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LCA – A CRITICAL REVIEW AND ESTIMATION OF ITS UNCERTAINTY IN THE COMPARATIVE EVALUATION OF PACKAGING SYSTEMS

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LCA – A CRITICAL REVIEW AND ESTIMATION OF ITS UNCERTAINTY IN THE COMPARATIVE EVALUATION OF PACKAGING SYSTEMS

By

Dario Martino

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ABSTRACT

LCA – A CRITICAL REVIEW AND ESTIMATION OF ITS UNCERTAINTY IN THE COMPARATIVE EVALUATION OF PACKAGING SYSTEMS

By

Dario Martino

Life cycle assessment (LCA), defined as a systematic approach to evaluate environmental burdens associated with a product, process or activity covering its whole life cycle (i.e. from raw material acquisition to disposal) has been increasingly used by firms and government agencies to try to estimate the environmental impact of packaging options. Among the many LCA studies published, a great number of them are comparative studies to identify the environmentally better option, but whose results end up often being challenged by the losing parties.

This dissertation elaborates that conflictive results are a sign of LCA limits, which in turn can be traced back to the uncertainties inherent to the LCA approach. In order to fully represent these ideas, this work offers a two-step approach. First a critical overview of the state of the LCA method, its disparate applications and impact on topics that are part of the packaging field is offered. Second, the effect of some types of uncertainty (i.e. inventory data and scenario) in the outcome of a packaging based LCA featuring three different packaging materials is analyzed for a hypothetical drink product using published average process values. The three packaging systems are based on PET bottles, PLA bottles and an aluminum cans, along with their whole set of distribution packaging, and transportation services and end of life alternatives which were evaluated with regards to

four environmental burdens: energy use, water use, global warming potential (GWP) and ozone depletion potential (ODP).

The stochastic simulation for the simultaneous assessment of uncertainty due to LCI data, through Monte Carlo simulation, and scenarios relevant to packaging, through a non-parametric procedure, has demonstrated that, in this particular comparison, LCI parameter uncertainty seems to have a dominant effect in the outcome of the LCA results considered. Furthermore, by including the aforementioned noise, or considering temporal or spatial information or allocation procedures, the overlapping of uncertainty intervals (at 95% confidence) indicates that, despite the differences in the primary raw materials, the systems under study can be similar to one another, at least with regard to the environmental indicators of this work.

The dissertation also demonstrates that when the knowledge of the systems under study is limited to average values of the different operations, or to conservative uncertainty estimations, a clear option is unlikely to surface. On the other hand, the dissertation shows that when more information is known about how, where and when the system operates (i.e. location and time of operations, type of technology, raw material quality) more conclusive results can be obtained within an impact indicator, but it also shows that this gain cannot be fully exploited unless a priority-based approach across indicators is used for performing the impact assessment study.

DEDICATION

I would like to thank Dr. Susan Selke, my major advisor, for her patience and endless support. Many times, throughout the duration of this research she encouraged me to challenge the framework of many ideas that I assumed for granted, and that, I wish to believe is one of the contributions of this research. I would also like to thank Dr. Satish Joshi for sharing with me his experiences in LCA research and Dr. Amar Mohanty for providing me an insight on the tendency of LCA use in the biodegradable material field. I want to express gratitude also to Dr. Bruce Harte for being a mentor figure not only during this research but also during my whole stay at Michigan State. Our dialogs will be among my most cherished experiences of Grads School.

It will be an injustice if I don't include in these paragraphs, Dr. Seungdo Kim from the Chemical Engineering Department who allowed me access to the DEAM inventory database modules that are used in this study. Gratitude also to Dr. Keoleian and David Spitzley from the Center for Sustainable Systems at University of Michigan, both of which allowed me to attend seminars in their department that helped me better understand the nature of LCA. Thanks also to Dr. Huijbregts and Dr. Ragas, from University of Nijmeden (The Netherlands), for their correspondence with advice in the use of Crystal Ball, and to Mary Curran (U.S. EPA) for her advice in evaluating the literature review chapter of this dissertation.

Last, but not least, I want to express gratitude from the heart to my beloved Sophia and Karina, who stood by my side throughout the fortunes of the past years.

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LIST OF ABREVIATIONS

GWP: Global warming potential

ISO: International Organization for Standardization

LCA: Life cycle assessment

LCC: Life cycle costing

LCI: Life cycle inventory

LCIA: Life cycle impact assessment

LDPE: Low density polyethylene

MC: Monte Carlo (simulation)

ODP: Ozone depletion potential

PDS: Product delivery system

PET: Polyethylene terephthalate

PLA: Polylactide

PP: Polypropylene

SETAC: Society of Environmental Toxicology and Chemistry

CHAPTER 1 OVERVIEW OF THE RESEARCH

I. Introduction

The final goal of this dissertation is to provide a thorough understanding of some of the reasons behind the complex (and many times conflictive) results of comparative packaging based LCA. In order to achieve this, two major issues need to be addressed. First, the scope and limits of the Life Cycle Assessment (LCA) method need to be known. This dissertation achieves this task by offering a critical overview of the state of the LCA method, its disparate applications and impact on topics that are part of the packaging field. Second, as will be seen in later sections of this dissertation, when LCA is used for comparison purposes – as it is commonly done in packaging – it often reaches its limits by giving unclear and conflictive results. Many times these limits can be traced back to the uncertainties inherent to the LCA approach. Thus, the second task of the dissertation is to analyze how uncertainty in both the LCA method and packaging operations plays a role in actual comparative packaging-based LCA results.

The remainder of this introduction will provide a roadmap to the subsequent sections of Chapter One specifically, and the dissertation in general. Chapter One has several sections that explain in more depth the motivation and the approach taken to perform this packaging-based LCA study. Section II, in particular, highlights the relevance of conducting this LCA research. Additionally, this section touches upon some recent LCA methodology publications that are reviewed in detail in Chapter Two. Section III enumerates the objectives of the research. Section IV states the hypothesis of the dissertation. Section V reviews the overall research methods, outlining the steps followed throughout the dissertation to achieve the stated objectives. Section VI discusses the scope of the dissertation, while Section VII discusses the contributions of this research to the field. Lastly, Section VIII details the structure for the rest of the chapters in the dissertation.

II. The need for packaging-based LCA research

Life cycle assessment (LCA) is considered as one of several strategies known in the industrial ecology field for environmentally conscious manufacturing, pollution prevention, and "ecoefficient" or "green" design that is increasingly being used by stakeholders and third-party organizations. Defined as a process to evaluate environmental burdens associated with a product, process or activity over its whole life cycle (i.e. from raw material acquisition to disposal), LCA has been praised as the approach to assess the environmental footprint of industrial operations.

