DOKUZ EYLUL UNIVERSITY GRADUATE SCHOOL OF
NATURAL AND APPLIED SCIENCES

LIFE CYCLE ASSESSMENT OF WASTEWATER TREATMENT PLANTS

by
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October, 2009
İZMİR

LIFE CYCLE ASSESSMENT OF WASTEWATER TREATMENT PLANTS

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M.Sc. THESIS EXAMINATION RESULT FORM

We have read the thesis entitled **"LIFE CYCLE ASSESSMENT OF WASTEWATER TREATMENT PLANTS"** completed by **YASEMİN ÜN** under supervision of **ASSIST. PROF. DR. SEVGİ TOKGÖZ GÜNEŞ** and we certify that in our opinion it is fully adequate, in scope and in quality, as a thesis for the degree of Master of Science.

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LIFE CYCLE ASSESSMENT OF WASTEWATER TREATMENT PLANTS

ABSTRACT

The fact that the importance of environmental protection and its possible effects on the environment is gaining importance has also raised the concern for the better methods which will improve and make the situation more comprehensible. Life Cycle Assessment (LCA) is one of the techniques improved for these purposes. With raising human population and the increasing industrialization, there has been very substantial increase in waste products too. Nowadays with the increasing issue of the importance of the treatment of waste waters, domestic and industrial waste waters produced by production activities has gaining more attention in terms of the ecological balance. In this study, before applying life cycle assessment methodology to a system, a study was conducted which focused on the comprehension of the methods which are used for life cycle assessment and evaluation of environmental effects. In the proceeding sections of this study, Life Cycle Assessment method was used for evaluating the environmental advantages and expenses of other different wastewater treatment technologies and standards. An inventory of the input (chemical substances used, electrical energy etc.) and output (emissions releases into the water, earth and the air, amount of sludge etc.) of the plants where waste water is refined was documented, potential environmental effects of the input and the output was assessed, finally the obtained results were interpreted with regard to the objectives of this study. With regard to these studies, attention was drawn to the importance of wastewater treatment plants which are regularly managed. Utilization of resources about treatment systems and its effects on human health and ecology were assessed, finally the most suitable methods of wastewater treatment methods were tried to be explained with best examples.

Keywords: Life Cycle Assessment, Wastewater Treatment, Wastewater Treatment Methods and Environmental Effects.

ATIKSU ARITMA TESİSLERİNDE YAŞAM DÖNGÜSÜ ANALİZİ

ÖZ

Çevrenin korunmasının ve üretilen ürünlerin çevre üzerindeki muhtemel etkilerinin öneminin gittikçe daha iyi bir şekilde fark edilmekte oluşu, bu etkilerin daha iyi bir şekilde kavranıp anlaşılması ve azaltılması için metotların geliştirilmesi konusuna duyulan ilgiyi de arttırmıştır. Bu amaçla geliştirilen tekniklerden birisi de Yaşam Döngüsü Analizidir (YDA). Dünyada insan nüfusunun ve sanayileşmenin artması ile birlikte buna bağlı olarak da atık oluşumunda da artış gözlenmektedir. Üretim faaliyetleri sonucu oluşan evsel ve endüstriyel atıksular, atııksu artımının önem kazanması ile çevre dengesi açısından günümüzde daha fazla gündeme gelmeye başlamıştır. Bu çalışmada bir sistem için yaşam döngüsü analizi metodolojisi uygulanmadan önce YDA metodolojisinin temelini kavrama ve çevresel etkileri değerlendirilirken kullanılan yöntemlerin anlaşılması çalışması yapılmıştır. Çalışmanın ilerleyen bölümlerinde YDA metodu farklı atıksu arıtma teknoloji ve standartların çevresel masraflarını ve yararlarını değerlendirmek için kullanılmıştır. Atıksuların arıtıldığı tesislerin girdi (kullanılan kimyasal madde, elektrik enerjisi v.b.) ve çıktılarının (havaya, suya ve toprağa verilen emisyonlar, oluşan çamur miktarı v.b.) bir envanteri yapılmış, girdi ve çıktılarla ilgili muhtemel çevre etkileri değerlendirilmiş ve elde edilen sonuçlar çalışmanın amaçları ile bağlantılı bir şekilde yorumlanmıştır. Bu çalışmalar sonucunda iyi işletilmesi gereken atıksu arıtma tesislerinin gerekliliğine dikkat çekilmiştir. Arıtma sistemleri ile ilgili kaynakların kullanımı, insan sağlığı ve ekolojiye etkileri değerlendirilerek atıksu arıtımında kullanılabilecek en iyi yöntemler örneklerle açıklanmaya çalışılmıştır.

Anahtar Kelimeler: Yaşam Döngüsü Analizi, Atıksu Arıtımı, Atıksu Arıtma Yöntemleri ve Çevresel Etkiler.

CONTENTS

CHAPTER ONE
LIFE CYCLE ASSESSMENT

1.1 Introduction

The complex interaction between a product and the environment is dealt with in the Life Cycle Assessment (LCA) method. It is also known as Life Cycle Analysis or Ecobalance. LCA systematically describes and assesses all flows to and from nature, from a cradle to grave perspective (Curran, 2005).

LCA is the process of analyzing a product's environmental impact - energy and material use, water, air and soil contamination - during its whole product life cycle from 'cradle to grave'. This analysis includes the different phases of resources extraction, production, distribution, use and consumption, and disposal. ISO is currently developing LCA draft standards that define general requirements for conducting LCA’s and reporting their results. The purpose of LCA is to pin-point specific stages in a life cycle which contribute significantly to the burden on the environment. Hence, improvements in these stages would yield the greatest benefit to the environment. At its simplest level, a LCA study can be a listing of the environmental outputs pertaining to a product or process.

LCA is important in decision-making when choosing alternative raw materials and recycling strategies. Without LCA, such decisions could unwittingly cause adverse effects to the environment, as an improvement at one stage may result in an increased environmental burden at other stages. An example is disposable diapers which were thought to be environmentally-friendly, but studies show that they do not biodegrade easily when buried deep in landfills.

A product's life cycle starts when raw materials are extracted from the earth, followed by manufacturing, transport and use, and ends with waste management including recycling and final disposal. At every stage of the life cycle there are emissions and consumption of resources. The environmental impacts from the entire life cycle of products and services need to be addressed. To do this, life cycle thinking is required (ISO 14040).

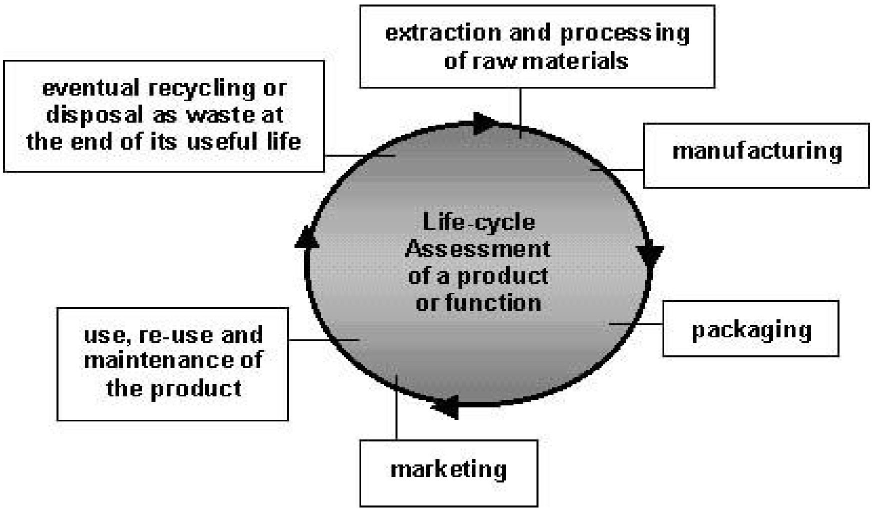

Figure 1.1 The phases of life cycle assessment (according to ISO 14040).

Life Cycle Assessment (LCA) is a tool for the systematic evaluation of the environmental aspects of a product or service system through all stages of its life cycle. LCA provides an adequate instrument for environmental decision support. Life cycle assessment has proven to be a valuable tool to document the environmental considerations that need to be part of decision-making towards sustainability. A reliable LCA performance is crucial to achieve a life-cycle economy. There are two main steps in a LCA (Curan, 2005).

1. Describe which emissions will occur and which raw materials are used during the life of a product. This is usually referred to as the **inventory** step.

2. Assess what the impacts of these emissions and raw material depletions are. This is referred to as the **impact assessment** step.

LCA is a quantitative environmental performance tool, essentially based around mass and energy balances but applied to a complete economic system rather than a single process. In terms of the system boundary definition, this represents an extension to the conventional system analysis, in which the system boundary is drawn around the process of interest only. Figure 1.2 illustrates the way in which LCA can complement conventional process analysis. While chemical or process engineering is normally concerned with the operations within system boundary 1, LCA considers the whole material and energy supply chains, so that the system of concern becomes everything within system boundary 2. The material and energy flows that enter, exist in or leave the system include material and energy resources and emissions to air, water and land. These are often referred to as environmental burdens and they arise from activities encompassing extraction and refining of raw materials, transportation, production, use and waste disposal of a product or process. The potential effects of the burdens on the environment, i.e. environmental impacts, normally include global warming potential (GWP), acidification, ozone depletion (OD), eutrophication etc. The LCA methodology is still under development. At present, the methodological framework comprises four phases (Azapagic & Clift, 1999):

1. **Goal and scope definition**, the product(s) or service(s) to be assessed are defined, a functional basis for comparison is chosen and the required level of detail is defined.

2. **Inventory analysis,** the energy carriers and raw materials used, the emissions to atmosphere, water and soil, and different types of land use are quantified for each process, then combined in the process flow chart and related to the functional basis.

3. **Impact assessment**, the effects of the resource use and emissions generated are grouped and quantified into a limited number of impact categories which may then be weighted for importance.

4. **Interpretation**, the results are reported in the most informative way possible and the need and opportunities to reduce the impact of the product(s) or service(s) on the environment are systematically evaluated.

Applied to process analysis, LCA can have two main objectives. The first is to quantify and evaluate the environmental performance of a process from 'cradle to grave' and so help decision-makers to choose a more sustainable option among alternatives. Another objective is to provide a basis for assessing potential improvements in the environmental performance of a system. Two main problems are associated with these objectives of LCA. First, in many cases there will be a number of options and possibilities for improvements and it may not always be obvious which of them represents the optimum solution. Therefore, some kind of system optimization will be necessary. Secondly, there may exist more than one optimum solution for improving the system's performance, in which case the issue becomes that of choosing the best compromise option from a number of optimum solutions (Azapagic & Clift, 1999).

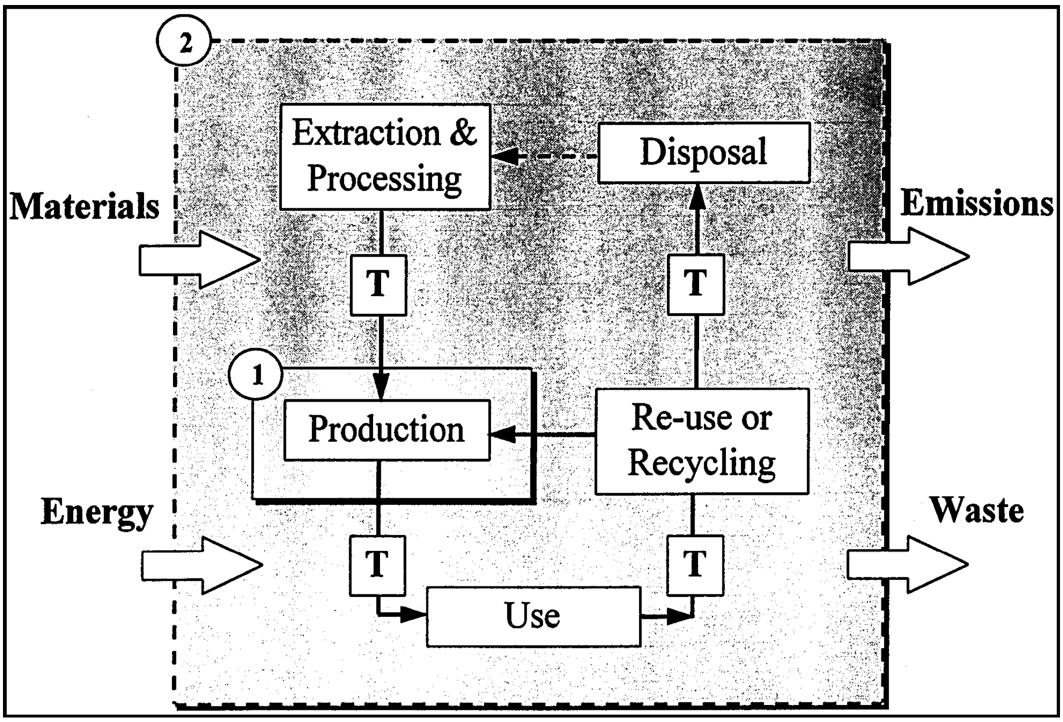

Figure 1.2 Stages in the life cycle of a product (system boundary: 1, process analysis; 2, life cycle assessment; T, transport).

1.2 History of LCA

Life Cycle Assessment (LCA) had its beginnings in the 1960's. Concerns over the limitations of raw materials and energy resources sparked interest in finding ways to cumulatively account for energy use and to project future resource supplies and use. In one of the first publications of its kind, Harold Smith reported his calculation of cumulative energy requirements for the production of chemical intermediates and products at the World Energy Conference in 1963.

Later in the 1960's, global modeling studies published in The Limits to Growth (Meadows et al 1972) and A Blueprint for Survival (Goldsmith et al 1972) resulted in predictions of the effects of the world's changing populations on the demand for finite raw materials and energy resources. The predictions for rapid depletion of fossil fuels and climatologically changes resulting from excess waste heat stimulated more detailed calculations of energy use and output in industrial processes. During this period, about a dozen studies were performed to estimate costs and environmental implications of alternative sources of energy (Curran, 2006).

In 1969, researchers initiated an internal study for The Coca-Cola Company that laid the foundation for the current methods of life cycle inventory analysis in the United States. In a comparison of different beverage containers to determine which container had the lowest releases to the environment and least affected the supply of natural resources, this study quantified the raw materials and fuels used and the environmental loadings from the manufacturing processes for each container. Other companies in both the United States and Europe performed similar comparative life cycle inventory analyses in the early 1970's. At that time, many of the available sources were derived from publicly-available sources such as government documents or technical papers, as specific industrial data were not available. The process of quantifying the resource use and environmental releases of products became known as a Resource and Environmental Profile Analysis (REPA), as practiced in the United States. In Europe, it was called an Ecobalance. With the formation of public interest groups encouraging industry to ensure the accuracy of information in the public domain, and with the oil shortages in the early 1970's, approximately 15

REPAs were performed between 1970 and 1975. Through this period, a protocol or standard research methodology for conducting these studies was developed. This multi-step methodology involves a number of assumptions. During these years, the assumptions and techniques used underwent considerable review by EPA and major industry representatives, with the result that reasonable methodologies were evolved.

From 1975 through the early 1980's, as interest in these comprehensive studies waned because of the fading influence of the oil crisis, environmental concerns shifted to issues of hazardous and household waste management. However, throughout this time, life cycle inventory analysis continued to be conducted and the methodology improved through a slow stream of about two studies per year, most of which focused on energy requirements. During this time, European interest grew with the establishment of an Environment Directorate (DG X1) by the European Commission. European LCA practitioners developed approaches parallel to those being used in the USA. Besides working to standardize pollution regulations throughout Europe, DG X1 issued the Liquid Food Container Directive in 1985, which charged member companies with monitoring the energy and raw materials consumption and solid waste generation of liquid food containers.

When solid waste became a worldwide issue in 1988, LCA again emerged as a tool for analyzing environmental problems. As interest in all areas affecting resources and the environment grows, the methodology for LCA is again being improved. A broad base of consultants and researchers across the globe has been further refining and expanding the methodology. The need to move beyond the inventory to impact assessment has brought LCA methodology to another point of evolution (SETAC 1991; SETAC 1993; SETAC 1997).

In 1991, concerns over the inappropriate use of LCAs to make broad marketing claims made by product manufacturers resulted in a statement issued by eleven State Attorneys General in the USA denouncing the use of LCA results to promote products until uniform methods for conducting such assessments are developed and a consensus reached on how this type of environmental comparison can be advertised non-deceptively. This action, along with pressure from other environmental organizations to standardize LCA methodology, led to the development of the LCA standards in the International Standards Organization (ISO) 14000 series (1997 through 2002).

In 2002, the United Nations Environment Programme (UNEP) joined forces with the Society of Environmental Toxicology and Chemistry (SETAC) to launch the Life Cycle Initiative, an international partnership. The three programs of the Initiative aim at putting life cycle thinking into practice and at improving the supporting tools through better data and indicators. The Life Cycle Management (LCM) program creates awareness and improves skills of decision-makers by producing information materials, establishing forums for sharing best practice, and carrying out training programs in all parts of the world. The Life Cycle Inventory (LCI) program improves global access to transparent, high quality life cycle data by hosting and facilitating expert groups whose work results in web-based information systems. The Life Cycle Impact Assessment (LCIA) program increases the quality and global reach of life cycle indicators by promoting the exchange of views among experts whose work results in a set of widely accepted recommendations (Curran, 2006).

1.3 Why Use LCA?

Governments and your customers simply expect that companies pay attention to the environmental properties of all products. EMAS, BS and ISO 14000 series demand continuous improvement in your environmental management system. LCA and its utilization for product/process improvement is the way to meet this demand.

The LCA methodology is described in detail by SETAC and CML (University of Leiden). In SETAC's Code of Practice, it is recommended that the LCA be split into five stages (Curran, 2005).

1. Planning

- statement of objectives
- definition of the product and its alternatives

- choice of system boundaries
- choice of environmental parameters
- choice of aggregation and evaluation method
- strategy for data collection

2. Screening

- Preliminary execution of the LCA
- Adjustment of plan

3. Data collection and data treatment

- Measurements, interviews, literature search, theoretical calculations, database search, qualified guessing
- Computation of the inventory table

4. Evaluation

- Classification of the inventory table into impact categories
- Aggregation within the category (characterization)
- Normalization
- Weighting of different categories (valuation)

5. Improvement assessment

- Sensitivity analysis
- Improvement priority and feasibility assessment

It is generally recognized that the first stage is extremely important. The result of the LCA is heavily dependent on the decisions taken in this phase. The screening LCA is a useful step to check the goal-definition phase. After screening it is much easier to plan the rest of the project.

SimaPro can be a very convenient tool for both screening LCA's and full LCA's. With software tool like SimaPro the border is actually rather vague. A screening LCA gradually becomes a full LCA as more data are entered. SimaPro comes with a large inventory database and several impact assessment methods (Curran, 2005).

1.4 Limitations of Conducting a LCA

The whole techniques are dependent on some limitations. Therefore the understood of the present limitations of LCA is important (EPA. 2001).

The main limitations are;

- Develop a systematic evaluation of the environmental consequences associated with a given product.
- Analyze the environmental trade-offs associated with one or more specific products/processes to help gain stakeholder (state, community, etc.) acceptance for a planned action.
- Quantify environmental releases to air, water, and land in relation to each life cycle stage and/or major contributing process.
- Assist in identifying significant shifts in environmental impacts between life cycle stages and environmental media.
- Assess the human and ecological effects of material consumption and environmental releases to the local community, region, and world.
- Compare the health and ecological impacts between two or more rival products/processes or identify the impacts of a specific product or process.
- Identify impacts to one or more specific environmental areas of concern.

Performing a LCA can be resource and time intensive. Depending upon how thorough an LCA the users wish to conduct, gathering the data can be problematic, and the availability of data can greatly impact the accuracy of the final results. Therefore, it is important to weigh the availability of data, the time necessary to conduct the study, and the financial resources required against the projected benefits of the LCA.

LCA will not determine which product or process is the most cost effective or works the best. Therefore, the information developed in a LCA study should be used as one component of a more comprehensive decision process assessing the trade-offs with cost and performance (EPA, 2001).

1.5 Key Features of LCA

Some major key-features of the LCA methodology are summarized (ISO 14001):

- LCA studies should systematically and adequately address the environmental aspects of product systems, from raw material acquisition to final disposal.
- The depth of detail and time frame of a LCA study may vary to a large extent, depending on the definition of goal and scope.
- The scope, assumptions, description of data quality, methodologies and output of LCA studies should be transparent. LCA studies should discuss and document the data sources, and be clearly and appropriately communicated.
- Provisions should be made, depending on the intended application of the LCA study, to respect confidential and proprietary matters.
- LCA methodology should be amenable to the inclusion of new scientific findings and improvements in state-of-the art technology.
- Specific requirements are applied to LCA studies which are used to make comparative assertions that are disclosed to the public.
- There is no scientific basis for reducing LCA results to a single overall score or number, since trade-offs and complexities exist for the systems analyzed at different stages of their life cycles.
- There is no single method for conducting LCA studies. Organizations should have flexibility to implement LCA practically as established in this International Standard, based upon the specific application and the requirements of the user (ISO 14040).

1.6 The Phases of a Life Cycle Assessment

LCA studies systematically and adequately address the environmental aspects of product systems, from raw material acquisition to final disposal (from "cradle to grave"). The analysis normally includes the full life cycle of a product from cradle to grave including the life cycle of all pre-products and energy carriers used. Many kinds of environmental interventions, e.g. emissions into water, air and soil as well as resource uses (primary energy carriers, land, etc.) are accounted for. Some authors include also additional effects, e.g. the direct health hazards for employees in the production facilities.

The method distinguishes four main phases, namely (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation (see Figure 1.3). The "Goal and scope definition" describes the underlying questions, the target audience, the system boundaries and the definition of a reference flow for the comparison of different alternatives. The inputs of resources, materials and energy as well as outputs of products and emissions are investigated and recorded in the "Life cycle inventory analysis". Its result is a list of resources consumed and pollutants emitted along the life cycle of a product or system. These elementary flows (emissions and resource consumptions) are described, characterized and aggregated during the "Impact assessment". Conclusions are drawn during the "Interpretation". Normally LCA aims at analyzing and comparing different products, processes or services that fulfill the same utility (e.g. 1kg of synthetic ethanol against 1kg of ethanol from sugar beets) (ISO 14040).

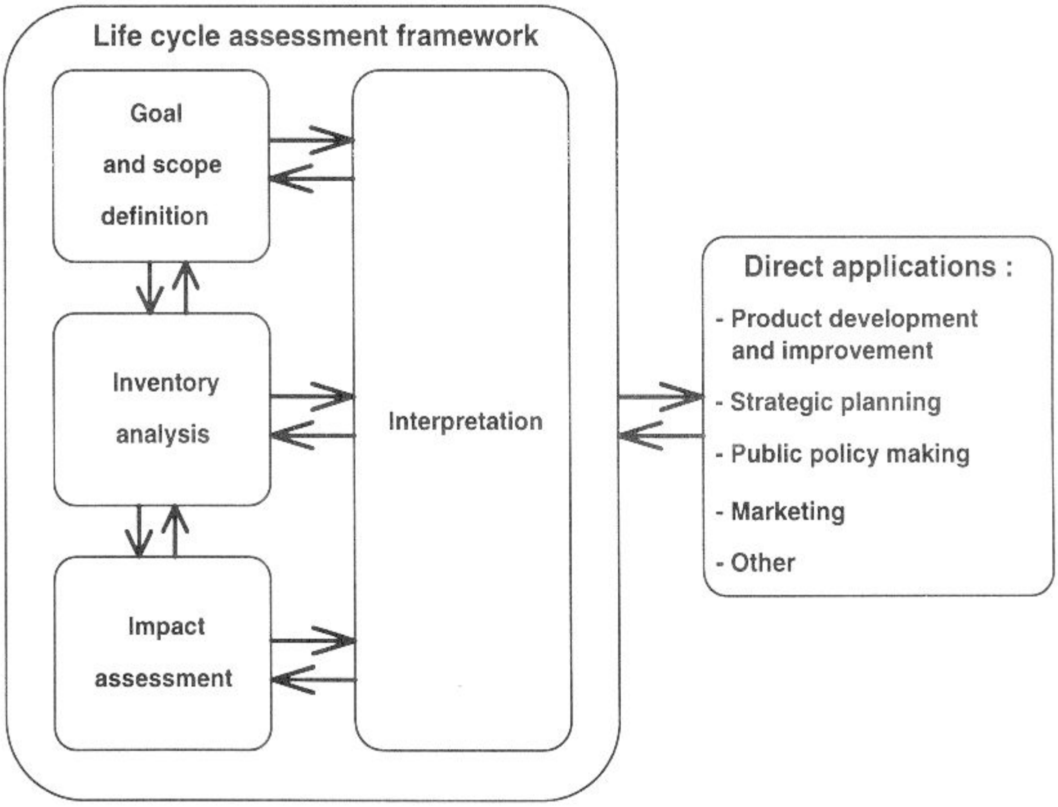

Figure 1.3 Phases of a LCA (International Organization for Standardization ISO 14040 1997- 2000).

