ecotoxicity, marine ecotoxicity) they show higher environmental impacts. An important contribution to the reduced impact of small plants equipped with aerobic sludge digestion is due to the sludge application on agricultural soil. This is confirmed by the results of the sensitivity analysis on final sludge treatment for plants equipped with anaerobic digestion. A further sensitivity analysis, changing the LCIA method, proved the validity of the LCIA results for climate mitigation, meanwhile toxicity-related scores confirmed to be sensitive to the choice of the impact assessment method. Finally, the influence of input data variability tested with the uncertainty analysis via Monte Carlo simulation showed that the impact categories with highest uncertainty (freshwater eutrophication and marine ecotoxicity) are those highly dependent on the effluent emissions. As the variability of water and sludge quality parameters greatly affect LCIA results, in the context of WWTP LCA, it is important to use measured data for modeling effluent and sludge quality data, including an assessment of the influence of their variability.

From the three case studies, we conclude that LCAs lacking explicit interpretation of the degree of uncertainty and sensitivities are of limited value as robust evidence for decision making or comparative assertions. Further exploration of methodological aspects in data quality analysis for LCI and LCA studies is needed in order to enable clear demonstration of how aspects of representativeness, robustness and confidence in inventory data and LCIA are handled within a given LCA study.

With regard to the second aspect connected with increasing the applicability of the LCA methodology, the focus was given to the optimization of data collection, discussing the use of streamlined techniques.

Firstly, we tested the effectiveness of a LCI indicator, such as non-renewable fossil Cumulative Energy Demand, as a screening indicator within the milk packaging sector in order to detect those situations in which beverage companies can benefit from the use of this proxy indicators before starting a full LCA application. Starting from a case study of two milk containers (HDPE bottle and laminated carton container), we concluded that: in the beverage packaging sector, the use of the screening LCI indicator non-renewable fossil CED can be useful to determine which is the alternative with the lowest environmental impact, but can lead to misleading decisions when assessing the contribution of life cycle stages to the overall impact. For product improvement also climate change impact category should be considered. Therefore companies in the beverage packaging sector can benefit for the use of non renewable fossil CED if their aim is to obtain a preliminary estimation of the life cycle environmental impacts of two or more competing products. As a consequence, non renewable fossil CED and climate change can be used together to provide a rough estimation of the environmental profile in the beverage packaging sector. It could be interesting to test if this correlation is valid also within other sectors, where the energy aspect is relevant. Additional research is also required to define which unit process could be safely omitted while analyzing or comparing different beverage packaging alternatives without greatly affecting the results.

Secondly, as product development support tool, we implemented a parametric model to describe the life cycle inventory of a wooden pallet through the definition of independent and dependent parameters. The LCI parametric model was tested to calculate the potential environmental impacts of a reference product, i.e. a non reversible pallet with 4-way blocks. From the analysis of the LCIA results, the most relevant life cycle stages were identified, as well as the parameters which greatly affect the score of each impact categories considered.

Moreover the LCI parametric model was applied to calculate the LCIA of further 12 products. Correlations, both with linear regression and multiple non linear regression, were defined between the most influencing LCI parameter and the corresponding impact category score.
The defined correlations can be used in the design of new products, providing a rough quantification of the environmental impact before starting a full LCA application. Finally, this approach can be replicated in other industrial sectors if the products manufactured by the company have similar characteristics.

# List of Publications

## International Journals

Published:

- Scipioni A., Niero M., Mazzi A., Manzardo A., Piubello S., 2012. Significance of the use of non-renewable fossil CED as proxy indicator for screening LCA in the beverage packaging sector. International Journal of Life Cycle Assessment, DOI: 10.1007/s11367-012-0484-x

Submitted:

- Fedele A., Niero M., Mazzi A., Zuliani F., Scipioni A., 2012.Comparative LCA case study of organic and conventional barley and soybean, including a sensitivity analysis based on a spring rainfall index. Journal of Cleaner Production.
- Toniolo S., Niero M., Mazzi A., Zuliani F., Scipioni A., 2012. Recycling in food packaging sector: from waste bottles to tray production. A case study. Resources, Conservation and Recycling.
- Manzardo A., Niero M., Rettore L., Scipioni A., 2012. Water pay-back time indicator to support decision making on water saving technologies. Ecological Indicators.
- Niero M., Pizzol M, Bruun, HG, Tychsen P, Thomsen M. 2013. Comparative life cycle assessment of multifunctional wastewater treatment in Denmark. Journal of Cleaner Production

To be submitted:

- Niero M., Mazzi A., Di Felice F., Scipioni A. Parametrization in the Life Cycle Inventory: a case study of wooden pallet. International Journal of life Cycle Assessment.
- Thomsen M., Niero M., Pizzol M., Bruun H.G., Hjelgaard K., Tychsen P., Martinsen L., Smart J., Hasler B. Sludge as a multifunctional resource – recycling of phosphorous from Danish wastewater treatment plants to agricultural soils. Journal of Cleaner Production

## Publications/abstracts in International conference proceedings

Niero M., Manzardo A., Scapinello A., Scipioni A., 2012. Sensitivity and uncertainty analysis to support raw materials selection for tissue paper: a case study of different rolls from virgin and recycled materials. In: *Abstract book SETAC Europe 18th Case study Symposium*. Copenhagen, Denmark, 26-28 November 2012, pp. 70-71

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# Appendix A

In this Appendix the Matlab program used to calculate the fitting equations in Chapter 6 is shown.

```
function regiss3()%The regission of the function
Y1=input('Input the Vector of Y1:');%Input the Vector of the dependent
variables
X1=input('Input the Vector of X1:');%Input the first group of
parameters(independent variables)
X2=input('Input the Vector of X2:');%Input the second group of
parameters(independent variables)
n=length(Y1);X1X2=[];X12=[];X22=[];X12X2=[];X1X22=[];X13=[];X23=[];
for i=1:n
    X1X2(i)=X1(i).*X2(i);
    X12(i)=X1(i).^2;
    X22(i)=X2(i).^2;
    X12X2(i)=(X1(i).^2).*X2(i);
    X1X22(i)=(X2(i).^2).*X1(i);
    X13(i)=X1(i).^3;
    X23(i)=X2(i).^3;
end
nX1=X1';nX2=X2';nX1X2=X1X2';nX12=X12';nX22=X22';nX12X2=X12X2';nX1X22=X1X22'
;nX13=X13';nX23=X23';
X=[ones(n,1),X1',X2',X1X2',X12',X22',X12X2',X1X22',X13',X23'];
Y=Y1';
P=regress(Y,X),
y1=[];
for i=1:n
y1(i)=P(1,1)+P(2,1)*X1(i)+P(3,1)*X2(i)+P(4,1)*(X1(i).*X2(i))+P(5,1)*(X1(i).
^2)+P(6,1)*(X2(i).^2)+P(7,1)*((X1(i).^2).*X2(i))+P(8,1)*((X2(i).^2).*X1(i))
+P(9,1)*(X1(i).^3)+P(10,1)*(X2(i).^3);
end
Y1%The actual values of the dependent variables
y1%The values calculated by the fitting equation
abs(y1-Y1)%The absolute error of the calculated values to the actual values
abs(y1-Y1)./Y1%The relative error
```

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