Almost naturally, packaging situations were one of the earliest applications of LCA. About three decades ago, public concern over increasing volumes of solid waste due to the use of plastic in packaging and later concerns about energy consumption became the major driving forces to study the effects of packaging on the environment, and life-cycle studies were devised to help assess the environmental footprint of packaging use. Since then, a great number of packaging-based life cycle assessment studies have been published. Many of these LCAs were and are comparative studies.

However, due to the uncertainties involved in both the LCA methodology and the operations that are the subject of the assessment, there is not (and there might not be) a single, definite one-size-fits-all approach to performing an LCA. This is a major problem when LCA is used for comparison purposes because this means that any result can be contested. Nevertheless it is not uncommon to find comparative packaging LCA studies that portray, deliberately or not, one packaging material as environmentally better than other.

With this in mind, the motivation of this dissertation is to try to understand some of the reasons behind the conflicts in comparative packaging based LCA results. In order to achieve this, it is proposed to address two major issues.

First, there is the need to understand the scope of the current LCA approach in general and when applied to packaging, and its improvements and the arguments behind the limitations of the LCA method. That is, there is need to identify major advances developed by the LCA community as well as current issues of the LCA methodology. For instance, according to the current understanding of the International Organization for Standardization (ISO 14040, 1997) and the Society of Environmental Toxicology and Chemistry (SETAC) (Barnthouse et al., 1998), LCA comprises four stages which are: (1) goal and scope definition, in which the purpose, boundary conditions, functional unit and allocation rules are set; (2) life cycle inventory (LCI), which is the accounting of emissions, raw materials and energy used; (3) life cycle impact assessment (LCIA), which links the LCI with identifiable environmental threats; and (4) critical interpretation of results, which is made throughout the study.

However, due to the uneven pace at which it has been embraced around the world, several aspects of LCA remain unclear. In fact, though developed more than thirty years ago in the U.S., the European willingness to incorporate it as a part of their environmental regulatory process is often cited as the reason why they lead the way in LCA research

(Curran, 1999). Only since the 1990's with ISO's release of its 14000 series of standards on Environmental Management, have many developing countries started to learn about this concept. This delay in coordination (i.e. internationally and nationally) is one reason why a common terminology has been slow to be developed and life cycle-based terms and approaches such as life cycle management or life cycle costing generate confusion. Further, since many LCA studies still remain unpublished or inaccessible (e.g. due to language barriers or confidentiality agreements), the assimilation of common methodologies is even more difficult.

In recent years, though, the private sector in developed countries has appeared more willing to embrace product life cycle concepts at the product design phase in order to respond to consumer expectations. As multinational firms operate around the world, their understanding of LCA concepts has been extended. In turn, efforts on harmonizing private life-cycle initiatives have started to occur (Hunkeler et al., 2002; Hunkeler et al., 2001; Sonnemann et al., 2001). Private and governmental agencies of the U.S. and of European countries have started the development of frameworks and partnerships with industry to help with the goal of making LCA a more useful tool for decision-makers. Current efforts by environmental certification third-party organizations in these countries follow harmonization steps as well as reaching out to similar bodies in developing countries. Moreover, a survey of the LCA work in general clearly shows the growing idea that the life cycle concept has evolved from a tool to help compare products to an essential part of achieving broader goals such as sustainability.

It is thus of particular importance, and not only to help understand the reasons behind conflicts in comparative packaging based LCA results, that a critical review of the literature detailing the state of the LCA method and its implications in packaging be provided, since utilization of this tool in this field is not restricted to comparative analysis, but also comprises a broad spectrum that includes product development, product improvement, development of pollution prevention or waste management policies, and environmental labeling certifications.

Second, as it will be seen in later sections of this dissertation, when LCA is used to compare the "environmental performance" of packaging materials, it often reaches its limits by giving unclear and conflictive results. In this dissertation it is argued that many times these limits can be traced back to the uncertainties inherent to the LCA approach. Moreover, it is argued that even within the same comparative life cycle assessment study of the same packaging options, conflictive results can still be obtained and a supposed preferability of alternatives can change.

In fact, with regard to the LCA method, Huijbregts (1998a) elaborated that LCA methodology is the subject of different kinds of uncertainties and variability along its different stages and because of them its results can be seriously undermined. In industrial systems, uncertainty (i.e. the lack of sureness about something) and variability (i.e. inherent variation of measured values) are responsible for many of the problems in an LCA. Uncertainty is important because when not evaluated, there is more risk that the impact predicted by the LCA may not match the actual environmental impact. Likewise, variability is important because it limits the precision of LCA results.

ISO standards (ISO 14040, 1997; ISO 14042.2, 1998; ISO 14041, 1998; ISO 14043, 2000) recognize the limitations that arise from not addressing uncertainty and variability and recommend the practitioner to do so in the LCA study. Nevertheless, the standards do not suggest any method to do so. In fact, Ross et al (Ross et al., 2002) reported that out of 30 LCA studies only 10% included a quantitative or qualitative uncertainty analysis. Thus, he concluded that the standards needed to be revised to ensure that LCA included at least a qualitative discussion of uncertainty and variability. Among attempts to provide a systematic analysis, just in the last few years the first general framework for comprehensive uncertainty and variability evaluation with the definition of key concepts in the LCA context was proposed (Huijbregts, 1998a; 1998b; 2001; Huijbregts et al., 2003; Huijbregts et al., 2001). But uncertainty analysis is something new in LCA (Barnthouse et al., 1998) and thus a number of methodologies for uncertainty estimation have been proposed. Currently, the most favored approach for uncertainty and variability analysis in LCA is incorporating it into the life cycle inventory (LCI) stage. However, though often acknowledged to suffer from uncertainty, the information in databases required for the analysis (e.g. ranges, standard deviation, probability distributions) is still very limited. Thus engineering uncertainty estimations along with computer simulation are becoming the method of choice. A number of recent uncertainty and variability methodologies have been proposed and incorporated into LCAs associated with different industry sectors and almost exclusively used Monte Carlo simulation (Canter et al., 2002; Ciroth et al., 2004; Contadini and Moore, 2003; Guyonnet et al., 1999; Huijbregts et al., 2003; Maurice et al., 2000; McCleese and LaPuma, 2002; Sonnemann et al., 2003). Others used fuzzy logic to perform a similar data evaluation (Guyonnet et al., 1999; Tan et al., 2002). In some of these cases, packaging situations with very limited scope have been used only as case studies to introduce new and often intricate uncertainty assessment methodologies (World largest PET LCA, 2004; Canter et al., 2002; Kennedy et al., 1997; Kennedy et al., 1996).