CHAPTER TWO
LCA METHODOLOGY

In the United States, the Society of Environmental Toxicology and Chemistry (SETAC) has been actively working to advance the methodology of life cycle assessment through workshops and publications. From their work, a three-component model for Life Cycle Assessment has been developed (SETAC, 1991), and is considered to be the best overarching guide for conducting such analyses. The three components are inventory, impact analysis, and improvement. The inventory stage involves quantifying the energy and material requirements, air and water emissions, and solid waste from all stages in the life of a product or process. The second element, impact assessment, examines the environmental and human health effects associated with the loadings quantified in the inventory stage. The final component is an improvement assessment in which means to reduce the environmental burden of a process are proposed and implemented. It should be emphasized that life cycle assessments are not necessarily performed step-wise and that they are dynamic rather than static. For example, process improvements may become obvious during the inventory assessment phase, and altering the process design will necessitate a reevaluation of the inventory. Additionally, depending on the purpose of the LCA, an impact assessment may not be necessary. Most importantly, a life cycle assessment needs to be evaluated periodically to take into account new data and experiences gained. To date, most work in life cycle assessment has focused on inventory, although efforts to advance impact assessment and improvement are significant. The International Organization for Standardization (ISO) is also involved in life cycle assessment development under the new IS0 14000 environmental management standards. Specifically, the Sub-Technical Advisor Group working on this task has made progress in constructing inventory assessment guidelines, but much disagreement remains on the impact and improvement elements.

The term “life cycle” refers to the major activities in the course of the product’s life-span from its manufacture, use, and maintenance, to its final disposal, including the raw material acquisition required manufacturing the product. Figure 2.1

illustrates the possible life cycle stages that can be considered in an LCA and the typical inputs/outputs measured (SETAC 1991).

Specifically, LCA is a technique to assess the environmental aspects and potential impacts associated with a product, process, or service, by:

- Compiling an inventory of relevant energy and material inputs and environmental releases.
- Evaluating the potential environmental impacts associated with identified inputs and releases.
- Interpreting the results to help decision-makers make a more informed decision.

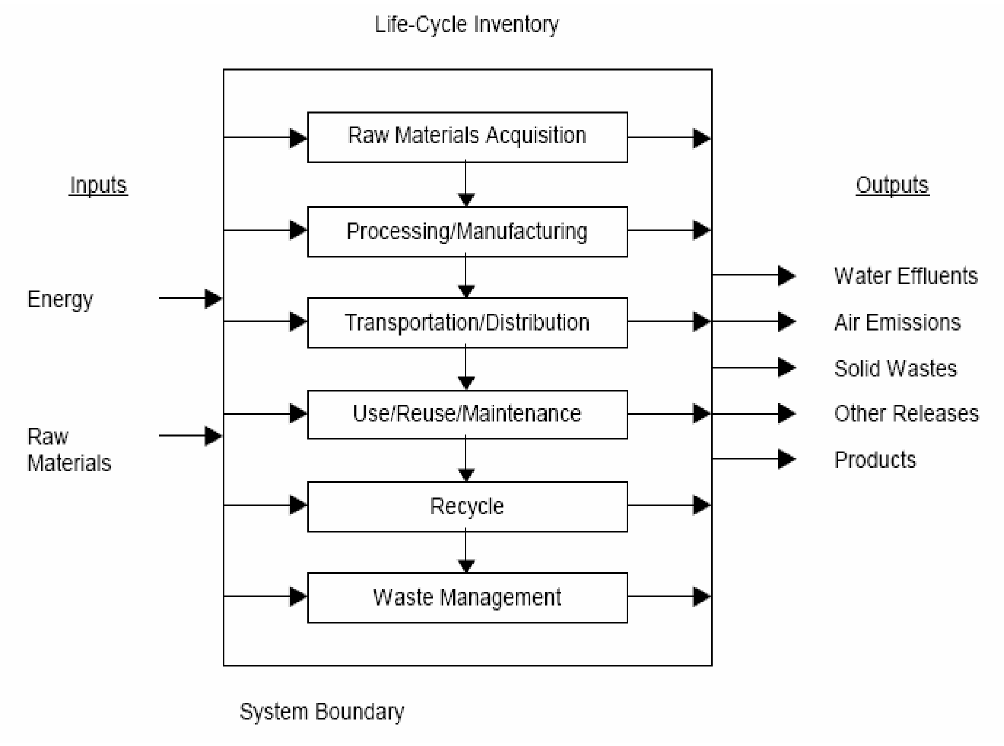

Figure 2.1 The elements of a Life-Cycle Assessment (SETAC 1991).

2.1 Defining Goal and Scope

As with all models of reality, one most understand that a model is a simplification of reality, and as with all simplifications, this means that the reality will be destroyed in some way. The challenge for the LCA practitioner is thus to develop the models in such a way that the simplifications and thus distortions do not influence the result too much.

The best way to deal with this problem is to carefully define a goal and scope of the LCA study before you start. In the goal and scope, the most important choices are described, such as:

- The reason for executing the LCA, and the questions, which need to be answered
- A precise definition of a product, its life cycle and the function and fulfils
- In case products are to be compared, a comparison basis is defined (functional unit)
- A description of the system boundaries
- A description of the way allocation problems will be dealt with
- Data and data quality requirements
- Assumptions and limitations
- The requirements regarding the life cycle impact assessment (LCA) procedure and the subsequent interpretation to be used
- The intended audiences and the way the results will be communicated
- If applicable, the way a peer review will be made
- The type and format of the report required for the study

The goal and scope definition is a guide that helps you to ensure the consistency of the LCA you perform. It is not to be used as a static document. During the LCA, one can make adjustments if at appears that the initial choices are not optimal or practicable. However, such adaptations should be made consciously and carefully (Molender, 2002).

2.2 Functional Unit and Reference Flow

The functions of the investigated system shall be clearly defined. Products or services are defined as a functional output. The functional unit is a measure of the performance of the functional outputs of the product system. The reference flow is a measure of the needed outputs from the product system that are required to fulfill the function expressed by the functional unit (International Organization for Standardization (ISO 1998) (Molender, 2002).

A particularly important in issue in product comparisons is the functional unit or comparison basis. In many cases, one cannot simply compare product A and B, as they may have different performance characteristics. For example, a milk carton can be used only once, while a returnable milk bottle can be used ten or more times. If the purpose of the LCA is to compare milk-packaging systems, one connet compare one milk carton with one bottle. A much better approach is to compare two ways of packaging and delivering 1000 litres of milk. In that case one would compare 1000 milk cartons with about 100 bottles and 900 washings (assuming 900 return trips for each bottle) (Curran 2006).

2.3 System Boundaries

The system boundaries define the unit processes to be included in the product system. The analysis of technical processes required to manufacture products and deliver services is based on environmental process chain analysis. In many cases there will not be sufficient time, data, or resources to conduct a fully comprehensive study (International Organization for Standardization (ISO) 2000b:5.3.3). According to ISO 14041 (International Organization for Standardization (ISO) 2000b) several criteria are used to decide which inputs to be studied, including a) mass, b) energy, and c) environmental relevance. Any decisions to omit life cycle stages, processes or inputs/outputs shall be clearly stated and justified. The criteria used in setting the system boundaries dictate the degree of confidence in ensuring that the results of the study have not been compromised and that the goal of the study will be met.

An important question for agricultural products is the definition of system boundaries between the technosphere system (agricultural production) and nature (e.g. agricultural soil or ground water). Here it has to be clearly defined which part of agricultural soil and groundwater system belongs to the technical system and which to the natural system (Curran 2006).

2.4 Data Quality Requirements

According to ISO 14041 (1998) some descriptions of data quality requirements should be included in the goal and scope definition. These descriptions should cover the following parameters (Curran 2006):

- time-related coverage
- geographical coverage
- technology coverage

Furthermore, for studies that intend to make a comparative assertion that is disclosed to the public, the following additional data quality requirements shall be considered:

- precision: measure of the variability of the data values for each data category expressed
- completeness: percentage of locations reporting primary data from the potential number in existence for each data category unit process
- representativeness: qualitative assessment of the degree to which the data set reflects the true population of interest
- consistency: qualitative assessment of how uniformly the study methodology is applied to the various components of the analysis
- reproducibility qualitative assessment of the extent to which information on the methodology and data values allows an independent practitioner to reproduce the results reported in the study

2.5 Interpretation

Within the interpretation part, a final discussion of the LCI and the LCIA results is made. This should be done according to the defined goal and scope of the study in order to reach consistent conclusions and recommendations. The interpretation phase may involve the iterative process of reviewing and revising the scope of the LCA. It is checked whether the nature and quality of the data collected is consistent with the defined goal. The findings of sensitivity analyses should also be reflected in the interpretation (International Organization for Standardization (ISO 2006).

2.6 Critical Review

A critical review facilitates the understanding and enhances the credibility of LCA studies. This is especially important if comparative assertions raise special concerns. The critical review is done by one or more external experts. The specification of the review process in the ISO documents is rather general. Some basic requirements for the nominations of the experts are listed (such as familiarity of the expert with the ISO 14040 standards as well as his or her technical and scientific expertise and publication of the review report within the LCA report). The critical review process shall ensure that (International Organization for Standardization (ISO 2006) :

- the methods used for the LCA are consistent with the international standard
- the methods are scientifically and technically valid
- the data used are appropriate and reasonable in relation to the goal of the study
- the interpretation reflects the limitations identified and the goal of the study
- the study report is transparent and consistent

CHAPTER THREE
LIFE CYCLE INVENTORY ANALYSES

The basis of any LCA study is the creation of an inventory of the inputs and outputs of most processes that occur during the life cycle of a product. This includes the production phase, distribution, use and final disposal of the product. A product's life cycle can be presented as a process tree.

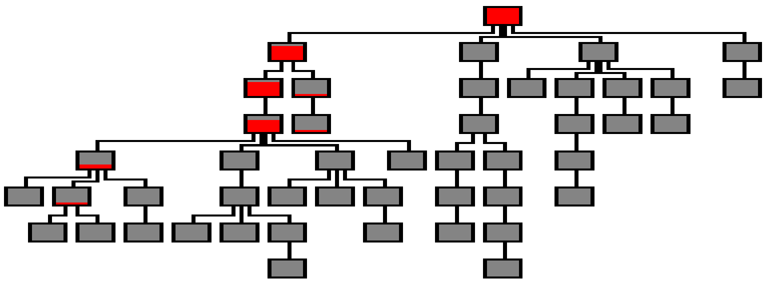

Figure 3.1 Example of a process tree. Each box represents a process which Forms part of the life cycle. Every process has defined inputs and outputs.

Process inputs can be divided into two kinds.

- Inputs of raw materials and energy resources (environmental input).
- Inputs of products, semi-finished products or energy, which are outputs from other processes (economic input).

Similarly, there are two kinds of outputs:

- Outputs of emissions (environmental output).
- Outputs of a product, a semi-finished product or energy (economic output).

With information about each process and a process tree of the life cycle, it is possible to draw up a life cycle inventory of all the environmental inputs and outputs associated with the product. The result is called the table of impacts. Each impact is expressed as a particular quantity of a substance (Curran, 2006).

The table below displays an example of a small part of the table of impacts for the production of two materials. A complete table can have hundreds of rows.

Table 3.1 Some impacts from the production of 1 kg of polyethylene and 1 kg of glass.

	Polyethylene	**Glass**	**Unit**
emission			
CO_2	1.792	0.4904	kg
NO_x	1.091×10^{-3}	1.586×10^{-3}	kg
SO_2	987.0×10^{-6}	2.652×10^{-3}	kg
CO	670.0×10^{-6}	57.00×10^{-6}	kg

It will be clear that such a table does not provide an immediate answer to a question such as whether 1 kg of polyethylene is more or less environmentally friendly than 1 kg of glass. "Impact assessment methods" have been developed which simplify this task of interpretation. Before going into these, there are some problems to be considered regarding the calculation of the table of impacts (Curran, 2006).

The inventory process seems simple enough in principle. In practice, it is subject to a number of practical and methodological problems. They are as follows:

3.1 System Boundaries

In breaking the life cycle down into processes, it is not always clear how far one should go in including processes belonging to the product concerned.

In the production of polyethylene, for example, oil has to be extracted; this oil is transported in a tanker; steel is needed to construct the tanker, and the raw materials needed to produce this steel also have to be extracted. For practical reasons a line must be drawn. For example, the production of capital goods is usually excluded (Curran, 2006).

3.2 Processes that Generate More Than One Product

For example the electrolysis of salt to produce chlorine; the environmental effects of the electrolysis process cannot be ascribed entirely to chlorine alone, as caustic soda and hydrogen are also produced. A suitable allocation rule is needed here, for instance allocation on mass basis or economic value of the products (Curran, 2006).

3.3 Avoided Impacts

When a disposal process generates a profitable output, such as energy generation at a municipal waste incineration plant, it not only causes impacts. It also saves impacts as it is no longer necessary to produce the energy or the material in a normal way.

To allow for this, avoided impacts are introduced. These are equivalent to the impacts that would have occurred in actual production of the material or energy. The avoided impacts of a process are deducted from the impacts caused by other processes. In SimaPro both the attribution of impacts concept and the avoided emissions concept can be used (Curran, 2006).

3.4 Geographical Variations

An electrolysis plant in Sweden uses much less environmentally detrimental electricity than an identical plant in Holland, as hydroelectric power is abundantly used in Sweden (Curran, 2006).

3.5 Data Quality

Publications on environmental process data are often incomplete or inaccurate. Moreover, the data are subject to obsolescence; there are many cases where processing industries have cut emissions by %90 during the last ten years. The use of obsolete data can therefore cause distortions (Curran, 2006).

3.6 Choice of technology

A distinction can be made between *worst, average,* and *best (or modern)* technology. Before starting to collect data it is important to be aware of which type of technology you are interested in. Sima-Pro we have collected average technology as far as possible.

Despite these problems, it is often quite feasible to carry out an impact inventory. It is unreasonable, however, to treat the results as an absolute truth. Factors such as the choice of technology and system boundaries, data quality etc. have to be taken into account when interpreting them. This is why there always seems to be disagreement among experts about the environmental soundness of a product.

Environmental Life Cycle Assessment (LCA) provides a framework for identifying and evaluating environmental burdens associated with the life cycles of materials and services in a "cradle-to-grave" approach. Efforts to develop LCA methodology first began in the US in the 1970s. More recently, the Society for Environmental Toxicology and Chemistry (SETAC) in North America and the US Environmental Protection Agency (USEPA) have sponsored workshops and other projects designed to develop and promote consensus on a framework for conducting life-cycle inventory analysis and impact assessment. Similar efforts have been undertaken by SETAC-Europe, other international organizations (such as the International Standards Organization, ISO), and LCA practitioners worldwide. As a result of these efforts, consensus has been achieved on an overall LCA framework and a well-defined inventory methodology.

LCA systematically identifies and evaluates opportunities for minimizing the overall environmental consequences of resource usage and environmental releases. Early research conducted by the USEPA in LCA methodology along with efforts by SETAC led to the four-part approach to LCA that is widely accepted today (Curran, 2006).

1. Specifically stating the purpose of the study and appropriately identifying the boundaries of the study (Goal and Scope Definition).
2. Quantifying the energy use and raw material inputs and environmental releases associated with each stage of the life cycle (Life Cycle Inventory, LCI).
3. Interpreting the results of the inventory to assess the impacts on human health and the environment (Life Cycle Impact Assessment, LCIA).
4. Evaluating opportunities to reduce energy, material inputs, or environmental impacts along the life cycle (Improvement Analysis, or Interpretation).

CHAPTER FOUR
LIFE CYCLE IMPACT ASSESSMENT

The Life Cycle Impact Assessment (LCIA) phase of an LCA is the evaluation of potential human health and environmental impacts of the environmental resources and releases identified during the life cycle inventory (LCI). Impact assessment should address ecological and human health effects; it can also address resource depletion. A life cycle impact assessment attempts to establish a linkage between the product or process and its potential environmental impacts.

The inventory table is the most objective result of a LCA study. However, a list of substances is difficult to interpret. To make this task easier, life cycle impact assessment (LCIA) is used for evaluation of the impacts.

- Classification and characterization
- Normalization
- Evaluation or weighting

Two problems exist in impact assessment:

1. There are not sufficient data to calculate the damage to ecosystems by an impact.
2. There is no generally accepted way of assessing the value of the damage to ecosystems if this damage can be calculated.

One of the oldest impact assessment methods is the EPS (Environmental Priority Strategy) system as developed by the IVL in Sweden. In this method, the complete chain of cause and effect from each impact on a human equivalent is calculated.

Another method is the Ecopoints method, developed for the Swiss government. It is based on the distance-to-target principle. The distance between the current level of an impact and the target level is assumed to be representative of the seriousness of the emission (Curran 2005).

Classification

In the classification step, all substances are sorted into classes according to the effect they have on the environment. For example, substances that contribute to the greenhouse effect or that contribute to ozone layer depletion are divided into two classes. Certain substances are included in more than one class. For example, NOx is found to be toxic, acidifying and causing eutrophication (Curran 2005).

Characterization

The substances are aggregated within each class to produce an effect score. It is not sufficient just to add up the quantities of substances involved without applying weightings. Some substances may have a more intense effect than others. This problem is dealt with by applying weighting factors to the different substances. This step is referred to as the characterization step (Curran 2005).

Normalization

In order to gain a better understanding of the relative size of an effect, a normalization step is required. Each effect calculated for the life cycle of a product is benchmarked against the known total effect for this class. For example, the Eco-indicator method normalizes with effects caused by the average European during a year. Of course it is possible to choose another basis for normalization (Curran 2005).

Evaluation or weighting

Normalization considerably improves our insight into the results. However, no final judgment can be made as not all effects are considered to be of equal importance. In the evaluation phase the normalized effect scores are multiplied by a weighting factor representing the relative importance of the effect (Curran 2005).

4.1 Evaluation of Environmental Impact

It is very hard to quantify the environmental impacts while making them comparable among different processes. As my mentor said “how do you quantify the loss of a beautiful mountain scene destroyed by our actions?” These kinds of nontangibles are very hard to quantify even under the best of circumstances. However, being able to quantify these issues and others is very important because they are encountered in every study using LCA because we have to attempt to quantify the total environmental impact of a process. At this stage of an LCA, these issues are mostly under-developed because we do not have a good model to calculate the impact indices for some measures. It is especially true that overall toxicity effects of individual chemicals or mixtures of chemicals are not well understood. Much modeling work in these areas has to be done before a more accurate LCA study can be done.

The full LCA assessment also helps to focus work on any subsequent qualification of the data of the inventory. It shows which of the interchanges have the largest potential impacts and it should thus be performed with the highest possible degree of precision and certainty. The impact assessment thus qualifies the inventory as a basis for decisions in comparisons between products, and it can also be used to focus the further collection of data to areas where uncertainties exist.

The LCA method's impact assessment phase is subject to the same general requirements with respect to transparency, reproducibility and scientific foundations as the other phases in the LCA. However, it is in the nature of the concept that an assessment can never be totally objective because of the issues described earlier in this chapter about the inability to have an absolute measure for some environmental effects (Zhu, 2004).

The impact assessment progresses through three steps:

Calculation of potential contributions to various categories of impact.

In the first step of an LCA impact assessment, the types of environmental impacts which attribute to the interchange are assessed. For each individual emission to the environment, a calculation is then made of the magnitude of the contributions to various impact categories. The main categories are: Abiotic depletion (ADP); Energy depletion (EDP); Global warming (GWP); Human toxicity (HT); Ecotoxicity (ECA/ECT); Acidification (AP); Nitrification (NP); Ozone depletion (ODP); Photochemical oxidant creation potential (POCP) (Zhu, 2004).

These contributions are called the emission's environmental impact potentials.

Comparison of impact potentials and resource consumptions with a common reference to show which are large and which small.

In many cases, the process or material alternative that will have the least impact cannot be determined on the basis of the summarized resource consumption or the potential impacts on the working environment or the calculated environmental impact potentials.

One alternative will often have the least impact in some areas, while another alternative will have the least impact in others. In such situations, it is important to be able to assess which of the potential impacts and resource consumptions are large and which are small and to weight them in such a way that an aggregate environmental impact can be determined. This assessment can be difficult to perform on the basis of the figures alone. For an example, we use CO2 as an equivalency to evaluate different chemicals. More examples will be shown when we go through the major categories in LCA (Zhu, 2004).

Weighting of the normalized impact potentials and resource consumptions to determine which impacts are most significant overall.

Before the normalized impact potentials or resource consumptions can be made to be compared directly, account must be taken of the seriousness of each individual impact in relation to the others. Scientific, political and normative considerations are involved in this step, expressed in a weighting factor for each of the impact categories and resource consumptions.

The normalization and weighting elements are interdependent of the LCA method and are therefore presented together in the total result. For the environmental impact categories, the quantitative assessment of "ecotoxicity" and "human toxicity" involves a considerable amount of work. A qualitative assessment method for these impact categories has therefore also been developed as a part of the LCA method by others, based on the information presented in the chemicals' hazard classification and labeling (Zhu, 2004).

4.2 Environmental Impact Categories

Description of most common environmental impact categories:

Classification and characterization are a calculation process in which each impact parameter of the inventory is converted to a contribution to environmental impact. Generally, the following environmental themes are considered.

Abiotic depletion potential (ADP) - abiotic depletion concerns the extraction of nonrenewable raw materials such as ores.

Energy depletion potential (EDP) - energy depletion concerns the extraction of nonrenewable energy carriers.

Global warming potential (GWP) - an increasing amount of CO_2 in the earth's atmosphere leads to more absorption of radioactive energy, and consequently, to an increase in temperatures on Earth. This is referred to as global warming. CO_2, N_2O, CH_4, and aerosols all contribute to global warming.

Human toxicity (HT) - exposure of humans to toxic substances causes health problems. Exposure can take place through air, water, or soil, especially via the food chain.

Ecotoxicity (ECA/ECT) - exposure of flora and fauna to toxic substances cause health problems in them. Ecotoxicity is defined for water (aquatic ecotoxicity) and soil (terrestrial ecotoxicity).

Acidification potential (AP) - acid deposition onto soil and into water may lead, depending on the local situation, to changes in the degree of acidity. This affects flora and fauna mostly in negative ways.

Nitrification potential (NP) - addition of nutrients to water or soil will increase the production of biomass. This in turn leads to reduction in the oxygen concentration, which affects higher organisms like fish, can lead to undesirable shifts in the number of species in an ecosystem, and thus to a threat to biodiversity. Main elements in this section are nitrogen containing substances, phosphates, and organic materials.

Ozone depletion (ODP) - depletion of the ozone layer leads to an increase in the amount of UV light reaching the earth's surface. This may lead to human diseases and may influence ecosystems in a negative way.

Photochemical oxidant creation potential (POCP) - Reaction of NOx with volatile organic substances leads, under influence of UV light, to photochemical oxidant creation, which causes smog.

Each category of environmental impact is calculated separately before being weighted and aggregated. However, criteria are established to determine whether a substance contributes to a certain environmental impact category before it will be included. An equivalency factor expresses the potential environmental impact for a substance as the quantity of a reference substance which will make the same contribution to the environmental impact as one gram of the substance. For each environmental impact category, the reference substance is chosen as a typical or important contributor. For instance, CO_2 is used in the case of green house gases because it is the biggest contributor of the green house effect.

For non-global impact categories, it can be relevant to consider use of site factors in the calculation of potential impacts. Inclusion of site factors will qualify the calculated impact potentials for that local region. The use of site factors is, however, not yet generally implemented in the LCA method (Zhu, 2004).

4.3 Quantifying Environmental Impact

There are thousands of different substances which can contribute to the impacts considered under "ecotoxicity" and "human toxicity". Equivalency factors have, however, only been calculated for a few hundred substances due to the difficulty of data collection, especially for human impacts. Many users of the LCA method will not possess the necessary chemical or eco-toxicological backgrounds in order to calculate equivalency factors for these two categories themselves. The work must then be given to an external consultant with the requisite expertise before the full LCA incomplete. Another approach is to do approximations and estimations of those approximations on the overall results so that uncertainties in data can be handled (Zhu, 2004).

4.3.1 Global Warming Potential

For a substance to be regarded as contributing to global warming, it must be a gas at normal atmospheric temperatures and:

- be able to absorb infrared radiation and be stable in the atmosphere with a residence time of years to centuries, or

- be of fossil origin and converted to CO_2 on degradation in the atmosphere

The Intergovernmental Panel on Climate Change (IPCC) has developed an equivalency factor system which can weight the various substances according to their efficiencies as greenhouse gases. GWP, global warming potential, is a characterization factor that defines to characterization of potential contribution from a given substance which is in use for a time horizon of 100 years (standard). CO_2 is used as a reference material, so all the emissions which are characterized by this method are expressed as equivalent emissions of the CO_2. The LCA method's criteria for which substances contribute to global warming generally follow the IPCC's recommendation of excluding indirect contributions to the greenhouse effect. The indirect effects are difficult to model, and the IPCC is therefore refraining, for the time being, from quantifying indirect contributions with the exception of contributions from methane gas (Zhu, 2004).

Calculate the Global Warming Potential

One can calculate the global warming potential by multiplying the magnitude of the emissions with the equivalency factor (Zhu, 2004):

$$EP(gw) = Q \bullet GWPi\ (gw). \qquad (4.1)$$

$$GWP_i = \frac{\text{contribution to GWP global warming from gas } \boldsymbol{i} \text{ over T years}}{\text{contribution to global warming from } \boldsymbol{CO} \text{ over T years}}$$

$$= \frac{\int_0^T a_2(t) * c_i(t)dt}{\int_0^T a_{CO2}(t) * c_{CO2}(t)dt} \quad (4.2)$$

ai (W/m^2p_{mol}) is the gas's specific IR absorption coefficient, its instantaneous radioactive forcing, assuming that the composition of the atmosphere remains the same.

$c_i(t)$ (p_{mol}) is the time-dependent residual concentration of gas 'i' deriving from the pulse-emission in question in 1986,

a_{CO2} (t) c_{CO2} (t) and are the magnitudes of the corresponding emission of CO_2. Examples of results are shown in Table 4.1 below.

Table 4.1 Global warming potentials (GWP) and atmospheric lifetimes (years) used in the inventory source: IPCC (1996).

Gas Atmospheric	**Lifetime**	**100-year GWP 20-year**	**GWP**	**500-year**
Carbon Dioxide	50-200	1	1	1
Methane (CH_4)	123	21	56	6.5
Nitrous Oxide (N_2O)	120	310	280	170
HFC-23	264	11,700	9,100	9,800
HFC-125	32.6	2,800	4,600	920
CF4	50,000	6,500	4,400	10,000
C2F6	10,000	9,200	6,200	14,000
C4F10	2,600	7,000	4,800	10,100
C6F14	3,200	7,400	5,000	10,700

Indirect contributions to GWP are hard to calculate because the IPCC does not include indirect contributions from gases other than methane. The LCA method, nevertheless, offers the option of including that part of the indirect contribution from volatile organic compounds (VOCs) and carbon monoxide (CO) attributable to their predictable conversion to CO_2. This applies only if the gases originate from fossil resources.

As is clear from Table 4.1, the choice of the time scale, t, plays a large role in the magnitude of the equivalency factor. For those gases with atmospheric lives significantly shorter than that of the reference gas CO_2, the equivalency factor decreases with an increase in t. The opposite is the case for those gases with significantly longer lives than CO_2. In accordance with general LCA practice, one uses a time scale of 100 years, but equivalency factors for 20 and 500 years are also given in the table in order to show the significance of t and to provide an option of alternative choices on this method.

GWP values allow policy makers to compare the impacts of emissions from different gases. According to the IPCC, GWPs typically have an uncertainty of roughly 35 percent, though some GWPs have larger uncertainties than others, especially those in which lifetimes have not yet been ascertained. In the following work, we have chosen to use the 100 year time horizon which is consistent GWPs from the IPCC Second Assessment Report (SAR) (Zhu, 2004).

4.3.2 Stratospheric Ozone Depletion

Stratospheric ozone is broken down as a consequence of man-made emissions of halocarbons, i.e. CFCs, HCFCs, halons and other long-lived gases containing chlorine and bromine. This can have dangerous consequences in the form of increased frequency of skin cancer in humans and damage to the plants which form the basis of all ecosystems. The stratospheric depletion of ozone is an impact which affects the environment on a global scale (Zhu, 2004).

Determine Which Substances Contribute to Ozone Depletion

For a substance to be considered as contributing to stratospheric ozone depletion, it must.

- be a gas at normal atmospheric temperatures
- contain chlorine or bromine
- be stable with a lifetime in the atmosphere of several years to centuries, to allow for its transportation up into the stratosphere

The most important groups of ozone-depleting halocarbons are the CFCs, the HCFCs, the halons and methyl bromide. In contrast to these, the HFCs are a group of halocarbons which contain neither chlorine nor bromine but only fluorine, and which are therefore not regarded as contributors to stratospheric depletion of ozone.

CFCs and HCFCs are used mainly as foaming agents in foam plastic, as refrigerants, and as solvents. Halons are used as fire-extinguishing agents in firefighting equipment. Methyl bromide is used in the disinfection of buildings and of soil in market gardens.

The production of halocarbons is regulated under the Montreal Protocol by the United Nation. Consumption of methyl bromid ewas frozen in 1995, and consumption of HCFCs is to be decreased gradually. The deadlines for phasing out have been brought forward in a number of countries. CFCs and halons can, however, continue to be produced in Third World countries until 2010 (UNEP, 1993), and they will therefore also occur in future inventories of product systems.

Calculate the Ozone Depletion Potential

First, one chooses the time scale for which the ozone depletion potential is to be calculated. Unless specific reasons indicate otherwise, one selects a infinite time scale. After finding the substance's equivalency factor for the chosen time scale, calculate the ozone depletion potential by multiplying the magnitude of the emission by the equivalency factor (Zhu, 2004):

$$EP(od) = Q \bullet EF(od) \tag{4.3}$$

Together with UNEP (United Nations Environment Programme) and a number of other organizations, the World Meteorological Organization (WMO) organizes the "Global Ozone Research and Monitoring Project", a research network of experts in atmospheric chemistry. The network reviews international developments in scientific knowledge of stratospheric ozone depletion and every few years issues status reports summarizing the latest findings The status reports present the Ozone Depletion Potentials (ODPs), which for individual gases express the ozone depletion potential as an equivalent emission of a reference substance CFC11 (CFC13). These ODP values are used as equivalency factors in the calculation of the ozone depletion potential. The equivalency factor is thus defined as:

$$ODP_i = \frac{\textit{contribution to } \text{stratospheric ozone depletion from gas i}}{\textit{contribution to stratospheric ozone depletion from CFC}11} \tag{4.4}$$

For the most short-lived of the gases, especially the HCFCs, this will result in some markedly larger equivalency factors.

In accordance with general LCA practice, however, recommend use, for equivalency factors of ODP values representing the gases' full contributions, but the in most references also gives equivalency factors for 5, 20 and 100 years for some of the gases

4.3.3 Photochemical Ozone Formation

When solvents and other volatile organic compounds are released to the atmosphere, they are often degraded within a few days. The reaction involved is an oxidation, which occurs under the influence of light from the sun. In the presence of oxides of nitrogen (NO_X), ozone can be formed. The oxides of nitrogen are not consumed during ozone formation, but have a catalyst-like function. This process is termed photochemical ozone formation.

The volatile organic compounds are broken down especially in the troposphere, the lowest region of the atmosphere, to which they are emitted. The most significant man-made sources of VOCs are road transport with its emission of unburned gasoline and diesel fuel and the use of organic solvents, e.g. in paints.

Ozone attacks organic compounds in plants and animals or materials exposed to air. This leads to an increased frequency of problems of the respiratory tract in humans during periods of photochemical smog in cities. For agriculture, it causes a reduction in yield which for Denmark is conservatively estimated to be about %10 of total production (Zhu, 2004).

Determine which substances contribute to photochemical ozone formation

Photochemical ozone formation is an impact which affects the environment on both local and regional scales. The substance is consider as a contributor to photochemical ozone formation if Check whether the substance is an organic compound with a boiling point below 250°C and

- contains hydrogen or
- contains double bonds between carbon atoms

Consider carbon monoxide CO as a further contributor to photochemical ozone formation. The presence of oxides of nitrogen can be equally important a man-made factor in photochemical ozone formation as emission of VOCs. Despite this, a contribution from oxides of nitrogen to photochemical ozone formation cannot be calculated because the equivalency factor system used for calculation of ozone formation potentials does not facilitate calculation of an equivalency factor for NO_X. The significance of NO_X for ozone formation is, however, reflected in the fact that two sets of equivalency factors are used: one for emissions of VOCs occurring in areas with a low background concentration of NOX and one for emissions occurring in areas with a high background concentration of NO_X (Zhu, 2004).

Calculate the photochemical ozone formation potential

Calculate the photochemical ozone formation potential by multiplying the magnitude of the emission by the equivalency factor found:

$$EP(po) = Q \bullet EF(po) \tag{4.5}$$

In the same way as the GWP values for global warming and the ODP values for stratospheric ozone depletion, the POCP values express the ozone formation potential as an equivalent emission of a chosen reference substance. For photochemical ozone formation the reference substance is the gas ethylene (C_2H_4).

$$POCP_i = \frac{\textit{POCP contribution to ozone formation from gas i}}{\textit{contribution to ozone formation from } C_2H_4} \tag{4.6}$$

There is no international panel of experts for the environmental impact of photochemical ozone formation such as there are for other global environmental impacts (Zhu, 2004).

The POCP values are calculated with the aid of atmospheric chemical models and a series of assumptions must be made on climatic conditions and the magnitude of the simultaneous emissions of a number of other VOCs and of NO_X. The assumptions underlying the POCP values in appendix correspond to typical situations in areas with low and high background concentrations of NO_X. The assumptions are discussed in the references presenting these POCP values.

A time scale must also be chosen for ozone formation in model calculations of POCP values. A POCP value for a short time scale of 24 hours describes the photochemical ozone formation immediately surrounding the place of emission corresponding to potential contribution to episodes of photochemical smog. For a longer time scale, such as a week, most of the VOCs will be broken down and the POCP value provides a better expression of the total ozone formation potential.

POCP values have been calculated only for the individual VOCs of greatest significance for total photochemical ozone formation. But these are not necessarily the compounds of greatest significance for the ozone formation potential in an LCA. For example, styrene will give a significant contribution to the ozone formation potential in the LCA of the polymer polystyrene, but none of the references gives a POCP value for styrene. It can therefore be an advantage to be able to make an estimation of missing POCP values. Hauschild & Wenzel describe various methods of estimating POCP values.

Emissions of VOCs will often figure in the inventory for a product system, without an indication of which individual compounds they are composed of. The composition can vary greatly for different sources, but if the source of the VOC emission is known (e.g. "exhaust from gas-engine cars"), it may be possible in to find an average POCP value which is representative for the VOC mixture. The average POCP values are calculated as a weighted average for the VOC mixture which is characteristic of the type of source (Zhu, 2004).

4.3.4 Acidification

When acids and compounds that are convertible to acids are emitted to the atmosphere and deposited in water and soil, this may eventually result in a decrease in pH, which causes an increase in acidity. This has consequences in the form of a widespread decline of coniferous forests in many places in Europe and the USA and increased fish mortality in mountain lakes in Scandinavia and central Europe. Corrosion damage to metals and disintegration of surface coatings and mineral building materials are also caused by acidification on exposure to wind and weather.

Combustion processes in electricity and heating production are the most significant man-made sources of acidification. The contribution to acidification is greatest when the fuels contain sulphur. Acidification is an impact which mainly affects the environment on a regional scale (Zhu, 2004).

Determine which substances contribute to acidification

1. For a substance to be considered a contributor to acidification, it must cause introduction of or release of hydrogen ions in the environment, and

2. The anions which accompany the hydrogen ions must be leached or washed out from the system.

The addition of hydrogen ions occurs either when the substance itself is an acid or is converted to an acid, or when hydrogen ions are released as the substance is converted in the environment.

The number of substances which should be considered contributors to acidification is not large, and in practice the list of equivalency factors is calculated which can be used to decide whether a substance contributes to acidification. Note that emission of organic acids is not regarded as a contribution to acidification (Zhu, 2004).

Calculate the acidification potential

1. Find the substance's equivalency factor
2. Calculate the acidification potential by multiplying magnitude of the emission by the equivalency factor found:

$$EP\ (ac) = Q \bullet EF\ (ac) \tag{4.7}$$

There is no internationally accepted system of equivalency factors for acidifying substances. In contrast to the global environmental impacts and photochemical ozone formation, it has therefore been necessary to develop equivalency factors for acidification.

Calculation of the equivalency factor for a substance is based on the number of hydrogen ions which can theoretically be released from the substance directly or after any conversions in the environment.

Whether or not the acidification potential is realized depends on the accompanying anion release from the ecosystem which receives the emission. For some substances, the proportion of anions released can vary from ecosystem to ecosystem.

As for the potentials for the other types of environmental impacts, the acidification potential is expressed as an equivalent quantity of a reference substance. Sulphur dioxide (SO_2) is used as the reference substance for acidification. Should the inventory include compounds causing acidification, it is easy to calculate equivalency factors for them (Zhu, 2004).

4.3.5 Ecotoxicity

Chemicals emitted as a consequence of human activities contribute to "ecotoxicity" if they affect the function and structure of the ecosystems by exerting toxic effects on the organisms. If the concentrations of environmentally hazardous substances caused by the emission are high enough, the toxic effects can occur as soon as the substances are released. This form of toxic effect is called acute ecotoxicity**.** It often results in the death of organisms exposed.

Toxic effects which are not acutely lethal and which first appear after repeated or long-term exposure to the substance are called "chronic ecotoxicity". Chronic ecotoxicity is often caused by substances which have a low degradability in the environment and which can therefore remain for a long time after their emission (persistent substances). Some substances also have a tendency to accumulate in living organisms so that tissues and organs can be exposed to concentrations of the substance which are far higher than the concentrations in the surrounding environment. The chronic ecotoxicity of a compound is thus determined by its toxicity, its biodegradability and its ability to accumulate in living organisms. The result of a chronic ecotoxic impact can, for example, be reduced reproductive capacity, which means that the species' chances of survival in the long term are reduced.

Ecotoxicity is an impact which predominantly affects the environment on local and regional scales. It can be a global impact for some toxic substances of very low biodegradability with a strong tendency to accumulate in living organisms.

The calculation of the equivalency factors is based on considerations of the substance's potential fate in the environmental and effects on the ecosystems exposed to it. Potential contributions to ecotoxicity are considered for the following emissions in a product system, in water, in soil, in wasted water treatment plants.

$$EPi=EFi\times Qi \tag{4.8}$$

EFi is the equivalency factor for ecotoxicity from substance i in compartment C

Total understanding of ecotoxicity has not been reached yet at the present stage. The quantifying methods that are used in research are normally based on empirical methods. This is an area that needs much improvement, which can lead us to less uncertainty involved with any LCA study.

4.3.6 Human Toxicity

In contrast to the other impact categories, human toxicity includes many different impact mechanisms, such as damage to DNA, induction of allergy or inhibition of specific enzymes.

It is still in a development stage; because the mechanism of action is not known for many substances. The different mechanisms which underlie toxicity are therefore treated here as if there were one primary impact mechanism. The list of substances from the product system classified as contributing to human toxicity is more comprehensive and less uniform than the corresponding list for the other environmental impact categories.

Ways to become exposed to impacts of pollutants in the environmental include: inhalation, ingestion of polluted groundwater, surface water or soil. Indirect exposure also occurs via: ingestion of primary producers which are exposed to pollution, ingestion of consumers or their products (Zhu, 2004).

Key properties to evaluate:

1. Toxicity (empirically determined)
2. Persistence (empirically determined in biodegradability tests)
3. Bioaccumulation potential (empirically determined or estimated)

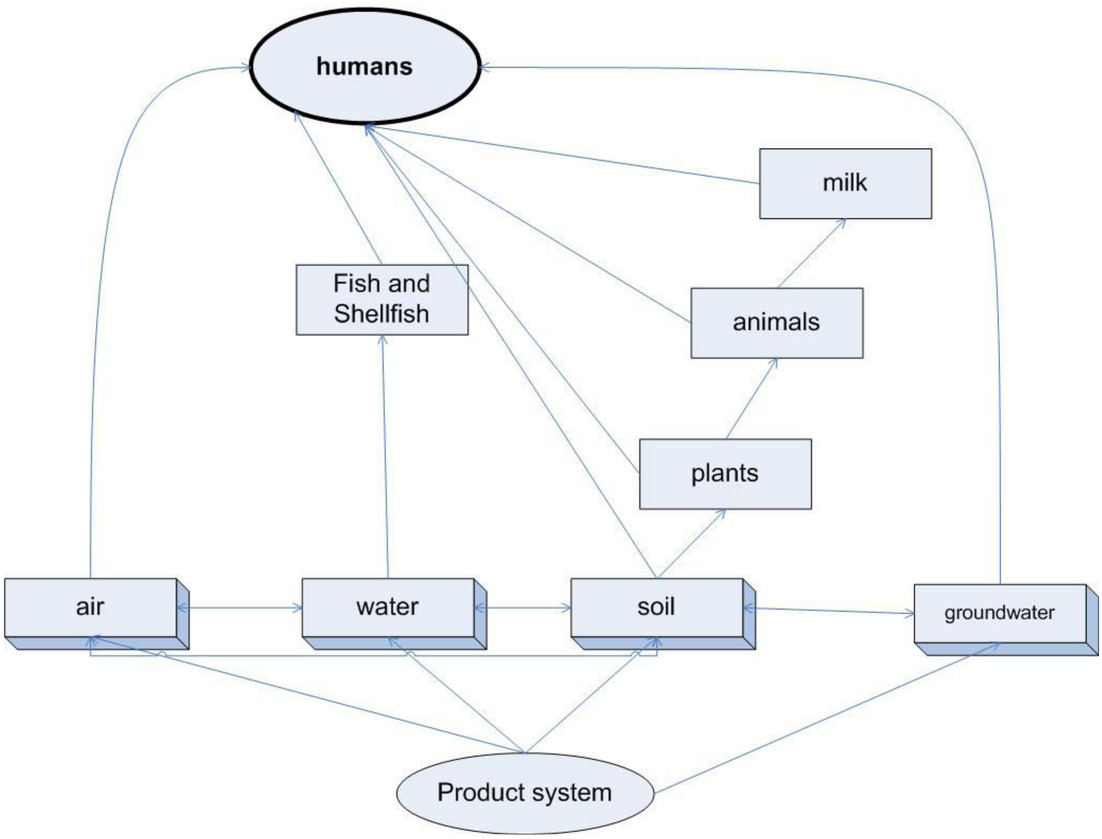

Figure 4.1 The exposure routes under consideration for humans in the environment. 7 exposure routes are outlined.

To calculate the equivalency factor for human toxicity form substance i in compartment C:

EP htc EF htc Qi = × (4.9)

EF(htc)i is the equivalency factor for toxicity from substance i in compartment C.

The expression for the equivalency factor is constructed as shown in Figure 4.2.

Transfer

Biodegradability

Intake

$$EF(htc)_i = \boxed{I_c \times T_{c,i} \times BIO_i}$$

Effect

Figure 4.2 Composition of human toxicity: intake, biodegradability, transfer.

4.3.7 Resource Depletion

The procedure of assessing resource consumption enables a ranking of consumptions of the various types of resources which occur in a product system. SETAC has a working group to develop the method.

In order to make an inventory of consumptions of various types of resources, it is necessary to follow the consumption of all materials and fuels back to extraction of primary resources form nature, including the earth crust, the sea, the forest, etc. All consumptions of primary resources in the product system must in principle be inventoried. This principle is inherent from the LCA itself.

The first step in the method is therefore to express all intakes in the product system as primary intakes, which is as pure resources (Zhu, 2004).

4.3.8 Working Environment

A range of relevant exposures was selected for each of the selected impact categories:

1. Chemical: exposure to carcinogens, exposure to neurotoxins, and exposure to allergens
2. Noise: exposure to noise which causes hearing impairment
3. Work accidents: mainly includes bodily damage
4. Odor: the odor of chemicals sometimes causing harsh working environment sometimes causing health problems for workers
5. Monotonous work: effects on the musculoskeletal apparatus

An exposure threshold is defined for each of the listed major category: for chemical exposure is %10 of the working environment limit value or skin contact. For noise is 80 dB (A), and for monotonous repetitive work it is repetition of the same movement more than twice per minute. The odor threshold value of a substance is defined as the concentration of that substance under defined standard conditions at which 50% of a representative sample of the population can just detect the difference between a sample of air mixed with that substance and a sample of clean air.

The environmental impacts are classified as global, regional and local. Impacts on working environment are highly local (Zhu, 2004).

4.4 Implementation of Life Cycle Impact Assessment Methods

The ecoinvent database offers life cycle inventory (LCI) and life cycle impact assessment (LCIA) results. LCIA methods do normally assign a factor to single elementary flows in an inventory table. There are different types of factors, which are shortly described in Table 4.2 (Forster, 1998; Jungbluth & Frischknecht, 2000).

Table 4.2 Type of factors provided by LCIA methods.

Factor name	**Description**
Characterisation factor	The importance of single flows in relation to a specific basic flow is characterised with a factor, e.g. global warming potential of greenhouse gases in relation to CO_2.
Normalized factor	Another factor, e.g. a characterisation factor, is normalized by division through the total sum of the characterised flows in a certain area and within a certain time.
Weighted factor	A weighting is applied to the characterised or normalised results from different categories in order to calculate a final score.
Damage factor	The possible damage due to an emission is described with a factor. This can include a modeling for the environmental fate, a characterisation of the substances and a final

There are a range of methodological problems while linking the LCIA methods with the elementary flows of a database. Major problems are;

- substance names of elementary flows in the LCIA method and in the database do not match
- elementary flows in the database are not considered by the method
- factors in the method do not have a corresponding flow in the database

- modeling in LCIA and in the database overlaps or does not match

In the past the methodological problems have lead to different results even if the same LCIA method has been applied to the same inventory results. Therefore implementation reports for the assignment of LCIA methods to inventory results have also been published earlier (Forster, 1998; Jungbluth & Frischknecht, 2000).

4.4.1 Description of the Different Methods

4.4.1.1 CML 2001

In 2001, a group of scientists under the lead of CML (Center of Environmental Science of Leiden University) published a new "operational guide to the ISO standards". In this guide the authors propose a set of impact categories and characterisation methods for the impact assessment step. A "problem oriented approach" and a "damage approach" are differentiated. Since the damage approaches chosen are the Eco-indicator 99 and the EPS method, the impact assessment method implemented in ecoinvent as CML methodology is the set of impact categories defined for the problem oriented approach.

In order to use this method, it is necessary to link the elementary flows of ecoinvent data to the substance names given in the publication of the characterisation factors. This background paper describes the implementation of the problem oriented approach according to CML 2001 with its difficulties in the assignment and some assumptions that had to be made (Ginee et al., 2001).

4.4.1.2 Cumulative Energy Demand

Cumulative Energy Requirements Analysis (CERA) aims to investigate the energy use throughout the life cycle of a good or a service. This includes the direct uses as well as the indirect or grey consumption of energy due to the use of, e.g.

construction materials or raw materials. This method has been developed in the early seventies after the first oil price crisis and has a long tradition.

Cumulative energy analysis can be a good 'entry point' into life cycle thinking. But it does not replace an assessment with the help of comprehensive impact assessment methods such as Eco-indicator 99 or ecological scarcity. If more detailed information on the actual environmental burdens and especially on process-specific emissions are available - and the ecoinvent database provides such information - more reliable results are available with such methods. Thus (Kasser & Poll, 1999) e.g. write that the CED (Cumulative Energy Demand) “makes only sense in combination with other methods" (Boesch et al., 2007).

4.4.1.3 Eco-Indicator 99

In 1997 a group of scientists introduced a new method for life cycle impact assessment – The Eco-indicator 99. The final report was published in 1999 and a first revised issue has been made available via the Internet in 2000 (Goedkoop & Spriensma, 2000 a; b).

In order to implement this method in the ecoinvent LCI (life cycle inventory) database it is necessary to assign the damage factors to the elementary flows of resources (Nemecek et al., 2004).

Eco-indicator 99 valuation factors are calculated in three steps:

- Damage factors for the pollutants or resource uses are calculated for different impact categories.
- Normalisation of the damage factors on the level of damage categories.
- Weighting for the three damage categories and calculation of weighted Eco-indicator 99 damage factors.