In turn, uncertainties in many packaging operations can be understood as scenario uncertainties. For instance, differences in percent recycled content in a packaging material, packaging end-of-life practices (e.g. landfilling, incineration, recovering), and product distribution distances all may change the outcome of the LCA. Even within the same country, the same packaging end-of-life alternative can be viewed both as environmentally friendly and environmentally unsound. In fact, analyzing several comparative packaging based LCA results, it is not uncommon to find directly opposite results that are often the subject of heated debates among industry groups and even countries (World largest PET LCA, 2004; Hockerts et al., 1999).

Thus, in order to understand the nature behind the conflicts in comparative packaging based LCA results, it is of key importance to analyze how uncertainties in both the LCA method and packaging operations affect an actual outcome of a comparative packaging-based LCA.

III. Objectives

The primary objectives of this study are then:

i. To provide a critical review of the state of the environmental Life Cycle Assessment (LCA) method and its application to packaging.

ii. To analyze how uncertainties in both the LCA method and packaging operations play a role in actual comparative packaging-based LCA results.

To help accomplish the second primary objective, it was divided into the two following secondary objectives:

a. To develop a life cycle assessment (LCA) study to compare the environmental performance of hypothetical drink delivery systems (PDSs) featuring three different primary packaging materials.

b. To critically analyze the results of an uncertainty and variability analysis of the comparative packaging based LCA outcome. Specifically, uncertainty in packaging operations will be expressed by the analysis of the comparative LCA outcome of different scenarios relevant to packaging operations. The uncertainty in the LCA method will be expressed as inventory parameter uncertainty through stochastic simulation.

IV. Hypothesis

This dissertation investigated the hypothesis that when uncertainties inherent to the methodology and the nature of industrial processes are considered, the current life cycle assessment approach does not provide enough evidence to rule out (or favor) one alternative versus another when packaging systems are studied and compared based on their environmental performance. Indeed, the expression "environmental performance" by itself is very general and could encompass any set of environmental indicators. More specifically, it is argued that under the stated uncertainties within a same set of boundaries and same options of packaging materials, the LCA approach still cannot provide a clear more environmental option when packaging systems are compared.

V. Research methodology

To achieve the first primary objective, a critical review of the literature with regards to LCA and its applications to packaging operations was performed and is presented in Chapter Two. The main bodies of literature reviewed in this chapter involve actual LCA studies, recent journal publications regarding advances in LCA methodology and case studies, ISO 14000 series of standards, reports from governmental agencies and private sector literature.

The research methodology followed to achieve the second primary objective was the following. First, a comparative LCA study featuring three different packaging materials was developed for a hypothetical drink product. Following ISO's LCA steps, extensive details and information sources used for this comparative LCA are presented in Chapter Three. The three material/container alternatives involved in this study were a PET bottle, a PLA bottle and an aluminum can. In order to represent realistic implications of actual packaging operations, production and use of distribution packaging (i.e. corrugated trays, stretch wrap, and pallets), as well as transportation steps through several parts of the packaging life cycle were included. Relevant end-of-life scenarios such as landfilling, incineration and recycling were also taken into account in this analysis. Each of the main material alternatives (i.e. PET, PLA, aluminum) along with its whole set of operations was defined as a product delivery system (PDS) and environmental burdens associated with each of the three PDSs were included in the study. The actual calculation procedures used in this comparative LCA were performed using a series of electronic spreadsheets in Microsoft Excel (Microsoft Excel, 2000).

In Chapter Four the actual uncertainty analysis of the comparative LCA is performed. Specifically, the uncertainty analysis in this study comprises two aspects. First, a scenario analysis was performed identifying different domains that may have the potential to change the outcome of the comparative packaging based LCA. Under each domain two alternatives are proposed. Then, a procedure is used to select an alternative from each domain and randomly create different scenarios. The second aspect of the uncertainty analysis in this study comprises inventory parameter uncertainty. This analysis is achieved by the use of appropriate published uncertainty estimations along with stochastic simulation (i.e. Monte Carlo simulation) in order to represent inventory data uncertainty as probability distributions. By this analysis, relevant inventory parameters are identified and further analyzed, and final results of the comparative LCA along with the combined uncertainties from inventory data and scenarios are obtained. Finally in this chapter, the consequences of the comparative outcomes of the LCA are extensively discussed.

Last, Chapter Five is where conclusions, future work and recommendations are presented.

VI. Scope of the dissertation

This dissertation contains information regarding advances and limitations of the current LCA methodology and its uses and implications for packaging. Regarding the LCA study presented here, it comprises the stages of a packaging delivery system including material production, container manufacture, filling, and end-of-life operations. Transportation and use of distribution packaging to transport packaging components are also included. The life cycle model does not include the usage stage nor the transportation that is related to this stage because emissions and energy use data about these operations were not readily available. Furthermore, it was reasoned that the omission of these operations would not impact or change the characteristics of the LCA since its contribution was assumed negligible (e.g. when compared to the material production phase)

The impact assessment phase of the comparative LCA in this dissertation comprises impact categories such as energy and water use, ozone depletion potential (ODP), and global warming potential (GWP). In essence, the first two are actually inventory results for which information is often available in the literature and thus commonly tracked in LCAs and therefore their selection here. In turn, GWP and ODP are indeed environmental impact indicators which were selected since by being the only impact indicators with a global scope (i.e. not dependent on location) and whose characterization methodologies are among the most accepted in the LCA community, they offered what were expected to be the most robust environmental impact estimations.

The uncertainty and variability analysis includes the study of the outcomes of a comparative packaging based LCA result using different scenarios that are relevant to packaging operations. This study further uses Monte Carlo simulation in order to help with the parameter uncertainty analysis that comprises the life cycle inventory phase.

VII. Contributions of the research

Primarily, this dissertation is a conceptual analysis and pondering of ideas based on the packaging-based comparative LCA outcome. The contribution of this dissertation is in providing information and discussion to help understand some of the reasons behind the conflicts in comparative packaging based LCA results.

This research also satisfies other academic and practitioner interests in life cycle assessment practices used in the packaging field. For instance, information in the literature review of the dissertation can serve not only to help understand the current state of LCA, but also to screen for appropriate resources at the moment of deciding to do a packaging-based LCA. This section is also of value since it provides an overview of the LCA use associated with implementing programs with impact on packaging, such as product take-back initiatives as well as environmental labeling certifications.