The Eco-indicator 99 damage, normalisation and weighting factors have been implemented in an EXCEL worksheet (/ecoinventTools/3_EI'99.xls). All inputs are linked together in the table according to the Eco-indicator 99 method. Thus a change of the normalisation factor leads for example to an automatic recalculation of all results for Eco-indicator 99 factors. The calculation for the work sheet consists of the following tables:

- Intro
- EI'99 damage factors
- Normalization & Weights
- X-Impact Factor (with the weighted damage factors implemented in the database)

Repeated formulas have been removed from two worksheets of the EXCEL-table in order to minimize the file size for downloading. After opening the EXCEL-worksheet it is necessary to change the worksheets "X-Impact Factor" (Nemecek et al., 2004).

4.4.1.4 Ecological Footprint

The ecological footprint is defined as the biologically productive land and water a population requires to produce the resources it consumes and to absorb part of the waste generated by fossil and nuclear fuel consumption. In the context of LCA, the ecological footprint of a product is defined as the sum of time integrated direct land occupation and indirect land occupation, related to nuclear energy use and to CO_2 emissions from fossil energy use, clinker production (e.g. CO_2 emitted when burning the limestone for cement production) (Doka, 2007):

$$EF = EF_{direct} + EF_{CO2} + EF_{nuclear} \quad (4.10)$$

4.4.1.5 EDIP'97 – Environmental Design of Industrial Products (Version 1997)

The EDIP'97 method (EDIP is the abbreviation of "Environmental Design of Industrial Products") is the result of a four year effort in the Mid-1990s in Denmark, including the Technical University of Denmark, several Danish industry companies as well as the Danish Environmental Protection Agency. The final report of the project was published in 1997 (Wenzel et al., 1997); a report with more detailed scientific information concerning the different impact factors one year later (Hauschild & Wenzel, 1998). The implementation is based on an updated version of the characterisation factors, available on the website of the Danish LCA center (DK LCA Center, 2007)

In order to use this method together with the data from a database like ecoinvent, the equivalency factors from the EDIP'97 literature have to be linked to the respective elementary flows within ecoinvent (Dones, 2004):

According to Wenzel et al., 1997, the EDIP'97 method translates the cumulated inventory data of an examined system "into potential contributions to various impacts within the main groups' environment, resources and working environment". Due to the already mentioned lack of one part of the required information, only two of these groups – environment and resources are actually covered by the implementation. In order to have a maximum of transparency and reproducibility, the whole method distinguishes between three different steps:

1. **Environmental impact potentials.** Similar to most other methods (e.g. CML, Eco-indicator'99 ...), the contribution of each individual emission to the various impact categories is calculated by using the respective equivalency factors.

2. **Normalization with a common reference.** In order to see which of the various impact potentials, resource consumptions are relevant compared with a common reference (e.g. total European values).

3. **Weighting of the normalized impact potentials. According to Wenzel et al., 1997,** "before the normalized impact potentials / resource consumptions are directly comparable, account must be taken of the seriousness of each individual impact in relation to the others". Therefore, weighting factors have been calculated based on scientific, political and normative considerations.

4.4.1.6 EDIP 2003

The EDIP03 is an evolution of the EDIP97 method and includes spatially differentiated characterisation modelling. EDIP97 is not replaced by EDIP03. modeling.

Compared to the EDIP97 methodology, the models underlying the EDIP03 characterisation factors take a larger part of the causality chain into account for all the non-global impact categories. The EDIP03 factors thus include the modeling of the dispersion of the substance and the subsequent expo-sure increase. For a number of impact categories, the modeling also includes the background exposure and vulnerability of the target systems to allow assessment of the exceedance of thresholds.

Therefore, the environmental relevance of the calculated impacts is higher – they are expected to be in better agreement with the actual environmental effects from the substances that are observed, and they are easier and more certain to interpret in terms of environmental damage (Hauschild & Potting, 2005).

New characterization factors and accompanying normalization references have been developed for each of the non-global impact categories:

- acidification
- terrestrial eutrophication
- photochemical ozone exposure of plants
- photochemical ozone exposure of human beings

- aquatic eutrophication
- human toxicity via air exposure
- ecotoxicity

For the global impact categories global warming and stratospheric ozone depletion, the characterization factors are updated with the latest recommendations from IPCC and WMO/UNEP.

The EDIP03 methodology Guideline (Hauschild & Potting, 2005) recommends that the EDIP03 characterisation methodology be used as an alternative to EDIP97 for performing site-generic characterisation (i.e. disregarding spatial information). For the non-global impact categories, the environmental relevance of the site-generic EDIP03 impact potentials is higher, and they provide the option to quantify and reduce the spatial variation not taken into account. EDIP97 can still be used if a new LCA should be compared with prior results based on EDIP97 methodology and factors (Hauschild & Potting, 2005).

4.4.1.7 EPS 2000

The ESP method has been developed in 1990-1991 as a conceptual tool for LCA. The version 2000, implemented in ecoinvent and here with described, is an update of the 1996 version and the 1994 version. EPS system’s rules and terminology comply with the ISO standards for LCA.

To assess the added value from all types of impacts (accounted for); to communicate an understanding of the magnitude of the impact (in monetary terms, for easy weighting against other items that must be considered for product development); to provide a forum for the growth of the environmental strategy of a product.

The EPS system was developed as a tool for designers for product development within companies; use for other purposes like environmental declarations, purchasing decisions, education or environmental accounting requires knowledge of its features and limitations, because the models used to give a measure of impacts may not apply in different contexts; EPS cannot discriminate violations of an emission or quality standards (Edlund, 2001).

4.4.1.8 IMPACT 2002+

IMPACT 2002+ is an impact assessment methodology originally developed at the Swiss Federal Institute of Technology, - Lausanne (EPFL), with current developments carried out by the same team of researchers now under the name of ecointesys-life cycle systems (Lausanne). The present methodology proposes a feasible implementation of a combined midpoint/damage approach, linking all types of life cycle inventory results (elementary flows and other interventions) via 14 midpoint categories to four damage categories (Jolliet et al., 2003). This takes advantages both from midpoint-based indicators such as CML (Guinee et al., 2001) and from damage based methodologies as Eco-indicator 99 (Goedkoop & Spriensma, 2000).

The characterization factors for Human Toxicity and Aquatic & Terrestrial Ecotoxicity are taken from the methodology IMPACT 2002 - Impact Assessment of Chemical Toxics (Pennington et al., 2005). The characterization factors for other categories are adapted from existing characterizing methods, i.e. Eco-indicator 99, CML 2001, IPCC and the Cumulative Energy Demand (Humbert, 2007).

For IMPACT 2002+ new concepts and methods have been developed, especially for the comparative assessment of human toxicity and ecotoxicity. Human Damage Factors are calculated for carcinogens and non-carcinogens, employing intake fractions, best estimates of dose-response slope factors, as well as severities. The transfer of contaminants into the human food is no more based on consumption surveys, but accounts for agricultural and livestock production levels. In addition, the intermittent character of rainfall is considered. Both human toxicity and ecotoxicity effect factors are based on mean responses rather than on conservative assumptions (Humbert, 2007).

4.4.1.9 IPCC 2001 (Climate Change)

The characterisation of different gaseous emissions according to their global warming potential and the aggregation of different emissions in the impact category climate change is one of the most widely used methods in life cycle impact assessment (LCIA). Characterisation values for greenhouse gas emissions are normally based on global warming potentials published by the IPCC (Intergovernmental Panel on Climate Change) (Albritton & Meira-Filho, 2001; Houghton et al., 1996). The figures given in these publications are used not only for the characterisation of greenhouse gases (Guinee et al, 2001; Heijungs et al, 1992) but also within impact assessment methods like Eco-indicator 99 (Goedkoop et al., 1998) or environmental scarcity 1997 (Brand et al., 1998). All these methods evaluate the emissions of greenhouse gases due to anthropogenic activities investigated for the inventory table.

4.4.1.10 TRACI

From 1996 to 2003, the US EPA has focused on determining and developing the best impact assessment tool for Life Cycle Impact Assessment (LCIA), Pollution Prevention, and Sustainability Metrics for the US. A literature survey was conducted to ascertain the applicability, sophistication, and comprehensiveness of all existing methodologies. When the development of TRACI began, the state of the practice involved nearly all US practitioners utilizing European methodologies when conducting comprehensive impact assessments for US conditions simply because similar simulations had not been conducted within the US. Since no tool existed which would allow the sophistication, comprehensiveness, and applicability to the US which was desired, the US EPA decided to begin development of a tool which could be utilized to conduct impact assessment with the best applicable methodologies within each category. This research effort was called TRACI the

Tool for the Reduction and Assessment of Chemical and other environmental Impacts (Rossi, 2007).

The methodology has been developed specifically for the US using input parameters consistent with US locations. Site specificity is available for many of the impact categories, but in all cases a US average value exists when the location is undetermined. The average values were implemented in the ecoinvent data (Rossi, 2007).

4.4.1.11 Selected Life Cycle Inventory Indicators

The list of selected LCI indicators is divided in two: The first list contains the common set of elementary flows shown in the results discussion of the ecoinvent reports. One example is "fossil CO2 emissions to air". The second one contains additional elementary flows used in at least one of the ecoinvent reports. One example of this extended list is "actinides emitted to water".

The selection does not necessarily reflect the environmental importance of the listed pollutants and re-sources. The pollutants and resources are selected in view of a better characterisation of the analysed products and services.

The factors applied in the LCI indicators reflect a mere physical addition without any effect or damage assessment and without final active weighting. Nevertheless, the addition on the basis of physical properties contains an implicit weighting.

The selection helps practitioners to get a more convenient access to a selection of LCI results of products and services. It does not replace the use of the complete set of LCI results and the application of LCIA methods (Frischknecht, 2007).

CHAPTER FIVE
LIFE CYCLE INTERPRETATION

Life cycle interpretation is a systematic technique to identify, quantify, check, and evaluate information from the results of the life cycle inventory (LCI) and the life cycle impact assessment (LCIA), and communicate them effectively. Life cycle interpretation is the last phase of the LCA process. The International Organization for Standardization (ISO) has defined the following two objectives of life cycle interpretation (Curran, 2006):

1. Analyze results, reach conclusions, explain limitations and provide recommendations based on the findings of the preceding phases of the LCA and to report the results of the life cycle interpretation in a transparent manner.

2. Provide a readily understandable, complete, and consistent presentation of the results of an LCA study, in accordance with the goal and scope of the study (ISO 1998).

While conducting the LCI and LCIA it is necessary to make assumptions, engineering estimates, and decisions based on your values and the values of involved stakeholders. Each of these decisions must be included and communicated within the final results to clearly and comprehensively explain conclusions drawn from the data. In some cases, it may not be possible to state that one alternative is better than the others because of the uncertainty in the final results. This does not imply that efforts have been wasted. The LCA process will still provide decision-makers with a better understanding of the environmental and health impacts associated with each alternative, where they occur (locally, regionally, or globally), and the relative magnitude of each type of impact in comparison to each of the proposed alternatives included in the study. This information more fully reveals the pros and cons of each alternative (Curran, 2006).

CHAPTER SIX
REPORTING THE RESULTS

Now that the LCA has been completed, the materials must be assembled into a comprehensive report documenting the study in a clear and organized manner. This will help communicate the results of the assessment fairly, completely, and accurately to others interested in the results. The report presents the results, data, methods, assumptions and limitations in sufficient detail to allow the reader to comprehend the complexities and trade-offs inherent in the LCA study.

If the results will be reported to someone who was not involved in the LCA study, i.e., third-party, stakeholders, this report will serve as a reference document and should be provided to them to help prevent any misrepresentation of the results (Curran, 2006).

The reference document should consist of the following elements (ISO 1997):

1. Administrative Information

- Name and Address of LCA Practitioner (who conducted the LCA study)
- Date of Report
- Other Contact Information or Release Information

2. Definition of Goal and Scope

3. Life Cycle Inventory Analysis (data collection and calculation procedures)

4. Life Cycle Impact Assessment (methodology and results of the impact assessment that was performed)

5. Life Cycle Interpretation

- Results
- Assumptions and Limitations
- Data Quality Assessment

6. Critical Review (internal and external)

- Name and Affiliation of Reviewers
- Critical Review Reports
- Responses to Recommendation

CHAPTER SEVEN
CRITICAL REVIEW

The desirability of a peer review process has been a major focus of discussion in many life-cycle analysis forums. The discussion stems from concerns in four areas; lack of understanding regarding the methodology used or the scope of the study, desire to verify data and the analyst's compilations of data, questioning key assumptions and the overall results, and communication of results. For these reasons, it is recommended that a peer review process be established and implemented early in any study that will be used in a public forum.

The following discussion is not intended to be a blueprint of a specific approach. Instead, it is meant to point out issues that the practitioner or sponsor should keep in mind when establishing a peer review procedure. Overall, a peer review process should address the four areas previously identified:

- Scope/boundaries methodology
- Data acquisition/compilation
- Validity of key assumptions and results
- Communication of results

The peer review panel should participate in all phases of the study: (1) reviewing the purpose, system boundaries, assumptions, and data collection approach; (2) reviewing the compiled data and the associated quality measures; and, (3) reviewing the draft inventory report, including the intended communication strategy.

A spreadsheet, such as the one presented in Appendix A would be useful in addressing many of the issues surrounding scope/boundaries methodology, data/compilation of data, and validity of assumptions and results. Criteria may need to be established for communication of results. These criteria could include showing how changes in key assumptions could affect the study results, and guidance on how to publish and communicate results without disclosing proprietary data.

It is generally believed that the peer review panel should consist of a diverse group of three to five individuals representing various sectors, such as federal, state, and local governments, academia, industry, environmental or consumer groups, and LCA practitioners. Not all sectors need be represented on every panel. The credentials or background of individuals should include a reputation for objectivity, experience with the technical framework or conduct of life-cycle analysis studies, and a willingness to work as part of a team. Issues for which guidelines are still under development include panel selection, number of reviews, using the same reviewers for all life-cycle studies or varying the members between studies, and having the review open to the public prior to its release. The issue of how the reviews should be performed raises a number of questions, such as these: Should a standard spreadsheet be required? Should oral as well as written comments from the reviewers be accepted? How much time should be allotted for review? Who pays for the review process?

The peer review process should be flexible to accommodate variations in the application or scope of life-cycle studies. Peer review should improve the conduct of these studies, increase the understanding of the results, and aid in further identifying and subsequently reducing any environmental consequences of products or materials. EPA supports the use of peer reviews as a mechanism to increase the quality and consistency of life-cycle inventories (Curran, 2006).

CHAPTER EIGHT
THE APPLICATION OF LIFE CYCLE ASSESSMENT TO PROCESS OPTIMISATION

Life Cycle Assessment (LCA) represents an application of system analysis to problems of environmental management. Its embodiment of systems thinking is, at root, no different from the approaches normally used in selecting and designing processes. Yet, despite the fact that, compared to the technical effort required in designing and optimising a process, incorporation of LCA represents only slight incremental effort, the adoption of life cycle approaches by the process industries has been relatively slow. However, recent literature suggests that this attitude is changing and that LCA is gaining wider acceptance in many industrial sectors (Callaghan & Allen, 1995; Baumann, 1996; Curran, 1997; Wright, Allen, Clift & Sas, 1997; Clift, 1998), particularly in the process industries (Franke, Kluppel, Kirchert & Olschewski, 1995; Dobson, 1996; Ophus & Digernes, 1996; Yoda, 1996; Aresta & Tommasi, 1997; Bretz & Fankhauser, 1997). Some other examples of using LCA in corporate decision making include energy (Audus, 1996; Matsuhashi, Hikita & Ishitani, 1996; Tahara, Kojima & Inaba, 1997; Dones & Frischknecht, 1998), nuclear (Griffin, 1997; Solberg-Johansen, 1998), water (Roeleveld, Klapwijk, Eggels, Rulkens & Starkenburg, 1997; Dennison, Azapagic, Clift & Colbourne, 1998), electronic (Langhe, Criel & Ceuterick, 1998; Miyamoto & Tekawa, 1998) and other industries.

There are reasons in addition to disciplinary compatibility to expect the use of LCA in the process industries to expand rapidly. In the European Union, the Directive on Integrated Pollution Prevention and Control (IPPC) (EU, 1996) represents a significant shift in the basis of environmental regulation (Emmott & Haigh, 1996; Nicholas, 1998). IPPC incorporates the principle of integrated pollution control (IPC), introduced in the UK by the 1990 Environmental Protection Act to regulate processes which give rise to different emissions, particularly into different environmental media. However, IPPC goes beyond IPC to embrace the life cycle both of the process (including construction and decommissioning) and of materials and energy (including resource usage and waste) (Nicholas, 1998; RCEP, 1998). IPPC is planned to be implemented by EU member states by October 1999. If applied strictly, IPPC will mandate the use of LCA in identifying the best practicable environmental option (BPEO).

Although the use of LCA has traditionally been oriented towards improving the environmental performance of products (Fava et al., 1991; Tillman, Baumann, Eriksson & Rydberg, 1991; Boustead, 1992; Heijungs et al., 1992; Pedersen & Christiansen, 1992; Fava, Consoli, Dennison, Dickson, Mohin & Vigon, 1993; Guinee, Heijungs, Haes & Huppes, 1993; Keoleian, 1993; Pedersen, 1993; Vigon et al., 1993; Weidema & Kru, 1993; Azapagic, 1997; Fleischer & Schmidt, 1997), several authors have recently demonstrated the previously unexplored potential of LCA as a tool for process selection and BPEO (Golonka & Brennan, 1996; Rice, 1997; Clift & Azapagic, 1998; Yates, 1998), process design (Pesso, 1993; Stefanis, Livingston & Pistikopoulos, 1995; Kniel, Delmarco & Petrie, 1996; Pistikopoulos, Stefanis & Livingston, 1996; Stewart & Petrie, 1996; Stefanis, Livingston & Pistikopoulos, 1997) and optimisation (Azapagic & Clift, 1995; Azapagic, 1996; Azapagic & Clift, 1997; Azapagic & Clift, 1999a,b; Azapagic, Clift & Lamb, 1996). A more detailed exposition of the application of LCA to process selection and design is given elsewhere (Azapagic & Clift, 1999).

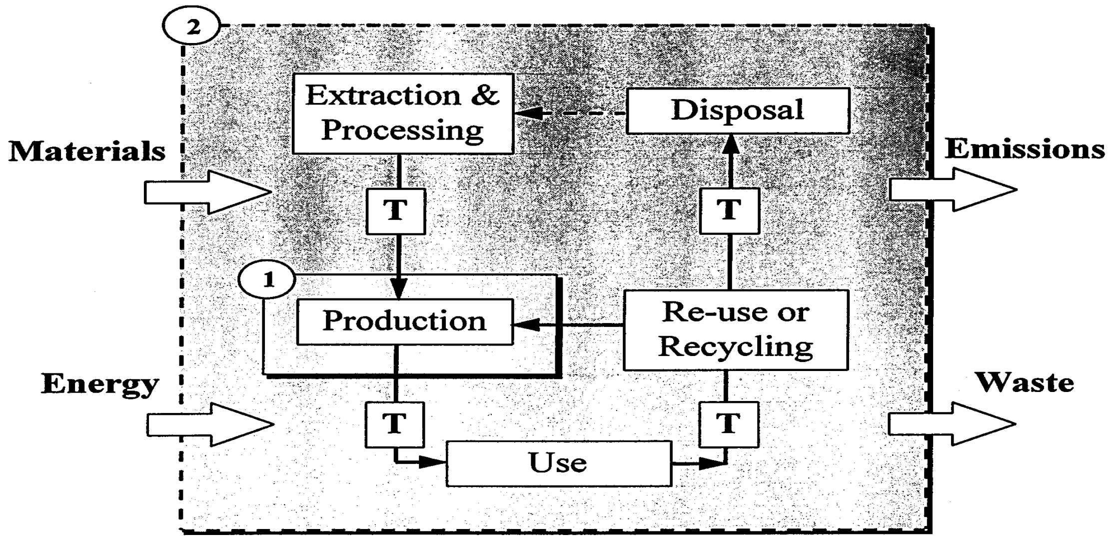

Figure 8.1 Stages in the life cycle of a product (system boundary: 1, process analysis; 2, life cycle assessment; T, transport.

Here, the focus is on the use of LCA for process optimisation. The aim is to show how the kind of analysis adapted from operations research and welfare economics can be combined with system analysis in the context of LCA to provide a powerful decision making tool for more sustainable performance of process industries. The potential of this approach is illustrated by the example of an industrial case study of a mineral-processing system (Azapagic & Clift, 1999).

8.1 LCA and System Optimisation

To describe and predict the behavior of complex industrial systems, it is often necessary to use elaborate mathematical modeling. In the same manner, identification of the optimum operating conditions that will ensure improved process performance usually renders the use of an optimisation technique essential. Historically, system optimisation in chemical and process engineering applications has focused on maximising the economic performance, subject to the certain con straints in the system. Over the past decade, optimisation of environmental performance has started to be incorporated into system optimisation, alongside traditional economic criteria. These approaches have mainly been focused on various waste minimisation techniques (Halwagi & Manousiouthakis, 1990; Ciric & Jia, 1994; Wang & Smith, 1994; Linninger, Stephanopoulos, Ali, Han & Stephanopoulos, 1995). The attempts to incorporate environmental considerations into the design and optimisation procedures represent the beginning of the paradigm shift in the process industry traditionally oriented towards the economic performance of the process. However, the main disadvantage of these approaches is that they concentrate on the emissions from the plant only, without considering other stages in the life cycle. Thus, it is possible for waste minimisation approaches to reduce the emissions from the plant but to increase the burdens elsewhere in the life cycle, so that overall environmental impacts are increased.

Consequently, the need to integrate life cycle thinking into process design and optimisation procedures has been recognised by a number of researchers (Azapagic, 1996; Pistikopoulos et al., 1996; Stewart & Petrie, 1996). One such approach that establishes a link between the environmental and economic performance of a process from 'cradle to grave' has been developed by Azapagic and co-workers (Azapagic & Clift, 1995; Azapagic, 1996; Azapagic et al., 1996; Azapagic, 1997; Bell, Azapagic, Faraday & Schulz, 1998; Azapagic, 1999; Azapagic & Clift, 1999). This method, here referred to as 'Optimum LCA Performance', is presented and discussed in the following sections (Azapagic & Clift, 1999).

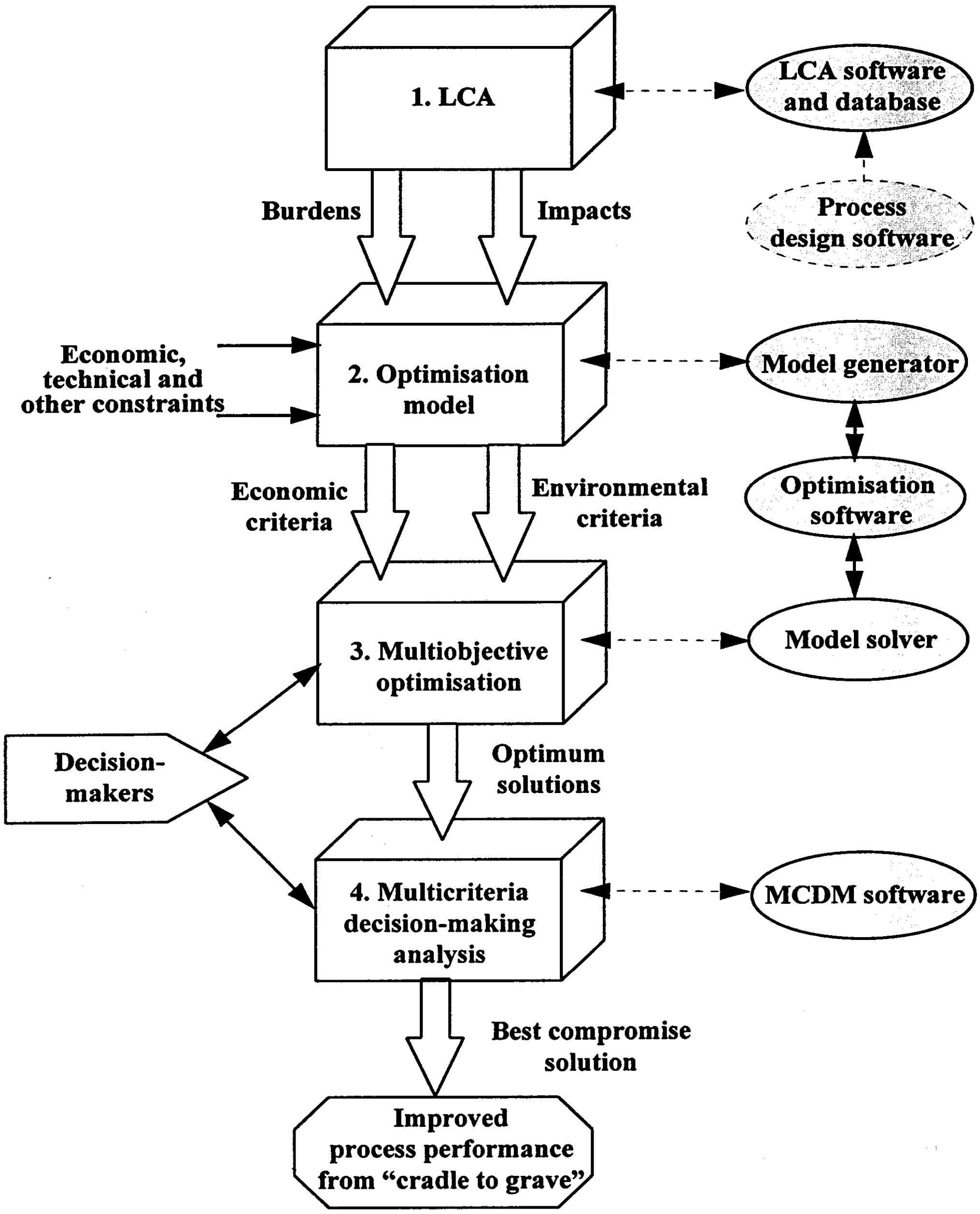

Figure 8.2 The methodological framework for Optimum LCA Performance (OLCAP).

8.1.1 Optimum LCA Performance (OLCAP)

A general framework for the optimum LCA performance (OLCAP) methodology comprises four steps:

1. Completion of the LCA study;
2. Formulation of the optimisation problem in the context of LCA;
3. Multiobjective optimisation (MO) on environmental and economic criteria;
4. Multicriteria decision analysis and choice of the best compromise solution.

The diagrammatic representation of the OLCAP approach is given in Figure 8.2. The first step in this procedure involves carrying out an LCA study of the system, by following the ISO (1997) methodology. As indicated in Fig. 6, appropriate LCA software, e.g. PEMS (PIRA International, 1998) or TEAM (Ecobalance, 1998), can be used to carry out material and energy balances and to quantify the burdens and impacts along the life cycle. The material and energy balances for the process itself (boundary 1 in Figure 8.1) can also be carried out within existing design operation software and these data can then be fed into the LCA software. The data for the other parts of the system (boundary 2 in Figure 8.1) can be sourced from a database which is normally an integral part of the LCA software. A more detailed exposition of the LCA methodology is given elsewhere (ISO, 1997) and is not discussed further here.

The environmental burdens and impacts quantified in step 1 represent an input into the optimisation model, which is formulated in step 2. In addition to environmental criteria, the model includes economic, technical, legislative and other constraints within which the system must operate. In step 3, the system is optimised on environmental and socioeconomic objectives of interest to the decision-makers, to yield a number of optimum solutions. A suitable optimisation technique and software must be used to generate and solve the optimisation problem. Finally, step 4 enables the decision makers to choose the best compromise alternative from a range of optimum solutions. Any of the multi criteria decision making techniques, some of which have been formalised in various software packages, can be used to facilitate the decision-making process (Azapagic & Clift, 1999).

8.1.2 Formulation of the Optimisation Problem

Because of the nature of LCA, where there are a number of distinct environmental burdens or impacts to be considered, optimisation problems in this context are inevitably multiobjective. Thus, conventional single-optimisation problems, involving one (usually economic) function are transformed into multiobjective problems, to include the environmental objectives. A Multi-Objective (MO) problem in the context of LCA can take the following form:

$$\min f(x, y) = [f_1, f_2, \ldots\ldots\ldots\ldots\ldots, f_n] \tag{8.1}$$

$$h(x, y) = 0$$

$$g(x, y) \leq 0$$

$$x \in X \subseteq R^n$$

$$y \in Y \subseteq Z^q \tag{8.2}$$

where $\boldsymbol{f}$ is a vector of economic and environmental objective functions; $h(\boldsymbol{x}, \boldsymbol{y})$_0 and $g(\boldsymbol{x}, \boldsymbol{y})$50 are equality and inequality constraints, and x and y are the vectors of continuous and integer (discrete) variables, respectively. For instance, the equality constraints may be defined by energy and material balances; the inequality constraints may describe material availabilities, heat requirements, capacities etc. A vector of n continuous variables may include material and energy flows, pressures, compositions, sizes of units etc., while a vector of q integer variables may be represented by alternative materials or processing routes in the system. If the integer set Z is empty and the constraints and objective functions are linear, then Eqs. (8.1) and (8.2) represent a Linear Programming (LP) problem; if the set of integer variables is nonempty and nonlinear terms exist in the objective functions and constraints, Eqs. (8.1) and (8.2) is a Mixed-Integer Nonlinear Programming (MINLP) problem. Mixed Integer Linear Programming (MILP) problems incorporate integer and linear variables only.

An economic objective typically involves a cost or profit function as defined by:

$$\min F = c^T y + f(x) \tag{8.3}$$

where c is a vector of cost or profit coefficients for integer variables and $f(x)$ is a linear or nonlinear function described by continuous variables. The environmental objectives in this context represent the burdens B_j or impacts E_k:

$$\min B_j = \sum_{n=1}^{N} b_{j,n} x_n \tag{8.4}$$

$$\min E_k = \sum_{j=1}^{J} e_{k,j} B_j \tag{8.5}$$

where $b_{j,n}$ represents emission coefficients associated with continuous variables xn. In Eq. (8.5), e_k, j represents the relative contribution of burden B_j to impact E_k, as defined by the 'problem oriented' approach to Impact Assessment (Heijungs et al., 1992). In this approach, for example, GWP factors, ek, j, for different greenhouse gases are expressed relative to the GWP of CO_2, which is therefore defined to be unity. If a different impact assessment approach is used, then Eq. (8.5) may be redefined accordingly. Note that at present the LCA approach assumes that environmental burdens and impacts functions are linear, i.e. they are directly proportional to the output of functional unit(s) and there are no synergistic or antagonistic effects.

8.1.3 Multiobjective Optimisation

The system is then optimised simultaneously on a number of environmental and economic objective functions to locate the multidimensional noninferior or Pareto surface which maps the optimal solutions. By definition, the noninferior state is achieved if no objective can be improved without worsening the value of some other objective. If examined more closely, it is obvious that this definition is identical to the Pareto optimality concept (Pareto, 1971) which marked the beginning of new welfare economics and has been influencing decision-making process ever since. Welfare economics, although historically divided into several periods, focuses on the general problem: how should resources is allocated for the production and consumption of goods so as to maximise social welfare? Although this predates the sustainability concept of today, the question asked remains the same; what changed over time, however, was the definition of 'social welfare' and the approaches to solving this problem.

The choice of environmental objectives for optimisation depends on the Goal and Scope of the study. Thus, optimisation can be performed either at the inventory or impact assessment levels, in which case the environmental objectives are defined as either burdens or impacts, respectively (Azapagic & Clift, 1999). In optimisation, local and global system improvements are found by first moving the system to conditions on the Pareto surface, and then 'surfing' on it. As already pointed out, all objectives on the surface are optimal in the Pareto sense and trade-offs between the objectives are necessary to identify the best compromise solution. For example, if the system is optimised simultaneously on two objectives-one economic and one environmental the resulting Pareto optimum does not necessarily mean that these functions are at their respective optima achieved when the system is optimised on each of them separately. The Pareto optimum, however, does mean that the set of best possible options has been identified for a system in which both objectives should be improved. This can be of particular relevance to the chemical and process industries, which face problems of having to keep total costs down while at the same time complying with ever tightening environmental legislation and other socioeconomic requirements.

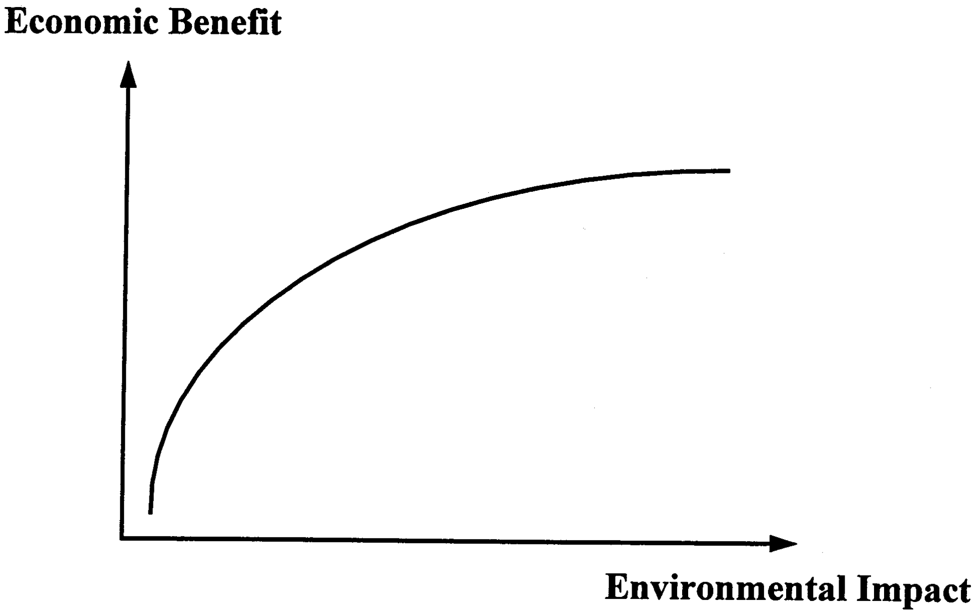

Figure 8.3 Noninferior curve obtained in multiobjective optimisation.

One possible approach to optimisation in the context of LCA would be to aggregate environmental and economic objectives into a single function by attaching weights to indicate their significance, so that the problem reduces to single objective optimisation. However, one of the main advantages of MO is that it does not require a priori articulation of preferences, so that the whole no inferior set of solutions can be explored. The emphasis is then on the range of choices from the set of no inferior solutions, rather than explicit definition of preferences before analysing all the trade-offs among objectives. Trade-offs between the no inferior solutions shows explicitly what can be gained and what lost by choosing each alternative. Where there are multiple decision-makers with conflicting interests, this technique can help to resolve disputes by generating different alternative solutions. Decision makers who understand the trade-offs and the alternatives are more likely to understand the interests of other parties and, therefore, to compromise. Although the evaluation of trade-offs between the objectives to choose the best compromise solution will still imply certain preferences and value judgments, at least the choice will be made from all possible no inferior solutions.

Furthermore, by being able to trade-off incommensurable objectives, e.g. environmental impacts and economic requirements, this approach avoids the well known problems encountered, for instance, in cost– benefit analysis (Pearce, Markandya & Barbier, 1989), i.e. reducing individual preferences to a market value or trying to express quality of the environment in financial terms. Cost-benefit analysis (CBA) is probably the tool most exploited by neoclassical1 economists in the decision- making process, particularly in the area of public investments. CBA is based on the idea of maximum net gain: it reduces aggregate social welfare to the monetary unit of net economic benefit. So for example, given several alternatives, the CBA approach would favor the one in which the difference between monetarised benefits and costs is the greatest. More recently, CBA has been applied in environmental decision-making. The most widely applied, and even more criticised, technique is 'contingent valuation' (CV). In CV, participants are asked to say how much they would be prepared to pay to protect an environmental asset ('willingness to pay') or how much they would be willing to accept for loss of that asset ('willingness to accept').

Limitations and difficulties of this approach have been recognised both by its proponents and critics. The latter (Jacobs, 1991; Adams, 1993; Clift, 1994) have pointed out that CBA has serious difficulties in dealing with problems of intergenerational equity and sustainability and in valuing the natural environment. They have also shown that CV is based on individual preferences which may not provide firm foundations for environmental decision-making. Furthermore, the results of the analysis largely depend on the way the questions are asked, and whether the participants are familiar with the asset in question. It is more likely that people who know nothing about the asset will place a nil value on it, although the life of others may depend on it. Also, the values that people place on things strongly depend on self-interest, which does not help resolving conflict between opposing parties.

To summarise, CBA and related economic approaches to decision making face at least three problems: the measurement of individual preferences, the interpersonal comparison of these preferences, and their aggregation into a social preference function. All these operations imply ethical value judgments, probably the least acceptable being the expression of individual preferences and values in monetary terms. Indeed, the controversial techniques of pricing nonmonetary objectives, such as environmental quality, and aggregating non-commensurable into a single 'utility' function provide a strong motivation for using multiobjective analysis in environmental decision making.

Furthermore, these approaches cannot provide information for decision-making on a 'local' level: for example, they cannot advise engineers on how to modify a process in order to improve its environmental performance. MO, on the other hand, does exactly this: it can optimise the operation of a system with environmental, technical, economic and other aspects taken into account. If applied in the LCA context, it can optimise the whole life cycle of a process or product and so provide a more effective approach to environmental management of a system (Azapagic & Clift, 1999).

8.1.4 Choice of the Best Compromise Solution

The no inferior solutions, obtained in step 3, provide input into the decision-making process in step 4 of OLCAP. To choose the best compromise solution out of a number of optimum alternatives, some articulation of preferences is necessary. However, these preferences are at least articulated by decision-makers in the post optimal analysis of all no inferior solutions and their trade-offs, as distinct from expressing preferences and aggregating the objectives prior to identifying all no inferior solutions. One of the possible ways to choose the 'best' solution is to consider a graphical representation of the no inferior set and then choose the best compromise solution on the basis of the trade-offs. However, this approach is limited to two or three objective functions at most; beyond that, graphical representation becomes too complex. Alternatively, the no inferior values of the objectives may be expressed in terms of the difference from the value at their individual optima. If all objectives are considered to be of the same importance, than the best compromise solution might be that which equalises the percentage by which all objectives differ from their optimum values. However, should any of the objectives be considered more important than the others, then other methods that allow ordering and quantifying of preferences, usually referred to as multicriteria decision making (MCDM) techniques, can be used to identify the best compromise solution.

MCDM techniques provide a structured approach to a decision making process. They enable systematic analysis and modeling of preferences with the aim of providing help and guidance to decision-makers in identifying their most desired solution. The major advantages of these techniques are that they are transparent, non-ambiguous and easy to use by non experts. Furthermore, the quantitative nature of these numerical methods may particularly be appealing to quantitatively oriented managers and engineers.

A number of methods for ordering and quantifying preferences have been developed over the past years and some of them include simple additive weighting, weighted product, median ranking method (Hwang, Paidy & Yoon, 1980), the analytic hierarchy process (Saaty, 1980), multiattribute utility theory (Keeney & Raiffa, 1976), simple multi-attribute rating technique (von Winterfeldt & Edwards, 1987). Extensive reviews of MCDM techniques can be found in Stewart (1992) and Yoon and Ching (1995). User friendly software with various MCDM methods to aid the decision making process are also available.

The choice of a suitable MCDM technique will depend on a given decision-making situation and the sophistication of the decision-makers. Most of these techniques are based on a definition of a multiattribute or utility function, which associates a number with each alternative to reflect the importance of the attribute in the opinion of the decision-maker, so that all alternatives may be ordered. For example, if there are five no inferior solutions identified in step 3, each with different values for the three objectives (attributes), i.e. GWP OD and costs, the decision-makers are then asked to articulate their preferences for each of the attributes on scale 1–10. The mathematical analysis or ordering of the preferences, for instance by a pair-wise comparison of attributes (Saaty, 1980), returns the best compromise solution for this particular example. It is important to note that the attributes and the preferences are always identified on a case by case basis within a bounded decision space, and that they only apply in that particular decision-making context. This avoids the criticism often voiced, in both LCA and CBA, of trying to use general weights or costs to indicate the importance of distinct criteria in different decision making situations (Azapagic & Clift, 1999).

CHAPTER NINE
STREAMLINING LCA

A continuing concern is the cost and time required for LCA. Some have questioned whether the LCA community has established a methodology that is, in fact, beyond the reach of most potential users. Others have questioned the relevance of LCA to the actual decisions that these potential users must make. These concerns have encouraged some practitioners to investigate the possibility of "streamlining" or simplifying LCA to make it more feasible and more immediately relevant without losing the key features of a life cycle approach.

When the concept of streamlining was first introduced, many LCA practitioners were skeptical, stating that LCA could not be streamlined. Over time, however, there has been growing recognition that "full-scale" LCA and streamlined LCA are not two separate approaches but are, instead, points on a continuum. Most LCA studies will fall somewhere along that continuum, in between the two extremes. As a result, streamlining an LCA becomes part of the scope and goal definition process. For example, as the study team decides what is and is not to be included in the study, they are engaged in streamlining - in addition to determining what will and will not be included, the study team will determine how to best achieve these requirements. The key is to ensure that the streamlining steps are consistent with the study goals and anticipated uses, and that the information produced will meet the users' needs. From this perspective, the scope and goal definition process involves determination of what needs to be included in the study to support the anticipated application and decision.

Other tools are being proposed which use the life cycle concept by looking across the life cycle stages of a product or process system, but in shortened versions. For example, an abridged LCA approach was developed and used by others in industry, such as Motorola. However, there is the potential that some attempts to shorten the LCA approach result in looking at each life cycle stage individually and lose the ability to identify environmental trade-offs between stages.

For example, in looking at the waste management stage of a consumer product, solid waste is identified as an area for improvement, such as increasing recycling. But this single-issue focus does not take into account the additional energy and potential environmental impacts that result from recycling operations through transportation and reprocessing. It seems that often environmental activities, such as Design for the Environment and Pollution Prevention, are based on a type of life-cycle "thinking" but are void of consideration of a trade-off analysis across the entire life cycle. Both DfE and P2 intend to include trade-off analysis within a given, site-specific application, such as a manufacturing plant, but neglect to consider the potential environmental impacts that may result outside their boundaries as part of the assessment. The bottom line is that in order for a study to be called an LCA it must be multi-media (quantify releases to air, water, and land), include all the life cycle stages from cradle to grave (raw material acquisition, manufacture, use/reuse, recycling, and disposal), and include some type of impact assessment upon which the results are interpreted. Further, the interconnectedness of the life cycle stages is an important aspect of interpretation through trade-off analysis (Curran, 2005).

9.1 Future Direction in LCA Development

While LCA use and activity is constantly increasing, there are several barriers that are prohibiting its wide-spread adoption. The three key barriers are (Curran, 2005):

1. Lack of awareness of the importance of using the life cycle concept,
2. Inaccessibility to life cycle inventory data and a measure of the quality of the data, and
3. Lack of understanding of impact assessment methodology and identifying what type of modeling is appropriate for the specific application.

Lack of awareness of the importance of the life cycle concept producers and decision-makers need to be made aware of the life cycle impacts that their activities carry and the importance of going beyond meeting compliance. More importantly, government offices that issue media-based or industry-focused regulations and policies need to begin using life-cycle thinking. There are numerous instances where life-cycle thinking is potentially beneficial in making public policy. Introducing LCA concepts into the rule-making process extends the regulatory analysis upstream and downstream and across all media to account for the effects of the proposed standard which may otherwise escape a traditional regulatory impact analysis.

Inaccessibility of Reliable LCA Data-Lack of data has hindered, perhaps prevented, many applications. Several efforts are underway in North America and Europe to make data more easily accessible. The Society for the Development of Life Cycle Assessment (SPOLD) developed a data exchange format that defines a logical structure for LCI data. The Swedish SPINE initiative proposed a standardized data model and a relational database structure that should serve as a "common language" for LCA. A SETAC-Europe workgroup on "Data Availability and Data Quality" was formed at the SETAC Europe Annual Meeting 1998 in Bordeaux, France, to investigate methods to improve the availability and the free exchange of LCI data, assess and improve data quality, develop uncertainty measures, and establish robustness checks. During its three-year term, the workgroup intends to produce guidance documents to help increase the usefulness and credibility of LCA. The workgroup is also working through the ISO process to propose the development of guidance under ISO 14000 series.

Lack of an Impact Assessment Method.-This seems to be more of a barrier in the US than in Europe where several attempts at LCIA have been published. However, there is no consensus on what methodology should be followed. Although the development of LCIA methodology is in the early stages, developers are beginning to recognize that a slate of impact assessment approaches defined by the study goal may be more appropriate than attempting to develop a "one-size-fits-all" approach (Curran, 2005).

9.2 LCA in Environmental Decision-Making

The split between the scientists and engineers who are trying to develop a scientifically-defensible tool and the business managers and policy makers who are trying to make sound environmental decisions is seen clearly within the environmental community in the US. Recently, a SETAC - North America workgroup on environmental decision-making tools and techniques started an effort with the goal of integrating the myriad of decision-support tools and techniques that are available to support decision-making. Initial discussions with decision-makers found that they are not interested in having tools, but instead want the information they need to help them make a decision. The result is the growing realization that the life cycle concept has grown beyond being simply a tool to compare products but is now seen as an essential part of achieving broader goals such as sustainability (Curran, 2005).

9.3 Interest in LCA Approaches is Growing Internationally

The very reasonableness of the life cycle concept in thinking about the entire spectrum of the environment is very appealing and hard to argue against. This, along with the LCA work being done under the International Standards Organization (ISO) 14000 on Environmental Management, is increasing awareness as people strive to learn more about the concept and how to do an assessment. LCA is beginning to be viewed as an important part of environmental management. Encouraging life cycle assessment, and actions based on such assessments, is a natural and necessary step in environmental management. Life cycle assessment can lead to identifying industrial environmental improvements and, in turn, reveal important economic and resource savings opportunities. It is generally acknowledged that trying to use traditional administrative processes to control pollutants throughout product life cycles would be very stifling to economies and probably ineffective environmentally. Thus, the growing attitude is to encourage private entities to adopt environmental management systems, and carry out self-analysis. In addition to comparing the environmental soundness of products, LCA is also being used to assess applications within industrial processes, such as supplementing pollution prevention activities. The following two examples demonstrate how LCA has been used to evaluate options for material substitution and raw material sourcing (Curran, 2005).

Example 1 - Solvent Substitution Using Aqueous Cleaners

While aqueous cleaners offer a suitable substitution for chlorinated cleaning solvents, energy use with aqueous cleaners may generally be higher than that required for chlorinated solvents. They generally require pretreatment prior to discharge to a POTW in order to adjust the pH, remove oil, grease and solids and to precipitate phosphates and inactive chelating agents. These pretreatment results in the use of energy to run equipment as well as generation of waste streams that must be disposed another consideration is that heating may be required during the cleaning process.

Example 2 - Biobased Feedstocks for Chemical Production

An alternative to natural gas-derived feedstock to produce1, 4-butanediol (BDO) is a feedstock process that is based on the fermentation of corn-derived glucose to succinic acid, followed by catalytic reduction to BDO. The higher energy use of the alternative process indicates that the overall environmental consequences would be greater than the conventional process. Because electricity generation is inefficient, and energy production in the US is mostly coal-based, the alternative process was analyzed to have a greater potential for impact in multiple impact categories, including global warming, acid rain, smog, water use, particulates, and solid waste (ash) disposal.

9.3.1 LCA within Industry

ISO 14000 has been both a help as well as a hindrance to LCA advancement. Its existence has been very instrumental in increasing the awareness of the life cycle concept within the environmental community. The development of the documents on

LCA (14040 on General Principles, 14041 on Inventory, published the at end of 1998, and 14042 on Impact Assessment and 14043 on Interpretation, both in draft (DIS) stage) have been very helpful in pulling the current thinking of LCA methodology together and making it available to the general public. ISO 14000 is a step in the right direction but there still remains a need to clarify terms and provide good methodology and data that can be applied to accomplish the goals of each study.

Many companies either continue or are starting to use the LCA concept for internal checks on their performance but are cautious to use the results in a public forum. This caution may also be attributable to the ISO14042 document on Life Cycle Impact Assessment that places rigorous reporting requirements on the use of LCA results in a "comparative assertion" (i.e. an LCA that is used to make a market claim that one product is better overall for the environment). Within industry, interest in LCA is driven by the larger, usually multi-national, companies. These companies often apply LCA to their products to identify areas for environmental improvement. They may work closely with their suppliers in order to ensure a continuous supply of preferred materials, e.g. recycled packaging.