With regard to the materials selected for the comparative LCA study, the consideration of a novel biobased material (i.e. PLA) as an alternative will also help understand some of the environmental implications of the use of a biobased polymer as a packaging system associated with a relevant packaging functional unit. As will become apparent in later chapters of this dissertation, the selection of this novel material also highlights a critical aspect of LCA, which is the limitation of the environmental assessment based on engineering estimates, diagrams, and theoretical studies instead of actual commercial production data.

Insights from the discussion of the effect of uncertainty and variability on the outcome of a packaging-based LCA outcome will also prove useful to motivate further analysis and extend it to other packaging systems and/or other scenarios with relevance in packaging operations.

VIII. Structure of the Dissertation

Chapter Two is a comprehensive review of the literature. The main bodies of literature involve LCA methodology, ISO 14000, publications by governmental agencies and private sector literature. Chapter Three introduces the details and data sources used for a comparative LCA study of drink delivery systems featuring three different packaging materials. Chapter Four presents an uncertainty analysis on scenarios and the inventory data and discusses the results and their implications. Chapter Five presents overall conclusions as well as recommendations and future work. Finally there are the appendices that contain data tables and statistical procedures used for uncertainty estimation.

CHAPTER 2 LITERATURE REVIEW

I. Introduction

Life cycle assessment (LCA) is defined as a cradle-to-grave analysis that can be used to move towards the ideal of sustainable development (Klöpffer, 2003). Sustainability, as adopted by the World Commission on Environment and Development (1987), describes the political goal for the future of mankind. Sustainable development implies an ideal balanced relationship between natural resources and human activity. LCA is intended to play an integral part in attaining this balance.

LCA is the result of the evolution of early "waste" and "energy analyses" performed by a few industries in the past. These waste-energy analyses were aimed at calculating the embodied energy and generation of solid wastes over different stages of the product life cycle. Later, due to their identical methodology, these were expanded to encompass the computation of total life cycle release of pollutants.

LCA is one of a number of environmental impact evaluation techniques and as such, it is recognized that it may not be appropriate for all situations. In fact, this tool presents limitations that are intrinsically related to the definition of the scope and interpretation of the systems that are being assessed, the modeling of emissions, fate and final impacts of substances released into the environment, and problems due to the data-intensive nature of the assessment. As a result, LCA is an evolving tool.

The purpose of this chapter is to provide an updated, structured review of LCA information for packaging-oriented LCA practitioners. To help present the information, this chapter has several sections. The first section describes the basic nature and characteristics of the LCA methodology, followed by a review of the limitations of LCA and recent enhancement approaches developed by the LCA community to address some of the limitations. Next, the use of LCA specifically in packaging applications is reviewed. A birds-eye description of related private and governmental initiatives with regard to LCA in packaging is included. The final section discusses the likely future of LCA applications in packaging.

II. Definition of LCA

In the last fifteen years, several definitions of LCA have been offered with minor variations from each other. Developed in 1991, the definition by the Society of Environmental Toxicology and Chemistry (SETAC)(Workshop report: A technical framework for life-cycle assessment, 1991), was among the first ones and describes LCA as:

"... an objective process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and material uses and releases to the environment, and to evaluate and implement opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing materials; manufacturing, transportation and distribution; use, reuse, maintenance; recycling and final disposal".

Since 1997, harmonization steps regarding a common LCA understanding have been made with the appearance of the International Organization for Standardization (ISO) LCA-related standards (ISO 14040, 1997; ISO 14042.2, 1998; ISO 14041, 1998; ISO 14024, 1999; ISO 14043, 2000) within the 14000 Series of Environmental

Management System (EMS) standards, and their subsequent rapid global adoption. ISO defines LCA as:

"a compilation and evaluation of inputs and outputs and the potential environmental impacts of a product system throughout its life cycle".

In either of these definitions, the life cycle of the product system, or cradle-to-grave, starts with the gathering of raw materials from the earth to create the product and finishes when they are returned back to the earth. LCA then considers the cumulative environmental impacts that occur due to all stages of the product life cycle.

Worth noticing is the fact that by considering the whole life cycle of a product, LCA has the potential for avoiding problem shifting. However, understandable or not, the LCA definition does not require that all possible environmental impacts be accounted for (i.e. a study that looks only at greenhouse gas emissions on a cradle-to-grave basis can be called an LCA), allowing room for burden shifting.

III. Product life cycle

LCA is based on the product "life cycle". Figure 1 shows a simplified diagram of a typical product life cycle which would start with the raw material acquisition (e.g. petroleum extraction and refinery for petroleum-based plastic products). After raw material acquisition, the cycle would include the material manufacture stage. Here, raw materials would be processed into basic manufactured materials (e.g. manufacture of plastic resin pellets). These materials would then be moved to the actual product manufacture stage where they would be made into products (e.g. plastic pellets extruded and molded into milk jugs). Eventually, they would be used (e.g. by milk distributor and consumer) and disposed. When disposed, they might go through waste management programs to be reused, and/or recycled, and/or incinerated, and/or sent to landfills. As shown in the diagram, all stages are interlinked, and along with the transport required to move products and materials, require energy and ancillary materials and produce wastes and emissions.