For the most part, US companies stay at the inventory level of methodology and focus on quantifying the inputs and outputs of the life cycle. In this way, the practice is still basically at the "less is best" level. In general, there is a feeling of frustration in US industry, which wants to do LCA but is looking for the definitive, simple, relatively inexpensive and timely approach to do it. Further, there is still the underlying belief that an LCA can be used to get any answer the study sponsor wants. Because there doesn't seem to be a single tool that can be applied and give reproducible results regardless of who does the study, many remain skeptical about the usefulness of LCA. The many pollution control regulations imposed on US industries leave few companies able to see the need or benefit of going "beyond regulatory compliance." Often for smaller companies it is not so much a matter of need but of necessity where resources are limited and they must use what they have to comply with existing regulations. Other larger companies, however, are seeing the possible benefits of looking holistically at their operations. To them LCA is a way to be proactive in environmental management by heading off potential problems, as well as benefitting from an improved corporate image.

While some companies have attempted life cycle impact assessment, the tendency has been to avoid using any formal approach to impact assessment, putting the U.S. practice behind European practice. In Europe, the majority regard LCA as a supporting tool for decision-making. Normative elements are not a problem as long as good procedure is followed with a clearly defined input from stakeholders, and as long as the results are presented in a transparent way (Curran, 2005).

9.3.2 LCA within Government

There are numerous international cases where life cycle concepts are potentially beneficial in making public policy. It is already becoming known that public life cycle assessments can provide very useful insights for policy-making. Examples can be found in several countries demonstrating a growing interest in integrating the life-cycle concept into different types of policies. Public life cycle assessments could be a valuable means of testing ways to implement a life cycle "approach"; in particular, as governments have extensive experience using expert panels and public hearings to generate information and discuss options. These same mechanisms could also be used to strengthen the state-of-the-art in life cycle opportunity assessment and decision-making. As experience is gained in the US with life cycle assessment, it is anticipated that government offices will begin to review their own administrative requirements to see how they conform to the insights being gained about the most cost-effective ways to reduce pollutants.

The governments within the European Union have been much more willing to use life cycle assessment approaches in developing policies and so lead in experience. Given the economic and environmental insights that flexible use of this approach should bring, it is vital to build on that experience and integrate it on a wider scale within the US. In the US, adoption of the life cycle concept within the government is lagging behind industry. A more limited from of the life cycle concept called Life Cycle Costing has been used by DOE since the late 1960's in analyzing energy use in certain processes and products. Because of this the DOE, as well as DOD, offices continue to be more open to adopting the life cycle concept. The USEPA regulatory offices, which are structured by air, water, and waste concerns, have been slower to integrate life cycle thinking into the development of regulations and policies although a few scattered examples can be found which indicate that this situation is changing. The prime example is Executive Order 13101 on "Greening the Government through Waste Prevention, Recycling, and Federal Acquisition" which requires the USEPA to issue guidance on the preference and purchase of environmentally preferable products. The proposed guidance encourages the use of a life cycle approach. In addition, a few attempts have been made in the USEPA's Office of Water and the Office of Pollution Prevention and Toxic Substances in using life data in the development of emission standards (Curran, 2005).

USEPA regulatory offices continue for the most part to follow their set lines of responsibility and maintain a single-issue focus, however, the life cycle concept is slowly being introduced in policy discussions, and the Office of Research and Development continues to support a strong LCA research program. The primary interest is in assisting in the development of guidelines and databases for use in the public and private sectors. The USEPA inventory guidance document published in 1993 has been followed by other documents furthering the methodology and showing applications. The US Departments of Energy and Defense (DOE and DOD) have worked with the EPA in developing LCA tools and data. As a result of Executive Order 13101 on "Greening Government" the Agency has established five guiding principles on environmental preferability of products. Although the guidance does not call out a full life cycle assessment as the way to evaluate the preferability of products, USEPA's Systems Analysis Branch prepared guidance on the use of life cycle assessment for exactly that end. The new guidance outlines how a life cycle assessment supports environmental preferability, through a new tool called FRED (Framework for Responsible Environmental Decision-making) [8]. By setting many of the parameters of a life cycle assessment, FRED simplifies data collection and provides a more uniform format. This better provides for comparisons between disparate products. So far government offices have refrained from being prescriptive about life cycle assessment methods, seeing the practicality of such methods within the decision-making framework of companies and not as a stand-alone tool (Curran, 2005).

CHAPTER TEN
LITERATURE REWIEV

In this chapter life cycle assessment methodology is applied to different treatment systems. For wastewater treatment plants applying the life cycle methodology is a difficult method. In this study we tried to find out how life cycle analysis can be applied to a wastewater treatment plant. The following parameters can be used for applying LCA of wastewater treatment plants.

using resources		obtaining raw materials		emissions
using energy	→	assembling of the plant	→	to air, water
influent and effluent		management of the plant		and land
wastewater characterization		dismantling and disposal of the plant		

According to this concept, to analyze a wastewater treatment plant we must have the following data;

- Description of the geographic area where the plant will be built
- The materials used during the construction of the plant (Cement and steel etc.)
- Amount of wastewater and the wastewater characterization
- Amount of the chemicals (If it is used)
- Amount of the energy used for plant operation. (Electricity, natural gas etc.)
- Information about the plant operation like treatment units
- Amount of emissions to air, water and solid formed
- Amount of sludge formed
- The disposal of the sludge
- Information about the disposal method of the wastes
- The effluent water characteristic
- Receiving area discharge criteria
- The effluent water characteristic
- Received area discharge criteria

In the following steps, we will give some examples of LCA of the wastewater treatment plants. Generally as a functional unit, $1m^3$ treated wastewater uses in LCA of the wastewater treatment plants.

Here are some examples to understand LCA application methodology for wastewater treatment systems;

10.1 Example 1 - Comparing Different Wastewater Treatment Systems with Using Life Cycle Assessment Methodology.

This example presents the preliminary results of Life Cycle Assessment (LCA) study comparing different wastewater treatment works, operated by Thames Water Utilities Ltd. In the UK fifteen works have been studied, representing a range of size and type of treatment works. Five management regimes for centralising sludge treatment and disposal were analyzed in the context of LCA to provide guidance on choosing the best practicable environmental option (Dennison & Azapagic, 1998).

This example considers the management of 15 wastewater treatment works within a particular Figure 10.1. The Water Service Utilities of geographical region, under the management of Thames England and Wale, Water Utilities (Ltd Figure 10.1).This includes a spectrum of large works, treating up to $11 * 10^9$ tones wastewater per year, to small works, treating $8 * 10^6$ tones wastewater per year (Table 10.1). Under the current management regime raw sludge is disposed to land from eleven of the works; this practice is being phased out where practical to minimise potential or perceived health risks. The sludge arising from smaller works is either transported directly to land by tanker for disposal; or to a larger works from where it is then disposed directly to land (Dennison & Azapagic, 1998).

Figure 10.1 The Water Service Utilities of England and Wales.

Current Management Regime

The method and amount of wastewater treated differs between the Thames Water sites (Table 10.1). The current management regime involves direct land disposal of sludge by sub-soil injection from nine works. The sludge arising from the remaining works is either transported directly to land by tanker for disposal, or to a larger works from where it is then disposed to land (Figure 10.2) (Dennison & Azapagic, 1998).

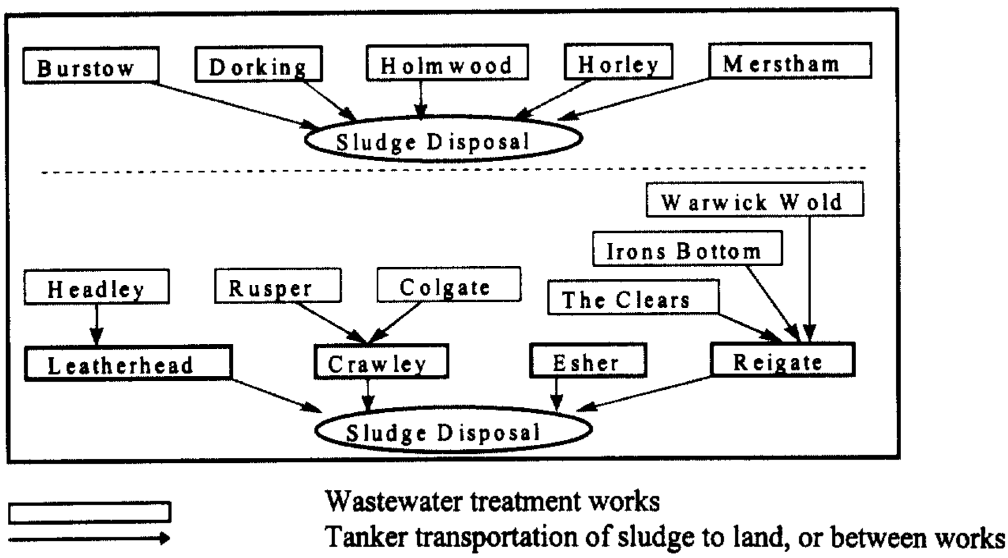

Figure 10.2 Diagrammatic representation of the current management regime.

Table 10.1 The fifteen wastewater treatment works considered.

Wastewater treatment works	Method of treatment	Tonnes of raw wastewater treated per annum ($x10^6$)	Type of sludge disposed to land, disposal route & mean distance travelled (km)
Esher	0,1,AS+PF	11244	Raw sludge direct to land - 32
Crawley	0,1,AS	10739	Digested cake direct to land - 18
Leatherhead	0,AS	4114	De-watered sludge direct to land - 26
Horley	0,AS,3	2824	Raw sludge direct to land - 21
Dorking	0,1,AS,3	2106	Digested sludge direct to land - 22
Reigate	0,1,PF	1639	Digested, thickened sludge direct to land - 19
Burstow	0,1,PF,3	1060	Raw sludge direct to land - 30
Merstham	0,1,PF	784	Raw sludge direct to land - 22
Holmwood	0,1,PF	615	Raw sludge direct to land - 20
Rusper	1,PF	38	Raw sludge to land via Crawley - 44
Headley	1,RBC	26	Raw sludge to land via Leatherhead - 46
The Clears	0,1,PF	16	Raw sludge to land via Reigate - 47
Colgate	1,PF	14	Raw sludge to land via Crawley - 44
Irons Bottom	1,PF	11	Raw sludge to land via Reigate - 37
Warwick Wold	1,PF	8	Raw sludge to land via Reigate - 49

Key:			
0	preliminary treatment	AS	secondary treatment by activated sludge
1	primary treatment	PF	secondary treatment by percolating filters
3	tertiary treatment	RBC	secondary treatment by rotating biological contactors

Proposed Management Regimes

It is the policy of Thmaes Water to centralise, where possible, sludge treatment and disposal. A proposal to upgrade the method of treatment at Crawley presented an opportunity to improve the sludge management of the surrounding works. Various options were considered for the Crawley Sludge Centre, including (Dennison & Azapagic, 1998):

Option 1: maintain current operations (Table 10.1 and Figure 10.2).

Option 2: increase digestion facilities at Crawley and tanker the sludge from all works, except Esher Leatherhead and Headley which remain unaltered, to Crawley for further treatment and disposal.

Option 3: the same as Option 2, but composting is adopted rather than increasing the digestion facility at Crawley (Figure 10.3).

Option 4: increase digestion at Crawley and tanker the sludge from all works to Crawley for further treatment and disposal (Figure10.4).

Option 5 the same as Option 4 but composting is adopted rather than increasing the digestion facility at Crawley.

In economic terms there is little differens between Options 2 to 5. For example the Capital Expenditure (CAPEX) constituted approximately %15 of the net present value over 20 years (NPV) for each Option. The CAPEX for Options 2 and 3 and 4 and 5 was estimated to differ by £300K, insignificant compared to NPVs of £37 and £44million for each respective pair of Options. This demonstrates that CAPEX does not always give a definitive indication of the preferred option when planning capital schemes. Accounting for additional factors such as environmental impacts could more fully inform the decision making process. The purpose of this work is to quantify the environmental impacts of the proposed management regimes and to determine if quantitative consideration of these impacts alters the feasibility of the regimes proposed, enabling a clearer to be made (Dennison & Azapagic, 1998).

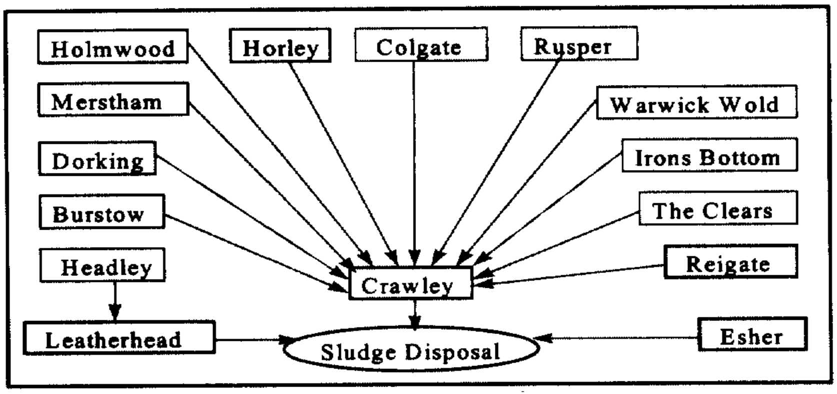

Figure 10.3 Diagrammatic representation of the proposed management regime: option 3 with composting at Crawley.

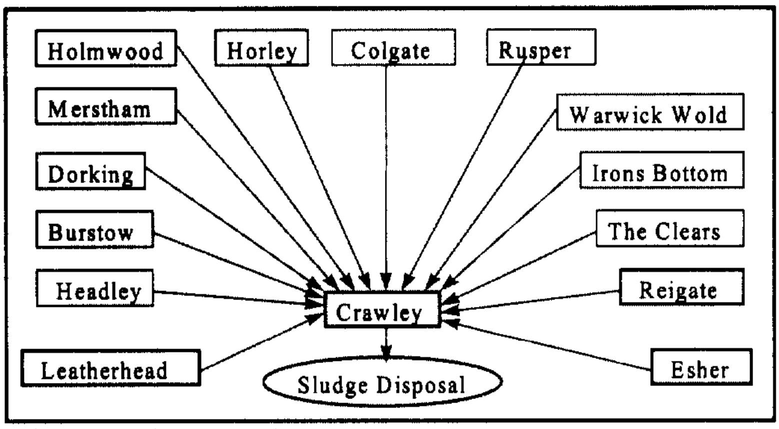

Figure 10.4 diagrammatic representation of the proposed management regime: option 4 with increased digestion at Crawley.

LCA of Wastewater Treatment Works and Associated Sludge Disposal

To quantify the environmental impacts associated with the various management regimes, and thus provide a basis for comparing the results, the functional unit was taken to be 9.4 * 10^8 kg of raw wastewater treated with subsequent sludge disposal. This is equivalent to the average mass of raw wastewater treated at Esher per month. Sludge disposal is modeled as sub-soil injection and allowance was made for the avoided burdens from the fertiliser value of the sludge. The system boundary is shown in Figure 10.5 (Dennison & Azapagic, 1998).

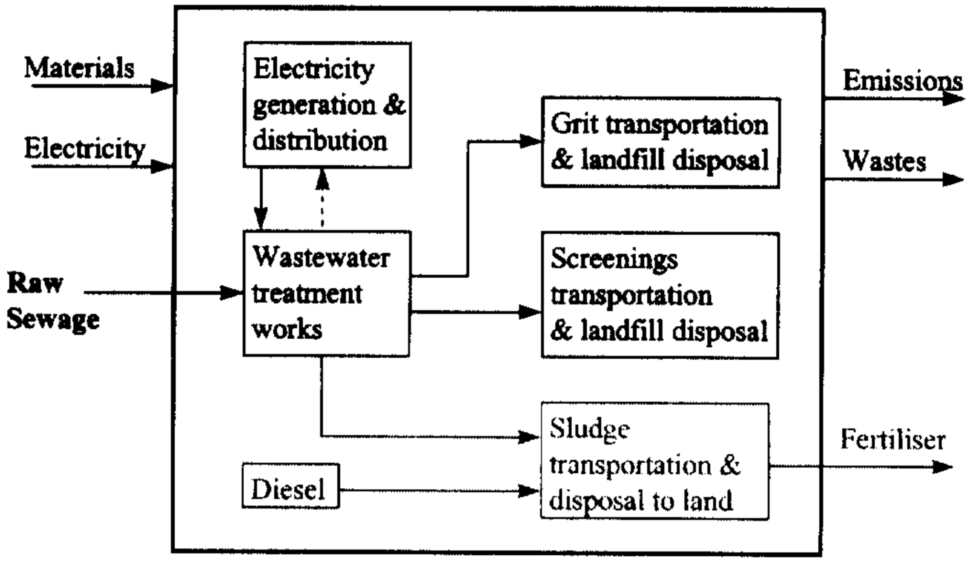

Figure 10.5 Flow diagram and system boundary indicating the process stages modeled for each works.

Where appropriate, the process stages indicated in Figure 10.5 were modeled for each works. For example electricity is only fed back into grid from Crawley via Combined Heat and Power plant; there is no electricity use at Colgate; grit and screenings disposal is site specific and only occurs at Holwood and larger works (Table 10.1). The treatment of wastewater at each site and subsequent transport of sludge (either directly to land, or via a larger works) was modeled for each of the five options. The application of compost to land (Options 3 and 5) was not modeled, but the sub-soil injection of sludge was (Options 1, 2 and 4). The production of polyethylene packaging for the distribution of compost was included in the study (Options 3 and 5) (Dennison & Azapagic, 1998).

The impacts arising from each works, under each management option, were summed to indicate the total environmental impact associated with each management regime. The environmental impacts considered include global warming potential, acidification and eutrophication. Global warming potential is a measure of the potential contribution of different gases to the greenhouse effect; it is calculated using carbon dioxide (CO_2) as a reference gas. Acidification as a measure of the phenomena known as acid rain which is caused by gaseous pollutants, it is calculated on the basis of hydrogen ions which can be produced per mole of sulphur dioxide (SO_2). Eutrophication is a measure of an increase in biomass due to the addition of nutrients to water or soil, it is calculated with reference to the capacity of phosphate (PO_4^{3-}) to form biomass (Dennison & Azapagic, 1998).

Assessing the Current Management Regime

According to me, the treatment process and sludge disposal in Burstow (Table 10.1) causes too many environmental impacts such as acidification and eutrophication (Figure 10.6).

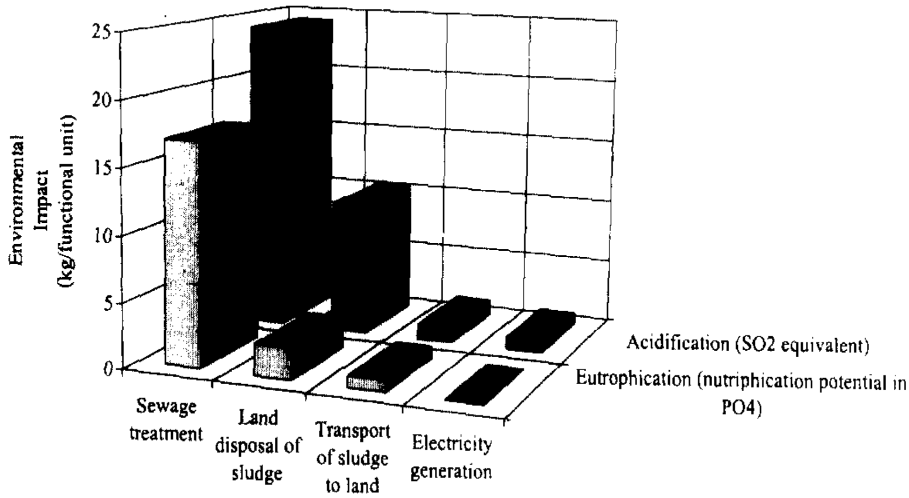

Figure 10.6 Environmental impacts arising from Burstow wastewater treatment works under the current management regime.

Colgate, a small works (Table 10.1), differs from Burstow, with the burdens from arising from sludge disposal making a greater contribution to acidification and eurtophication than the wastewater treatment process itself (Figure 10.7). Compared to Burstow treatment at Colgate is inefficient (Table 10.2), this is indicated by the greater mass of sludge disposed per functional unit of raw wastewater treated (Dennison & Azapagic, 1998).

Table 10.2 The mass of sludge disposed per functional unit of wastewater treated.

Wastewater Treatment Works	Burstow	Colgate
Annual raw wastewater treated (kg)	$1,000 * 10^6$	$14 * 10^6$
Annual mass of sludge disposed of (kg)	$4.6 * 10^6$	$0.2 * 10^6$
Mass of sludge disposed per functional unit (kg)	$4.0 * 10^6$	$15 * 10^6$

These results suggest that the burdens associated with sludge disposal are significant, especially when a significant mass of sludge is disposed from small works such as Colgate.

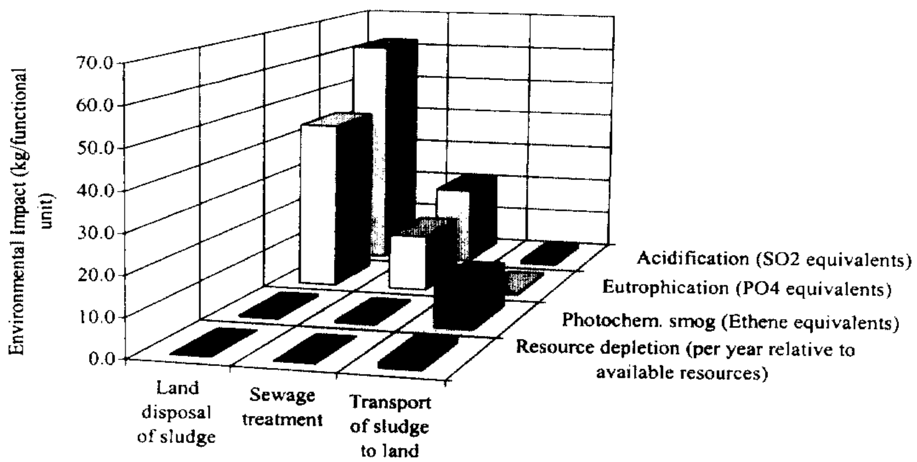

Figure 10.7 Environmental impacts Colgate from Burstow wastewater treatment works under the current management regime.

Assessing the Proposed Management Regimes

The results show that the proposed management regimes compared to the current regime significantly reduces the global warming potential (Figure 10.8). The addition of a polyelectrolyte in Options 2 to 5 significantly reduces volume of sludge requiring disposal as the percentage dry solids increases by a factor of five from %4 to %22 (Dennison & Azapagic, 1998).

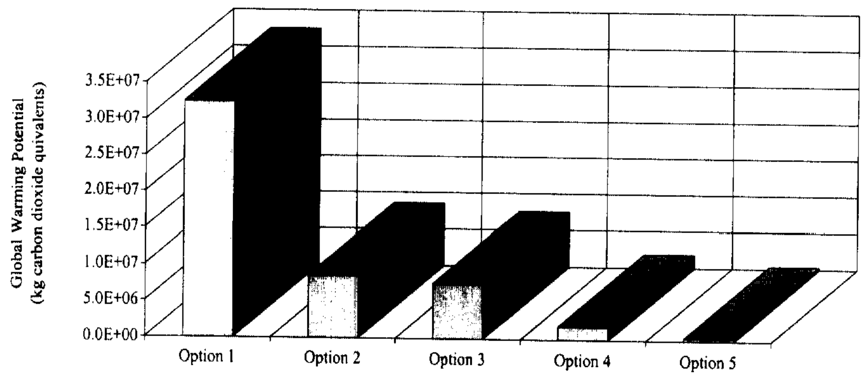

Figure 10.8 Total global warming potential of the current and proposed management regimes.

The results clearly show that complete centralisation of sludge for further de-watering at Crawley, prior to disposal, provides the greatest reduction in global warming potential (comparing Options 2 and 3, with 4 and 5). The results also indicate that by adopting composting (Options 3 and 5), as opposed to increasing the digestion facility (Options 2 and 4) at Crawley, a further reduction of global warming potential is possible. This may be explained by the significant burdens arising from sludge disposal via sub-soil injection (Figure 10.9). Figure 10.9 shows the contribution to global warming potential made by the individual life cycle stages within the composting and digestion systems, represented by Options 3 and 2 respectively. Figure 10.9 also indicates that environmental improvements may be made to the composting system if an alternative packaging material for compost distribution were to be chosen (Dennison & Azapagic, 1998).

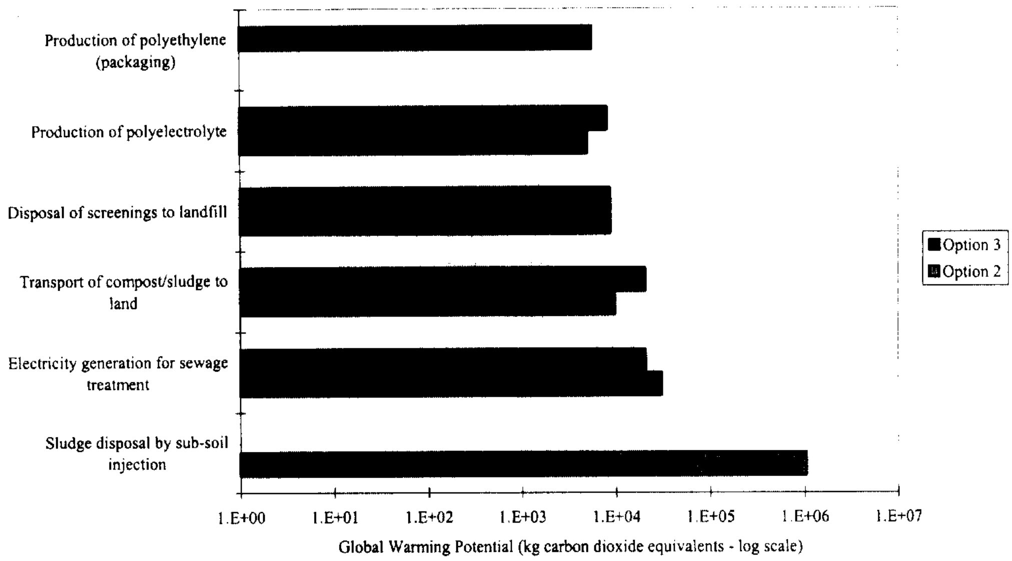

Figure 10.9 significant life cycle stages that contribute to the cumulative global warming potential of options 2 and 3.

The results shows that the environmental impacts with sludge disposal may be reduced by the dewatering the sludge at the wastewater treatment plants. It is clear that, Option 5 has the lowest global warming potential. The proposed management regimes have represents a significant environmental improvement upon the current management regime. On the other hand, the adoption of composting as opposed to increasing the digestion facility at Crawley has a lower environmental impact. These results demonstrate that centralisation of sludge for treatment and disposal is an environmental improvement upon the current practice (Dennison & Azapagic, 1998).

10.2 Example 2 - Antrophic Water Cycle and Life Cycle Assessment Methodology

This study was done to determine the environmental impacts arising from water production, water transposrt to the customer and wastewater treatment. This particular water cycle is called "anthropic water cycle".

Sewer construction is a significant element and has a great influence on the final result. A number of pollutants into water could not be integrated to the final result due to lack of indicators in EI99 and so, some work must be still be done to have complete results. Further steps of the study will include water production, water transport to the customer and sludge management (Renzoni & Germain, n.d).

Goal of The Study

The goal of the study comparing the environmental impact of wastewater treatment realised in a single centralised plant or in several smaller plants

Function and Functional Unit

In this study the function is wastewater treatment of a community of 11,0000 inhabitants. The functional unit is 1 cubic meter of water.

Wastewater Treatment Plants

Centralised Wastewater Treatment Plants

This centralised wastewater treatment plants is designed to receive wastewater of a population corresponding to 11,000 inhabitants. The life cycle of this plant includes construction and operation of the plant but also construction of the whole sewer network. This is an important parameter of the study. If the area covered by the wastewater treatment plant is large, the total length of the sewer network will increase considerably. Electricity and chemicals consumed during operating of the wastewater treatment plant are taken into account. It is assumed that the wastewater treatment plant and the sewer are made of reinforce concrete (Renzoni & Germain, n.d).

Decentralised Wastewater Treatment Plants

Several smaller decentralised wastewater treatment plants are placed. So 11,000 inhabitants have their wastewater cleaned but the total length of the sewer network is less important compared to the centralised wastewater treatment plants. The system boundaries include construction and operation of the decentralised plants. These types of small plants do not use any chemicals during operation and therefore, only electricity consumption is taken into account for the plants operating step. Figure 10.10 shows the boundaries o the regarded systems (Renzoni & Germain, n.d).

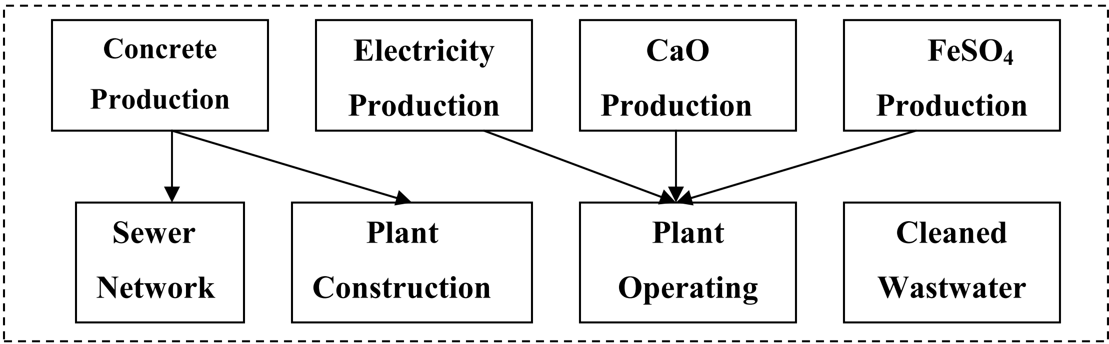

Figure 10.10 System boundaries.

Inventory Analysis

The study is done with Eco-Indicator 99 on the basis of an inventory of a great number of pollutants. Main results of Inventory obtained for major pollutants are shown in Figures 10.11, 10.12 10.13, 10.14, 10.15 and 10.16. These results are very characteristic of the study (Renzoni & Germain, n.d).

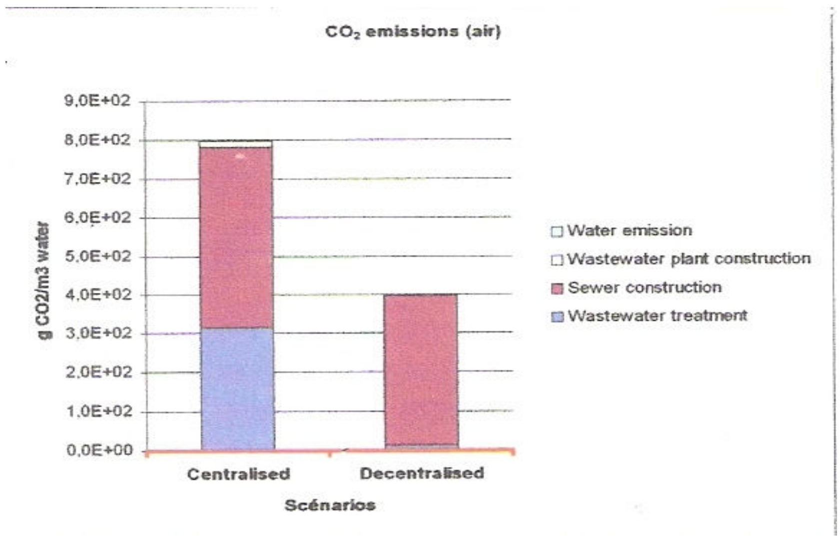

Figure 10.11 CO_2 emissions inventory.

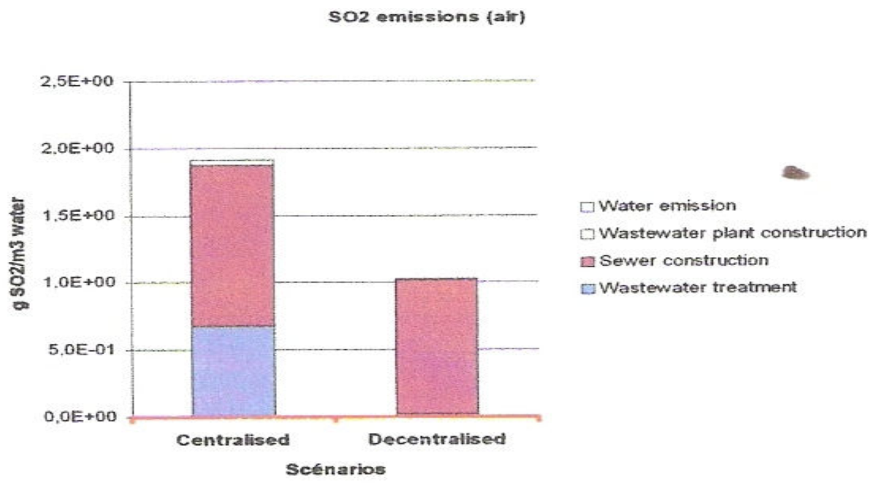

Figure 10.12 SO_2 emissions inventory.

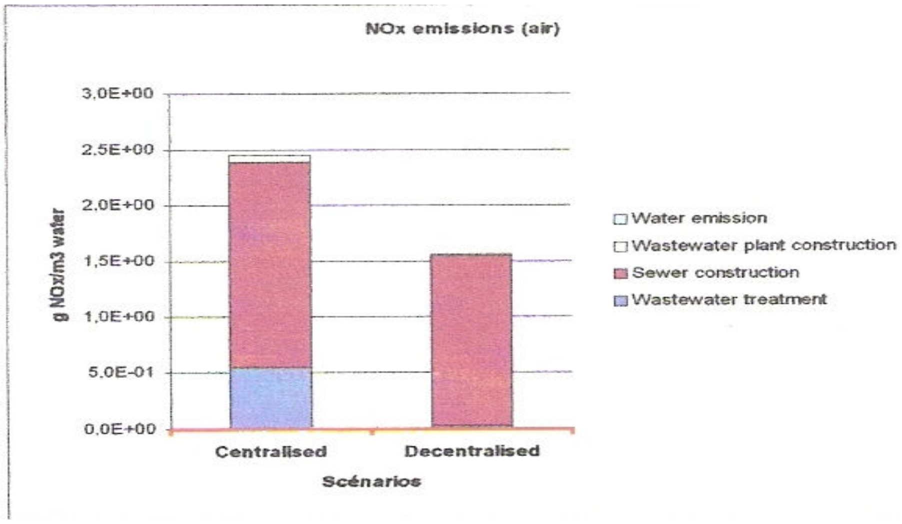

Figure 10.13 NO_X emissions inventory.

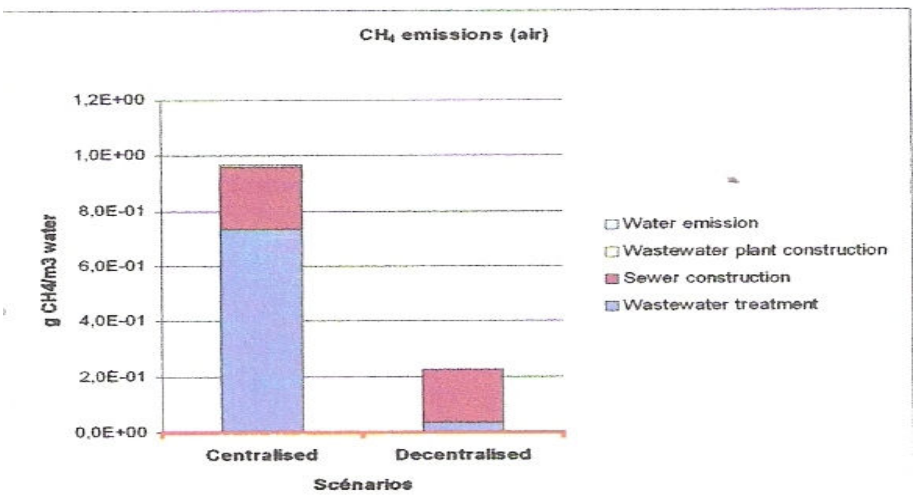

Figure 10.14 CH_4 emissions inventory.

On Figures 10.11, 10.12, 10.13 and 10.14 shown that, sewer constructions represent a great part of major atmospheric pollutants. For several decentralised wastewater treatment plants, sewer network is less extensive and therefore pollutants emissions are lower for this step. Production of chemicals in centralised wastewater treatment plants causes atmospheric emissions. Wastewater treatment plant constructions do not represent significant emission of major atmospheric pollutants (Renzoni & Germain, n.d).

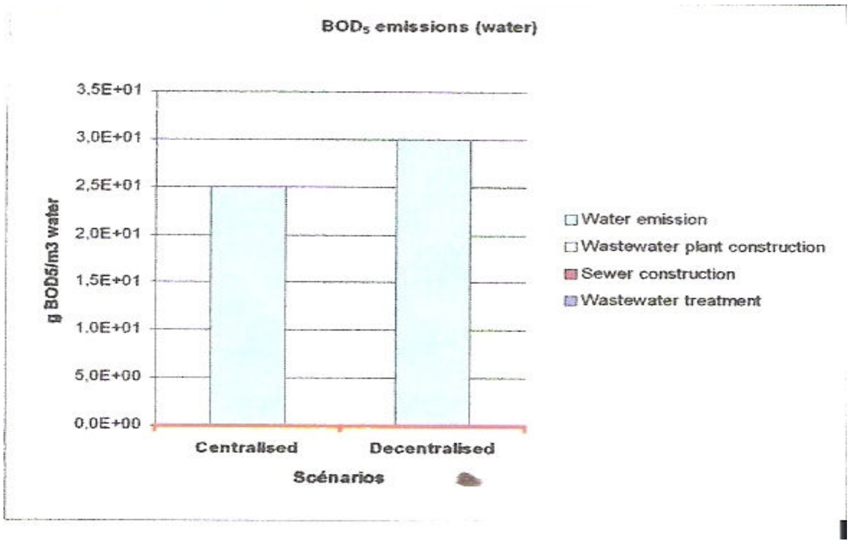

Figure 10.15 BOD_5 emissions inventory.

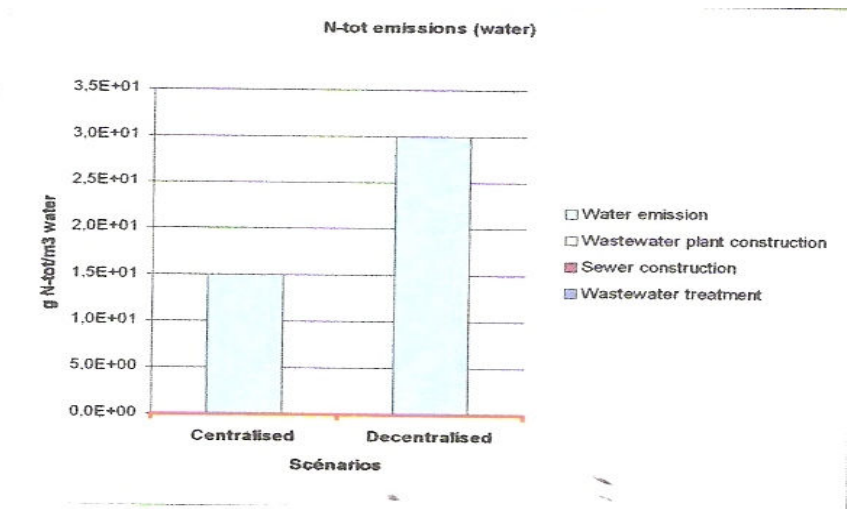

Figure 10.16 N-tot emissions inventory.

Figure 10.15 and 10.16 show the different inventory of emissions of pollutants into water. Wastewater treatment plant operating installations (sewer and plant) constructing phases do not represent important emissions into water after wastewater treatment, main emissions into water caused by cleaned water. For this particular case, a single centralised plant has a better cleaning efficiency than several smaller parts. Unfortunately, a number of pollutants into water like BOD, COD and N-tot are actually not taken into account by EI99 (Renzoni & Germain, n.d).

In my opinion Figure 10.17 shows the global eco-score resulting of the comparison of centralised and decentralised wastewater treatment plants. From these results, it is easy to say that several decentralised wastewater treatment plants are a best solution from an environmental viewpoint. However it has been noticed in the inventory that a number of pollutants could not be included in the final result.

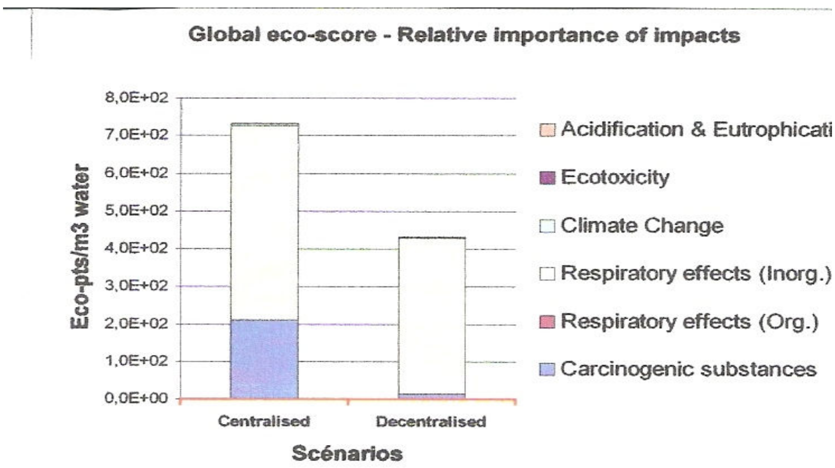

Figure 10.17 Global eco-score.

The study permits us to point:

- Sewer network is an important parameter.
- A lack of indicators for some pollutants into water.
- Interest of decentralised wastewater treatment plants.

10.3 Example 3 - Application Different Life Cycle Impact Assessment Methods to a Wastewater Treatment Plant

Introduction

The first aim of this example is to assess the environmental impacts of a wastewater treatment plant by using the Life Cycle Assessment (LCA) methodology. By identifying the sources of these impacts we will be able to propose solutions to improve the environmental performances of the plant.

The study was conducted by using three different Life Cycle Impact Assessment methods: Eco-Indicator 99, CML and Impact 2002+. It allowed us to highlight the similarities and differences between these methods. In this context, a wastewater treatment plant is a particularly interesting subject. Indeed, one of the main limits of the available Life Cycle Impact Assessment methods is the evaluation of the impacts at endpoint of eutrophying substances on aquatic ecosystems. The impact category "eutrophication" is one of the most important impact categories when studying a wastewater treatment plant (Halleux et al., n.d.).

Goal and Scope Definition Step

The studied system's treatment capacity is 170,000 inhabitant equivalents. The wastewater treatment includes:

- a preliminary treatment (elimination of trashes, sand, oils and greases),
- a primary treatment (floatation and decantation),
- a secondary treatment (biological treatment with activated sludge),
- a tertiary treatment (precipitation of phosphorus by adding iron chloride).

The performances of the plant are given in Table 10.3. The boundaries of the system include the building (production of the building materials: cement and steel) and the working of the wastewater treatment plant (primary, secondary, tertiary and sludge treatments), the effluents of the plant, the transport of the wastes to their final destination and the incineration of the sludge. The functional unit is the cubic meter of treated water (Halleux et al., n.d.).

Table 10.3 Performances of the plant.

	Concentration in wastewater (mg/l)	Concentration in effluent (mg/l)
BOD_5	300	<25
Nitrogen	55	<10
Phosphorus	22	<1

Impact Assessment Step

- Eco-Indictor 99

The Eco-Indicator 99 method doesn't enable us to evaluate the impacts of eutrophying substances (phosphorus, nitrogen) on aquatic ecosystems. The elimination of these substances is one of the aims of a wastewater treatment plant. That's why it is necessary to calculate impact factors for these substances if we want to study the plant with the Eco-Indicator 99 method. We have made the following assumptions (Halleux et al., n.d.):

The damage coefficient for eutrophication in Eco- Indicator 99 is proportional to the « midpoint » coefficient in CML.

The eutrophication is limited by the supply in phosphorus. Thus we have neglected the impacts of nitrogen and nitrate on aquatic eutrophication. Factors have although been calculated for the situations where nitrogen is the limiting substance.

The damage factors extrapolated from the NO_X eutrophication factor are given in Table 10.4 (Halleux et al., n.d.).

Table 10.4 Damage factors for eutrophication.

Substance	**Midpoint factor for CML (kg eq PO4/kg)**	**Damage factor calculated for Eco-Indicator 99 (PDF*m²*yr/kg)**
NO_X (reference)	0.13	9.52
COD	0.022	1.61
N-Kjeldahl	0.42	30.76
Nitrate	0.1	7.32
Phosphate	1	73.23

The results obtained after normalization step using these factors are shown on Figure 10.18.

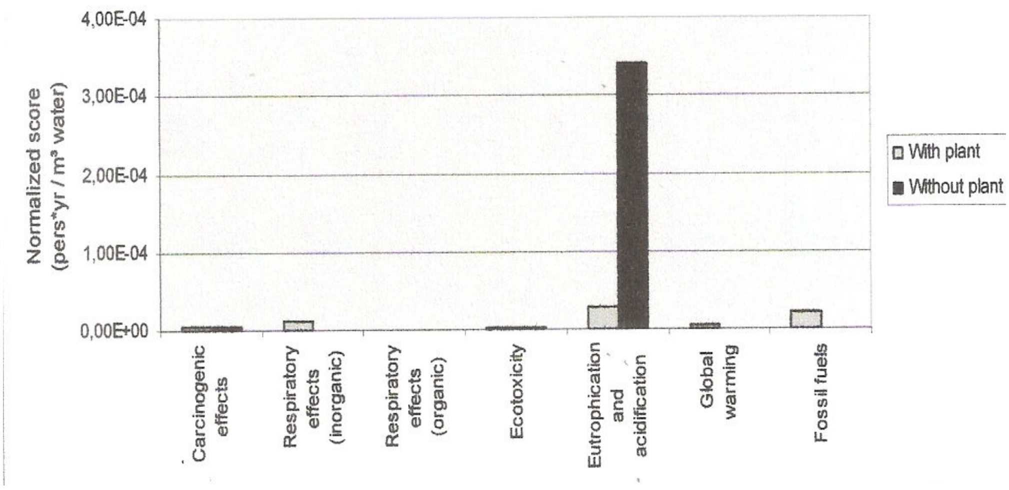

Figure 10.18 Environmental impacts after normalization step (Eco-Indicator 99).

On Figure 10.18, we understand that the eutrophication category dominates all the other impact categories. The eutrophication impact is much more important without wastewater treatment plant. As expected, the results of the analysis of the system with the Eco-Indicator 99 method are favorable to the building of a plant. This is confirmed by the results on Figure 10.19 which shows the global environmental impact after weighting (Halleux et al., n.d.).

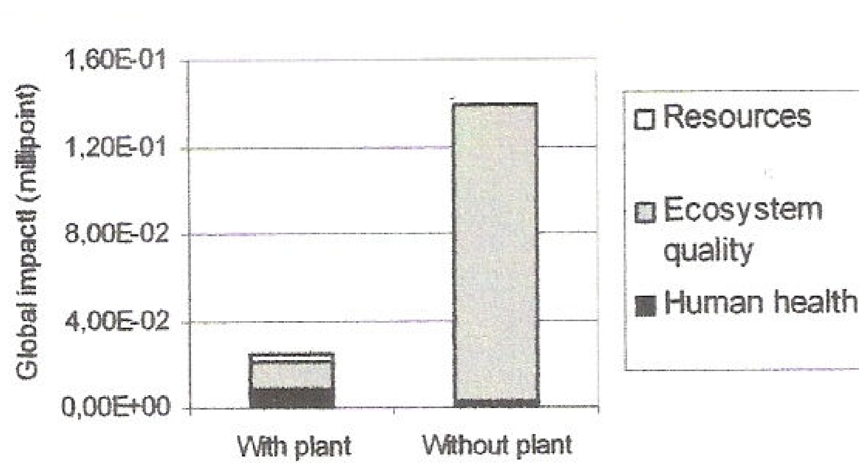

Figure 10.19 Global environmental impacts (Eco-Indicator 99).

- CML

The CML method can be applied to this system without adaptation. The results after the normalization step are shown on Figure 10.20.

According to CML, eutrophication and also aquatic ecotoxicity are the most important impact categories. In these two categories, the impacts are more important in the scenario without wastewater treatment plant. The method doesn't allow us to obtain a single weighted score, but we can easily conclude with Figure 10.20 that the environmental impacts are more important without the plant, and that the analysis is favorable to the building of a plant (Halleux et al., n.d.).