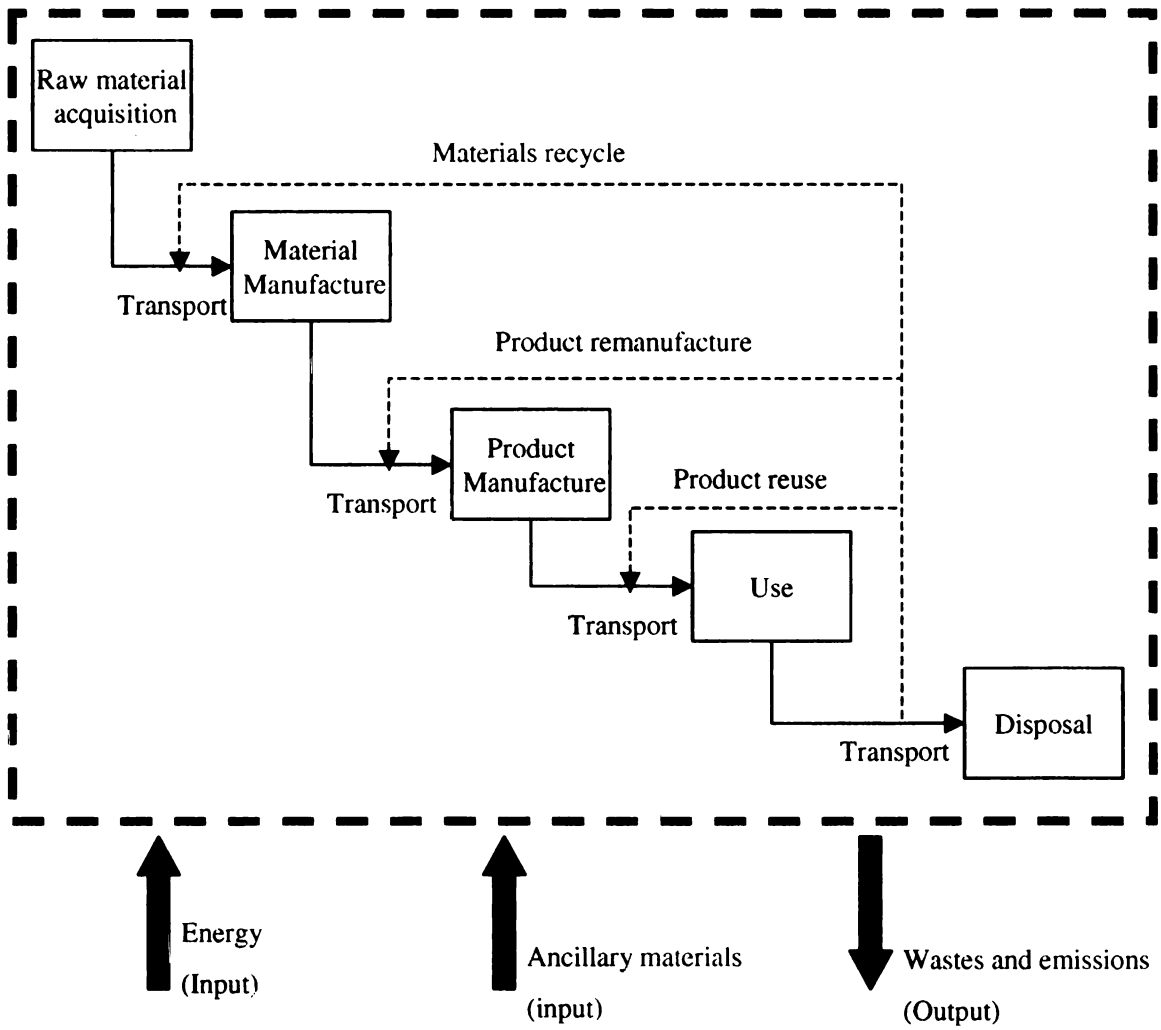


Figure 1. Typical product life cycle

IV. LCA Stages

LCA is an evolving technique. Currently, ISO and SETAC guidelines divide the LCA into four stages (ISO 14041, 1998) (see Figure 2).

Goal and scope definition: where the purpose of the study, its scope, functional unit and the procedure for quality assurance of the results are described. This step specifies the inputs and outputs selected for inventory and selects the functional unit, a common reference to which the inputs and outputs are related, associated with the function of the system under study.

Inventory analysis: which is the actual quantitative analysis of inputs of raw materials and fuels into a system and the outputs of solid, liquid and gaseous wastes from it. In the Life Cycle Inventory (LCI), data associated with the flows is collected using literature studies, interviews, measurements, theoretical calculations, data banks and qualified guesses. In theory, the application of allocation principles (i.e. partitioning the input or output flows of a unit process to the product system under study) and procedures should also be explained, and information required in recycle or reuse situations should also be presented. For transparency, the details of the methods for data collection and sources of the data should also be provided in this phase.

Impact Assessment: SETAC and ISO define environmental life cycle impact assessment (LCIA) as the stage whereby the inventory results are linked to the identifiable environmental problems. This stage is a technical, quantitative and/or qualitative process to characterize and assess the effects of the environmental emissions identified in inventory analysis.

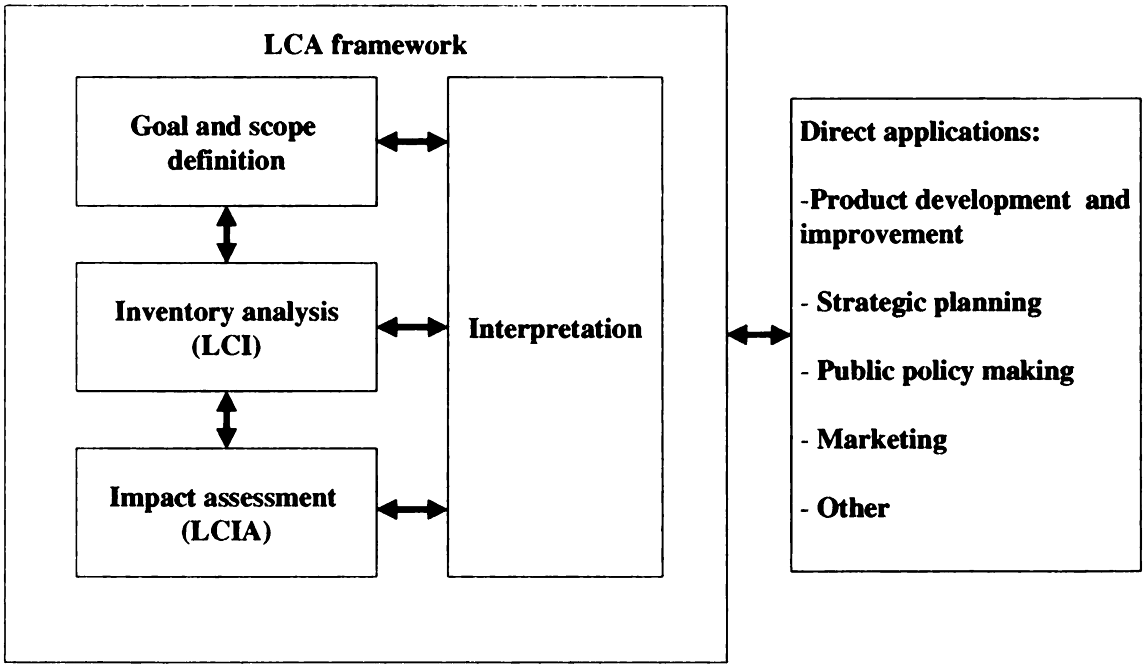


Figure 2. The stages of LCA and possible applications (ISO 14040, 1997)

There are a number of impact assessment methods (Bare et al., 2000b), and related concepts and terminology are still being developed, but in general they include three basic steps: step 1, identification of the potential environmental concerns (i.e. "impact categories") affected by the LCI component results; step 2 classification, actual assignment of LCI results under the identified impact categories (one LCI component may affect more than one impact category); and step 3 characterization, calculation of the contribution of the effect of LCI components to each identified environmental problem (i.e. category indicators).