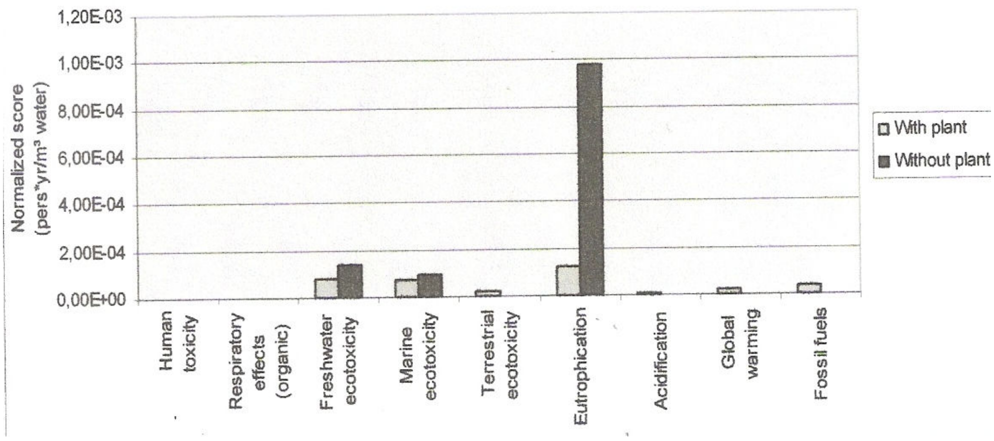

Figure 10.20 Environmental impacts after normalization step (CML).

- Impact 2002+

In this method we will only present the results obtained with the midpoint analysis. The results after the normalization step are shown on Figure 10.21. The results are similar to those obtained with the CML method and the analysis leads to the same conclusions. The only difference is that in addition to eutrophication and ecotoxicity, we can see that the non carcinogenic effects category has an important normalized impact. However this induces low damage. The characterization damage scores for human health are given in Table 10.5 (Halleux et al., n.d.).

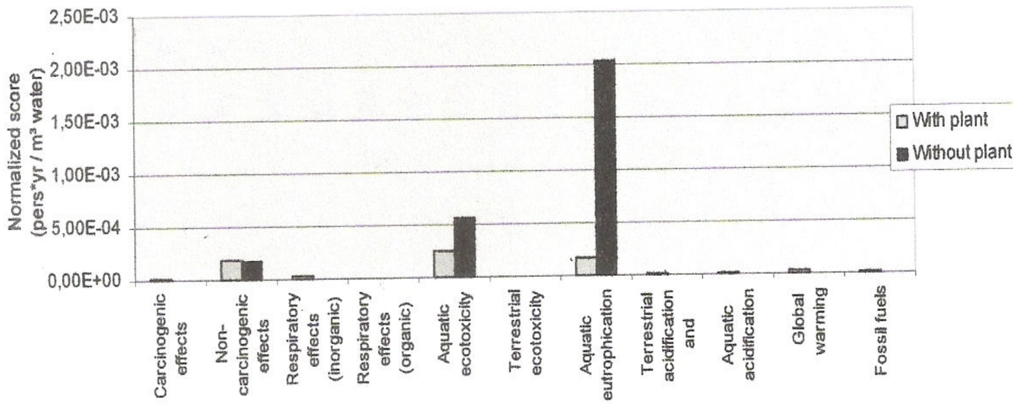

Figure 10.21 Environmental impacts after normalization step (Impact 2002+).

Table 10.5 Characterization scores (Impact 2002+).

Category	"Midpoint" score with plant	"Midpoint" score Without plant	Unit	"Damage" score with plant	"Damage" score without plant	Unit
Carcinogenic effects	$9{,}47*10^{-4}$	$6{,}97*10^{-20}$	kg eq chloroethylene	$1{,}37*10^{-9}$	$1{,}01*10^{-25}$	DALY
Non-carcinogenic effects	$3{,}17*10^{-2}$	$2{,}94*10^{-2}$	kg eq chloroethylene	$4{,}59*10^{-8}$	$4{,}25*10^{-8}$	DALY
Respiratory effects (inorganic)	$2{,}36*10^{-4}$	0,00	kg eq dust 2,5	$1{,}65*10^{-7}$	0,00	DALY
Respiratory effects (organic)	$3{,}95*10^{-5}$	0,00	kg eq ethylene	$8{,}40*10^{-11}$	0.00	DALY

Comparison of the Methods

All three methods are favorable to the building of a wastewater treatment plant. If the wastewater was thrown out in the river without treatment, the damages to aquatic ecosystems would be too high.

- Comparison of the Human Health Category

According to the Eco-Indicator 99 and Impact 2002+ methods, the large majority of the impacts on human health were due to emissions of NO_X and particulates (Respiratory effects inorganic). However, in the case of the CML method, emissions of these substances have only a marginal effect in comparison to that of heavy metals. Moreover, in the CML results, after normalization, human toxicity is the less important impact category, which is not the case for the two other methods. For the quantification of environmental impacts like photooxydant formation or global warming (which are considered as damages on human health in the Eco-Indicator 99 model), no important difference appears between the three methods (Halleux et al., n.d.).

- Comparison of the Ecosystem Quality

According to all three methods, eutrophication category and also the fact that this impact is considerably reduced by the wastewater treatment plant. CML considers simultaneously the impacts of phosphorus and nitrogen on aquatic ecosystems while Impact 2002+ only considers the impacts of a limiting substance. This difference doesn't appear on the figures, because the normalization factor is higher for CML as this method takes more substances into account. In the Eco-Indicator 99 method, aquatic eutrophication wasn't considered, and an adaptation of the model was necessary (Halleux et al., n.d.).

Moreover, All three methods shows that ecotoxicity, due to the presence of heavy metals in the wastewater, is noticeably reduced by the wastewater treatment plant, but the results differ from a method to another. First, on Figure 10.18 (Eco-Indicator 99), ecotoxicity is one of the less important impact categories while it is one of the most important according to CML and Impact 2002+. These two methods seem to have the same results if we only look at the normalized graph, but the pollutants contributing to the impact differ from one method to the other (according to CML, Ni causes the biggest impact while it is Zn according to Impact 2002+) (Halleux et al., n.d.).

- Comparison of the Resources

For the resources impact category, only fossil fuels consumption was taken into account, according to all methods. The only difference between the models concerns the coefficient attributed to hard coal. Some methods, like Impact 2002+, consider an infinite time horizon, and give impact factors to fuels equal to their heating value (in this context, hard coal has a coefficient near to that of crude oil). Other methods, like Eco-Indicator 99, attribute factors function of the heating value and the importance of the available reserves (Halleux et al., n.d.).

Interpretation Step

In order to improve the environmental performances of the plant, it is important to highlight the main contributors to the environmental load. This result is presented on Figure 10.22 for Eco-Indicator 99. This figure shows the importance of the contributions to the global environmental impact of the production of the electricity and the products consumed by the plant, of the production of the building materials (concrete, steel), of the transportation and the incineration of the sludge and finally of the pollutants still present in the treated water which is rejected in the river (Halleux et al., n.d.).

The majority (%51) of the global environmental impact is due to the treated water that still contains heavy metals and eutrophying substances; even if their concentrations and impacts are a lot lower than it would be without treatment. In consequence, to improve the performances of the plant, it would be necessary to reduce even more the concentration of these substances. The impacts of a more important consumption of products and energy would be compensated by a reduction of the impacts on aquatic ecosystems of the treated water (Halleux et al., n.d.).

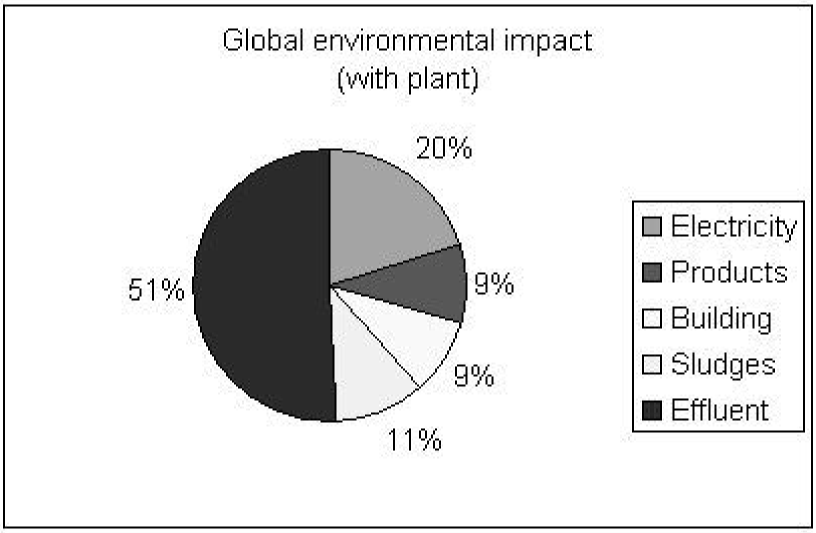

Figure 10.22 Global environmental impacts (Eco-Indicator 99).

All the experiments in this example have shown that the building of a wastewater treatment plant was necessary and that an improvement of the performances of the plant would be done by reducing even more the concentrations in pollutants like heavy metals and eutrophying substances in the effluent of the wastewater treatment plant. LCA is an essential tool for environmental analysis to avoid displacing pollution. But when the inventory becomes more complex (heavy metals, eutrophying substances) the limits of the impact assessment methods appear. A convergence in the models of the different methods for carcinogenic effects, ecotoxicity or eutrophication would be necessary (Halleux et al., n.d.).

CHAPTER ELEVEN
CASE STUDIES

11.1 Application of LCA to Two Kind of Wastewater that Come from Cartoon Package Factories

Wastewater treatment is one of the priorities of the environmental policy. Numerous plants have been built everywhere in the World and plenty of others are expected in the next few years. That is why it is necessary to assess and then reduce the environmental impacts of these plants.

In this study, we have analyzed two kinds of factory where cartoon package is produced. In the analysis, it is assumed that all factory compose the same amount of wastewater and LCA has done with attend to treatment processes, the chemicals that was used during the treatment process, the amount of sludge, investment and operation costs. Defining boundaries the system is the first aim in the Life Cycle Assessment methodology.

Knowing the processes in the production of cartoon package are important parameters for life cycle assessment to understand the sources of the pollutants. For this purpose, the phases of the cartoon packaging production are shown in Figure 11.1.

Wastewater results from washing the moulds and shaft of the machines during the operation. After analyzing the raw wastewater, chemical oxygen demand (COD), suspended solids (SS), oil-grease, Sulfate, Sulphide, Zink and pH are established as pollutants parameters. Next, we will choose the most suitable treatment system according to the discharge criteria related to the geographical area where the factory places.

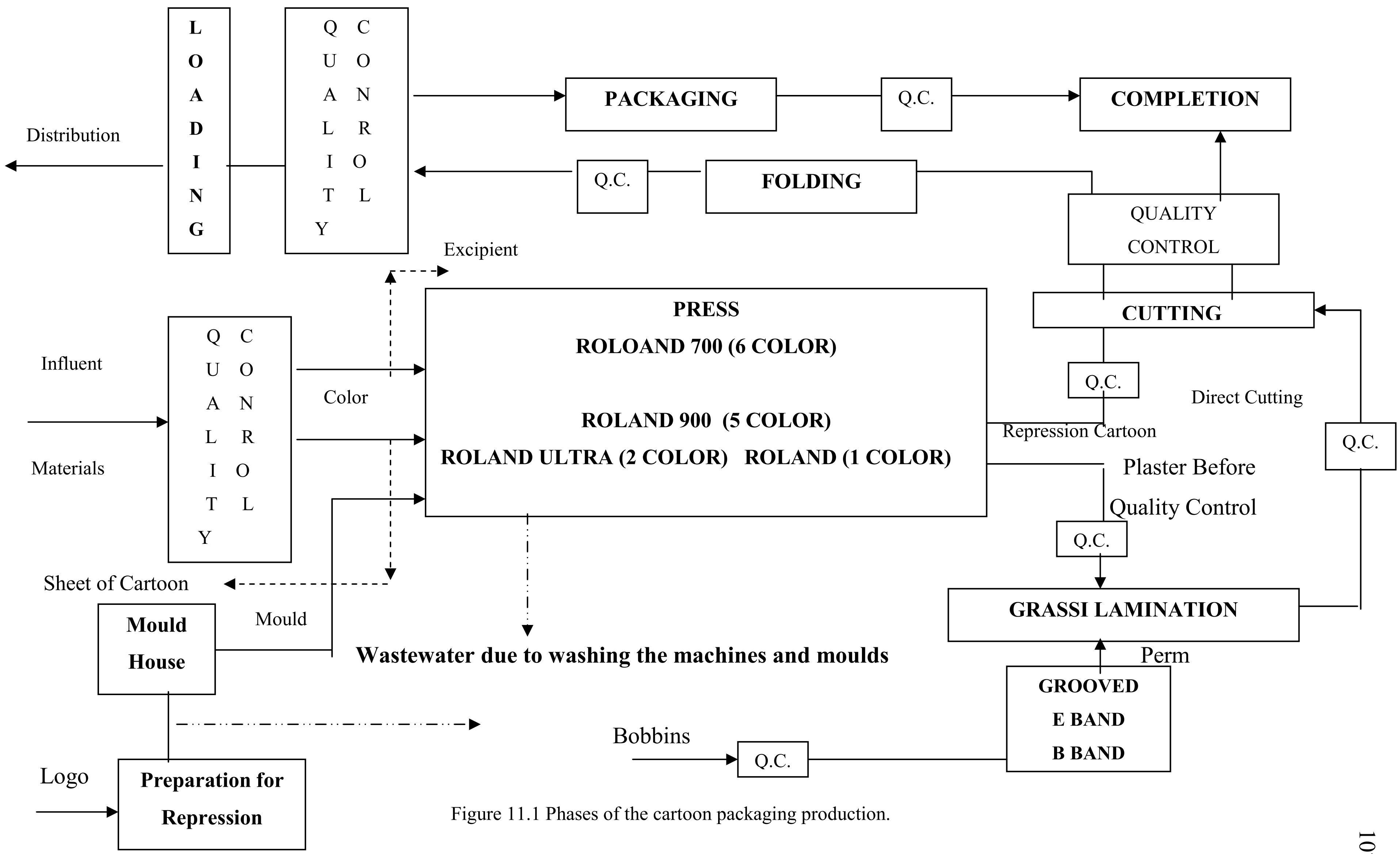

Figure 11.1 Phases of the cartoon packaging production.

Goal and Scope Definition

Goal of the Study

The aim of the study is doing an analysis for treatment methods of wastewater that comes from production of cartoon package

Functional Unit

The functional unit is the cubic meter of treated water.

Wastewater Treatment Alternatives

In this study two kind of wastewater treatment systems have analysed. The raw materials and chemicals that have used in the production are reference parameters to understand the sources of the pollutants in the wastewater.

The raw materials and chemicals that have used in the production gives in the Table 11.1.

Table 11.1 The raw materials and chemicals have used in the production.

Name	Unit	Amount
Cartoon	kg/year	5007,487
Paper	kg/year	3004,089
Ink	kg/year	15,959
Glue	kg/year	300,151

8100 kg cartoon package produce in a year in the factory and 0.5 m^3 wastewater is formed in a day after this operation which gives in the Figure 11.1. Value of the pollutants in raw wastewater and the effluent wastewater value are shown in the Table 11.2.

Either of the wastewater treatment system have same treatment efficiency. Just the same, chemicals that have used, selected equipments, amount of growing sludge and investment and management costs show difference between two treatment systems. As things stand, these parameters have differences shown in the Life Cycle Assessment methodology.

Table 11.2 Influent and effluent wastewater characteristics.

Influent Wastewater Characteristic		**Effluent Wastewater Characteristic**	
Parameter	**Value**	**Parameter**	**Value**
COD	2000	COD	1800
SS	1000	SS	400
Oil-Grease	350	Oil-Grease	140
Sulphate	100	Sulphate	100
Sulfide	<2	Sulfide	2
Zink	1	Zink	0,5
pH	4	pH	7

**All parameters' units are mg/l except pH.

Selected units to remove COD, SS, oil-grease, sulfate, sulphide and zink shows in the Table 11.3.

Table 11.3 Various wastewater treatment alternatives.

Alternative 1	Wastewater collection pond Grid Neutralization tank Sedimentation tank Sludge drying tank Discharge tank
Alternative 2	Raising manhole Grid Reaction and filter tank Sludge drying tank Discharge tank

Life Cycle Consideration Factors

The alternatives mentioned above, compared with each other in terms of life cycle assessment, chemical and energy usage, amount of sludge produced, investment and management costs.

For the above treatment alternatives, the weightage for each factor has been given four scales viz., no, low, medium and high impact and the same are shown in Table 11.4.

Table 11.4 Life Cycle considerations factors on various wastewater treatment alternatives.

Life cycle impact and other factors	**Wastewater Treatment Alternatives**	
	Alternative 1	**Alternative 2**
Chemical Usage	Low	Medium
Energy Usage	Medium	High
Sludge Production	Medium	High
Land Requirement	Medium	Law
Investment Cost	Law	Medium
Management Cost	Medium	Medium
Chemical Hazard/Risk	Law	No

For alternative 1 = 0 high, 4 medium, 2 law, 1 no;

Total impact value = 0x**3** + 4x**2** + 2x**1** + 1x **0** = 10

For alternative 2 = 2 high, 3 medium, 1 law, 1 no;

Total impact value = 2x**3** + 3x**2** + 1x**1** + 1x **0** = 13

Selection of the Best Wastewater Treatment Alternative

For selection of the best wastewater treatment alternative, the alternatives are to chosen based on the characteristics of the influent wastewater and requirement of quality of the treated wastewater to be disposed.

The effective alternative is the one with lowest total impact value. According to life cycle approach the alternative 1 is better alternative than alternative 2 to treatment of the cartoon package production wastewater.

In this example, a simple methodology has been developed for selection of wastewater treatment alternative incorporating life cycle impact and other factors.

11.2 Evaluation of LCA of Treatment Alternatives of Urban and Industrial Wastewater

In this study, for control of water pollution due to domestic and industrial wastewater, the conventional wastewater treatment using activated sludge process alone or in combination with chemical coagulation has been adopted. The treatment alternative selection in wastewater treatment plants is based on treatment requirement, land usability and capital costs. Rather, it should be based considering all the criteria over the life of the wastewater treatment plant including energy and chemical consumptions and overall environmental impacts. This sample study was done for selection of the best wastewater treatment alternative for industrial wastewater and domestic wastewater which produced by workers in the factories.

For this example; we assumed that different proven technologies for domestic and industrial wastewater treatment were considered. The treatment system includes primary treatment (with and without chemical addition), secondary treatment using aerobic and anaerobic process and tertiary treatment. Different possible treatment alternaatives for wastewater treatment plant are given below in Table 11.5.

Table 11.5 Various wastewater treatment alternatives.

Alternative 1	Physio-Chemical Treatment (PCT)+ Activated Sludge Process (ASP) + Chlorination
Alternative 2	Physio-Chemical Treatment (PCT)+ Activated Sludge Process (ASP) + Waste Stabilization Pond (WSP)
Alternative 3	Pre-settler (PS)+ Upflow Anaerobic Sludge Blanket (UASB) Reactor + Activated Sludge Process (ASP) + Chlorination
Alternative 4	Pre-settler (PS)+ Upflow Anaerobic Sludge Blanket (UASB) Reactor + Activated Sludge Process (ASP) + Waste Stabilization Pond (WSP)
Alternative 5	Upflow Anaerobic Sludge Blanket (UASB) Reactor + Waste Stabilization Pond (WSP)
Alternative 6	Physio-Chemical Treatment (PCT)+ Anaerobic lagoon (AL) + Activated sludge process + Waste Stabilization Pond (WSP)

Application of Life Cycle Assessment

The above alternatives were compared considering life cycle impact and other factors like;

- chemical and energy consumption,
- quantity of sludge generation
- emission of green house gases
- capital cost (civil construction and mechanical installation)
- maintenance cost
- land requirement

For the above various treatment alternatives, the weightage for each factor has been given four scales viz., no, low, medium and high impact and the same are shown in Table 11.6.

Choosing the Best Wastewater Treatment Alternative

Total impact value is to be calculated by assigning values 0, 1, 2, 3 for no, low, medium and high impact factors respectively for choosing the best wastewater treatment alternative.

Table 11.6 Life Cycle considerations factors on various wastewater treatment alternatives.

Life cycle impact and other factors	**Wastewater Treatment Alternatives (1 to 6)**					
	PCT + ASP+ Chlorination 1	**PCT + ASP+ WSP 2**	**PS+ UASB +ASP+ Chlorination 3**	**PS+ UASB+ ASP + WSP 4**	**UASB + WSP 5**	**PCT + AL+ASP+ WSP 6**
Chemical requirement	High	Medium	Medium	No	No	Medium
Energy requirement	High	High	Medium	Medium	Low	High
Green house gas emissions	Medium	Medium	Medium	Medium	Low	High
Sludge generation	High	High	Medium	Medium	Low	High
Capital cost	Medium	Medium	High	High	Medium	Medium
Land requirement	Low	Medium	Low	Medium	Medium	High
Chemical Hazard/ Risk	High	No	High	No	No	No

The alternatives 1, 2, and 5 will be able to meet environmental performance requirements for sewage treatment. For the four alternatives, total impact value are to calculated.

- For Alternative 1, High -4; Medium -2; Low-1 and No: 0;
  Total impact value = 4x**3** + 2x**2** + 1x**1** + 1x **0** = 17

- For Alternative 2, High -2; Medium -4; Low-0 and No: 1
  Total impact value = 2x**3** + 4x**2** + 0x**1** + 1x **0** = 14
- For Alternative 5, High -0; Medium -2; Low-3 and No: 2
  Total impact value = 0x**3** + 2x**2** + 3x**1** + 2x **0** = 7

Alternative 5 with lowest total impact value 7 is the best alternative considering the life cycle approach.

The alternatives 2, 4 and 6 will be able to meet environmental performance requirements for industrial wastewater. In industrial wastewater, chlorination or waste stabilization ponds con not use to reduce the pathogens. For the three alternatives, total impact value are to be calculated based on the life cycle factors given in Table 11.6 without impact due to land requirement and chemical hazard.

- For Alternative 2, High -2; Medium -3; Low-0 and No: 0
  Total impact value = 2x**3** + 3x**2** + 0x**1** + 0x**0** = 12
- For Alternative 4, High -1; Medium -3; Low-0 and No:1
  Total impact value = 1x**3** + 3x**2** + 0x**1** + 1x**0** =9
- For Alternative 6, High -3; Medium -2; Low-0 and No: 0
  Total impact value = 3x**3** + 2x**2** + 0x**1** + 0x**0** = 13

Alternative 4 is the best alternative considering the life cycle approach for wastewater treatment plant.

CHAPTER TWELVE
RESULTS AND DISCUSSION

In this study, Life Cycle Assessment methodology which analyzes contaminants that attract attention after industrialization and environmental pollution concepts and how LCA methodology is utilized in the water treatment are emphasized. The Life Cycle Assessment which is discussed in the early sections of this thesis is a still developing technique. The success of this technique is dependent on its flexibility, feasibility, financial convenience and technique reliability. These values come into prominence depending on the capacity of the factory.

Fundamentally, Life Cycle Assessment which is comprised of two stages, which are inventory and impact assessment studies respectively; is one of the environmental management techniques such as risk analysis, evaluation of the success of the environmental management techniques, environment control and environmental impact assessment.

All the material and energy supply chain is taken into consideration in the life cycle assessment of the wastewater. Products which, are formed by matter's and energy's penetrating into the system, staying there and leaving it, are the emissions released into the air, water and soil. Those emissions simply burden on the environment and are refined during the process of wastewater treatment. The dismissal of the contaminants which, are released during the formation, transmission and treatment of the wastewater, is one of the causes of the environmental pollution. The environmental effects of those contaminants triggered global threats such as: global warming, acid rains, holes in the ozone layer and eutrophication.

In the last part of this thesis, how LCA is utilized in the process of treatment of domestic and environmental wastewaters is emphasized. With the result of those studies, the importance of building a wastewater treatment plants has arisen. In some studies analyzed in this part, the comparisons about systems' environmental suitability with the Life Cycle Assessment method during the process of wastewater treatment have been made with different effect evaluation methods. While these researches are made, primarily it is useful for identifying the limits of the system to know the wastewater sources emerging during production. At this point, utilizing the facilities, planned according to wastewater characterization, in terms of environment gains importance.

For some examples, a simple methodology has been developed for selection of wastewater treatment alternative incorporating life cycle impact and other factors; chemical and energy consumption, quantity of sludge generation, emission of green house gases, capital cost (civil construction and mechanical installation), maintenance cost, land requirement. This approach for selection of wastewater alternative can be further improved by giving weightage for each factor and also by adding secondary parameters depending upon the site specific requirements.

With this thesis study, it is recommended that it will be better to choose the most environmentally suitable method by using the methods like Life Cycle Assessment, as it is not enough to plan only waste water treatment facilities to protect ecological balance. In order to have the practice of Life Cycle Assessment and to lower the cost of the method, it is necessary for researchers and executors to meet for sharing thoughts and information. This will bring a permanent partnership between the government, the university and industry and its being national and international will be important.

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