There are two main approaches for estimating category indicators as outlined in Figure 3. A category indicator can be located at any place between the LCI results and the environmental "endpoint" (Jolliet et al., 2004).

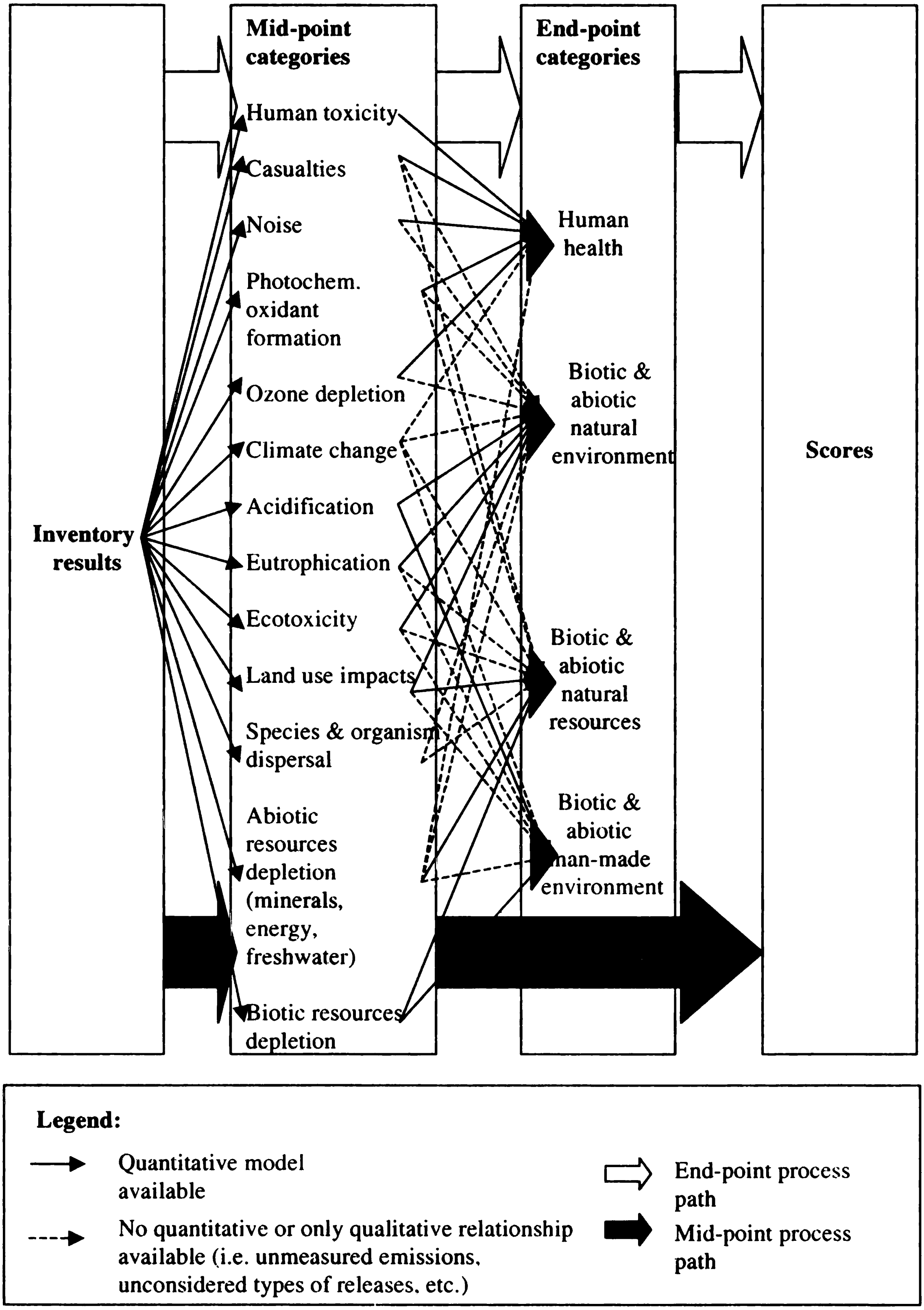


Figure 3. General representation of main differences between mid-point and end-point methods for impact assessment (based on Bare et al (1999) and Jolliet et al (2004))

One approach conforms to the so-called "mid-point" methods, which link the inventory results to environmental mid-point categories (e.g. ozone depletion or acidification or global warming potential). The term "midpoint" indicates that this point is located on the impact pathway at an intermediate location between the LCI results and the final environmental damage (i.e. endpoint). The alternate approach constitutes "end-point" or damage-oriented methods, which link the inventory results all the way to damages (e.g. damage to human health or animal species endangerment). In doing so, an additional step may be used to allocate the previous "midpoint categories" into one or more "damage categories" (Jolliet et al., 2004).

After the characterization step, though not required by ISO standards, additional procedures are usually followed to better organize the results under some type of rating to facilitate decision-making. In particular, normalization, valuation methods and/or weighting procedures of the resulting impact outputs can be used in order to convert characterization results into "impact scores" with the intention of facilitating the decision-making process.

Interpretation: where the results from the LCI, alone or combined with those from the LCIA, are integrated to reach conclusions and recommendations. This stage may involve the iterative revision of the goals and scope of the LCA, and assessment of the quality of the data.

V. Limitations

There is almost unanimous opinion about the enormous potential of LCA. However, the current LCA technique, though gaining international acceptance, is far from perfect. One thought that probably can sum up the skepticism is by M. Densie (Elkington and Hailes, 1993):

"The outcome of the LCA is the result of the inputs. The inputs are the result of the preferences of those who are paying for the study".

Further, since there is not a single, harmonized and standardized approach of performing an LCA (e.g. due to differences in allocation rules, data collection procedures, etc.), any result can be challenged. This is critical when LCA is used as a product comparison tool to determine product preferability. In fact, Finnveden (2000) argued that as long as no general framework is used in LCA, none of the studies can be used to show an overall environmental preference for any of the alternatives compared. The outlook is considered better when LCA is used as a tool for improving a system's environmental performance, since under the same approach on the same system, LCA could give useful information for strategic system improvement (e.g. for selecting the container size that appears to require the least life-cycle energy after analyzing the effect of using different container sizes to deliver a product to the consumer).

Regarding the LCA methodology, shortcomings along the different LCA stages were classified by Huijbregts (1998a) who elaborated that LCA limitations arise mainly due to different kinds of uncertainties and variability.

In industrial systems, uncertainty (i.e. the lack of sureness about something) and variability (i.e. the inherent variation of measured values) are responsible for many of the problems in an LCA (see Table 1).

Table 1. Critical issues at different stages of the LCA process (Based on Huijbregts (1998a) and Björklund (2002))

Problem	LCA phase					
	Goal and scope	Inventory (LCI)	Impact assessment (LCIA)			
			Choice of impact categories	Classifica tion	Characteriz ation	Weighting
Data uncertainty		Inaccurate or no emission measurements			Uncertainty in lifetimes of substances	Inaccurate normalizati on data
Model Uncertainty		Linear instead of non-linear modeling	Impact categories are not known	Contributi on of impact category is not known	Characteriz ation factors are not known	weighting criteria are not operational
Uncertainty due to choices (i.e. scenarios)	Choice of functional unit, system boundaries	Choice of allocation methods, technology level	Leaving out known impacts categories		Using several characteriz ation methods within one category	Using several weighting methods
Temporal variability		Differences in yearly emission inventories			Change of temperature over time	Change of social preferences over time
Spatial variability		Regional differences in emissions inventories			Regional differences in environmen tal sensitivity	Regional differences in distance to (political) targets
Variability between objects/source s		Differences in emissions between factories which produce the same product			Differences in human characterist ics	Differences in individual preferences , when using the panel method
Mistakes	Any	Any	Any	Any	Any	Any
Estimation of uncertainty		Estimation of uncertainty in inventory parameters			Differences in human characterist ics	Estimation of uncertainty in potential impacts

Uncertainty is important because when it is not evaluated, there is more risk that the impact predicted by the LCA will not match the actual environmental impact. Likewise, variability is important because it limits the precision of LCA results.

In the packaging field several of these limitations have been found to cause problems. For instance, Oki and Sasaki (2000) describe how sometimes it is difficult to take into account the packaging function as a basis for comparison. The authors compare a gas-barrier multilayer container with a monolayer container and claim that the current state of LCA would exclude the gas barrier function from the assessment. That would make the monolayer material more desirable because it means less material consumption and less processing energy, and lower environmental burdens. Though it is true that the gas-barrier material involves more energy and costs more, this material is winning the competition in the marketplace and its function reduces the transportation energy and emissions during distribution by extending the sales period.

Another popular source of uncertainty in LCA occurs when the effect of different "scenarios" such as for the "energy" used in the inventory stage is not discussed. It is argued that site-specific energy production data may produce very different conclusions than average or industrial world energy mixes, which are common scenarios used when site-specific data is not available (Paine, 2002). For example, results may differ when coal-generated electricity data is used for operations that occur in regions in which electricity is produced mainly by hydroelectric power. Coal-generated energy is considered "non-renewable" while hydroelectric power is produced from "renewable" resources and thus appears to be more environmentally friendly.

But perhaps one of the most uncertain parts of any LCA is the impact assessment stage. This is because of an array of reasons. For instance, as described earlier, there is no unified approach for implementation of the impact assessment process. In fact, in ISO words (ISO 14040, 1997): "*There are no generally accepted methodologies for consistently and accurately associating inventory data with specific potential environmental impacts*". The two approaches described earlier have their advantages and disadvantages. For instance, since they can aggregate categories under a common basis (e.g. DALYs: Disability Adjusted Life Years), endpoint methods may be preferred over midpoint methods for category weighting, but introduce more subjectivity and uncertainty (e.g. model, scenario and/or parameter) in the assessment since the closer one goes to the end-point categories (i.e. towards the right in Figure III), the more the models are highly dependent on the user's preferences (Bare et al., 2000a). Thus, though reconciling efforts are underway within the two main approaches, currently there is no consensus method (Jolliet et al., 2004) and comparison studies among these two approaches are often complex to develop and interpret (Dreyer et al., 2003; Pant et al., 2004).

Further, though according to ISO the LCA goal seems straightforward: "LCA is a technique for assessing the 'environmental aspects' and 'potential impacts' associated with a product throughout its whole life cycle", a methodology for studying the general environmental aspects has not even agreed upon. While some consensus has emerged on assessment methods to evaluate contributions to environmental impacts such as climate change, stratospheric ozone depletion, photochemical oxidant formation, acidification and eutrophication (Udo de Haes, 2002), the situation is not the same for impact categories such as resource extraction, land use and human health.

LCIA is further challenged by the fact that impact assessment methods rely on models, many of which are being developed or updated to account for current changes in the environment itself. And this stage is further hindered because of the sheer number of chemicals used today. In fact, it has been estimated that of the around 100,000 substances presently used in the world, only about 5% of even the 2000 most used substances have been screened for toxicity and fate (Tukker, 1999).

Lastly, several in the LCA community acknowledge that due to the enormous amount of uncertainty involved, product LCIA methodologies based exclusively on mathematical relations representing system flows from and into the environment have important limitations. An alternative is the use of value-based methods, but these in turn need to deal with the open issue of LCIA weighting (Bengtsson, 2001; Finnveden, 1997; Udo de Haes, 2000). Tucker (1998) offered three alternatives to this LCIA conundrum: use a "reductionalist approach" by reducing the LCIA scope to obtain a truly robust method; acknowledge the subjectivity of the LCIA method and develop an indicator system that reflects the views of an authoritative forum; or develop an LCIA method that includes the views of different sectors of the society and thus yields results in a more socially acceptable product evaluation.

ISO standards (ISO 14040, 1997; ISO 14042.2, 1998; ISO 14041, 1998; ISO 14043, 2000) recognize the limitations that arise from not effectively addressing uncertainty and variability effectively and urge the practitioner to do so in the LCA study. However, the ISO standards do not suggest any procedure to do so. Ross et al (Ross et al., 2002) found that out of 30 LCA studies only 3 (10%) included a quantitative or qualitative uncertainty analysis and conclude that the standards need to be revised to ensure that LCA includes at least a qualitative discussion of uncertainty and variability. Among attempts to provide a systematic analysis, Huijbregts (1998a; 1998b) and Huijbregts et al (2001) proposed the first general framework for comprehensive uncertainty and variability evaluation, adding the definition of key concepts in the LCA context. But uncertainty analysis is something new in LCA (Barnthouse et al., 1998) and thus a number of methodologies for uncertainty estimation have been proposed. Currently, the most favored approach for uncertainty and variability analysis in LCA is incorporating it into the life cycle inventory (LCI) stage. Computer simulation is becoming the method of choice. A number of recent uncertainty and variability methodologies have been proposed and incorporated into LCAs associated with different industry sectors and almost exclusively used Monte Carlo simulation (Canter et al., 2002; Ciroth et al., 2004; Contadini and Moore, 2003; Guyonnet et al., 1999; Huijbregts et al., 2003; Maurice et al., 2000; McCleese and LaPuma, 2002; Sonnemann et al., 2003). Others use fuzzy logic to perform a similar data evaluation (Guyonnet et al., 1999; Tan et al., 2002). In the packaging sector, despite the increasing use of LCA for evaluating alternatives, a few Monte Carlo uncertainty estimations have been published but with limited background information (World largest PET LCA, 2004; Canter et al., 2002; Kennedy et al., 1997; Kennedy et al., 1996). Moreover, although these studies have shown that uncertainty and variability can be incorporated in LCA, the exact implications for decision makers remain unclear.

VI. LCA enhancements

Enhancements, in the context of this explanation, refer to attempts and approaches to overcome some of the previously stated LCA limitations in order to increase or improve the LCA value, quality and ease of implementation. For the purpose of this discussion, these approaches are grouped into four categories: a) data-related improvements, b) streamlined LCA, c) input-output LCA, and d) economic analysis and LCA. These areas are some to which increasing research has been devoted in recent years, and it is expected that they will evolve from their current state into more established methods.

a) Data-related improvements

Data related improvements comprise three major aspects that are not necessarily separable: a) data collection, b) data availability, and c) data quality. Regarding data collection, the current status of LCI databases around the world can be used as an indicator of the situation. Worldwide, government, private organizations and research institutes are currently either expanding existing inventories or developing new ones (see Table 2).

Many LCI databases are sector oriented and some already follow ISO 14048 guidelines for data collection. For instance, the U.S. LCI database which is managed by the Athena Institute and hosted by the U.S. National Renewable Energy Laboratory (NREL) (Norris et al., 2003b) is a project to develop publicly available U.S.-based inventories originally focused on the building sector but later expanded to other industry sectors. However, this database has not been peer-reviewed yet and it is not as transparent as it needs to be for clear utilization. Other LCI databases cover several industry sectors and are usually accessible through commercial LCA-specific software developed by consulting companies.

Table 2. Summary of LCI databases and managing organizations and their status (updated from Norris and Notten) (2002)

Managed by \ Status	Completed†	Planned or under development
National and Multi-government*	Italy, Switzerland (BUWAL 250), Switzerland (Ecoinvent), SAEFL	Australia, Canada, Chinese Taipei, Japan, Korea, Sweden (SPINE), USA
Consultants and research institutes**	Denmark (EDIP), Sweden (CPM), Ecobilan (DEAM).	Austria, Denmark, France, Germany, Sweden, Switzerland, UK, USA
Industrial***	IISI, EAA, FEFCO, APME and PWMI, NiDI	
Academic/Decentralized****		Belgium, China, Chile, Estonia, Finland, India, Norway, The Netherlands, Portugal, Poland, South Africa, Spain, Vietnam, Argentina, Malaysia, Thailand

† may be updated

* Coordinated effort to produce nationally representative and accessible database. Typically involves collaboration between several organizations and some degree of government funding.

**Inventories produced by research organizations or consultants and made publicly available in a database, sometimes for a fee (e.g. databases included with LCA software).

***Inventories produced and published under the auspices of a particular industry organization. Includes cases where data made only partially available (e.g. for a fee, or only to parties with sufficient motivation for requesting the data). Most often data compiled by consultants, but includes cases where LCI development is done in-house, or by academic or other research organizations.

****Includes inventories compiled by academic or other research organizations, made either partially or fully available on an ad-hoc basis (e.g. through journal publications). Countries may have some degree of information sharing (e.g. an LCA society), but no coordinated data gathering effort (i.e. studies are not organized into an accessible database).

Regarding data availability, attempts have been made to improve the situation. For instance, the SETAC-Europe LCA Working Group on "Data Availability and Data Quality" was formed in 1998, with the goal to focus on the key features of improving the efficiency and quality of data collection (Huijbregts et al., 2001). Similarly, NREL in a joint project with the U.S. EPA has developed a global LCI inventory matrix to help centralize multiple-sector LCI databases. The information is obtained by voluntary submission and is categorized according to industry sector and life cycle stage. However, many of the specific LCI databases are accessible only by a fee (U.S. EPA, Global LCI matrix, 2002).

Data quality has been the subject of many publications in the LCA literature (Braam et al., 2001; Finnveden and Lindfors, 1998; Huijbregts et al., 2001; Rousseaux et al., 2001). Quality, often defined as "fitness for use", has been used to comprise aspects of the LCA data such as database format, uncertainty, reliability, completeness, age, geographical location and process technology. Thus, approaches to improve quality involve standardization in data collection and processing procedures; data validation by cross-checking of energy and mass balances; use of data quality goals (DQG) and data quality indicators (DQI); use of parameter estimation techniques; use of higher resolution models; critical review; and additional measurements (Björklund, 2002). Each of these approaches has its own disadvantages. For instance, since the development of standardized databases requires consensus among LCA practitioners, it is time and resource demanding. Likewise, making additional measurements of inventory data or using higher resolution models to obtain better estimates are time and resource intensive.

DQGs and DQIs are simpler and more flexible approaches but there is no consensus about their methodologies. In general, DQG is a qualitative scheme to specify the data quality requirements before actual data compilation. DQI can be either a qualitative, quantitative or semi-quantitative technique to assess the quality of already compiled data.

Alternatively, though limited in scope, sensitivity analysis, which is the study of the effect of changes in an independent variable on the LCA outcome, gives more knowledge about the behavior of the model. However, it is time-consuming when not coupled with some kind of data uncertainty importance analysis. In fact, data uncertainty importance analysis is a useful screening methodology that can help concentration on relevant model parameters by establishing some kind of contribution criteria to the overall LCA uncertainty outcome. However, as stated earlier, uncertainty analysis is still a new concept in LCA, and the information in databases required for the analysis (e.g. ranges, standard deviation, probability distributions) is very limited. In fact, publications considering the evaluation of data ranges are scarce. Among the few, Finnveden (Finnveden and Lindfors, 1998) analyzed the range of the common inventory data in a number of databases and his findings are shown in Table 3. The variation in this table indicates the typical uncertainty of European databases in the mid 1990s.

Table 3. Expected variations in LCI data (Finnveden and Lindfors, 1998)

Inventory parameter	Variation that can be expected
Central, non-substitutable resources	Factor of 2.
Less central and substitutable resources	A factor of 10 or more if they are completely substitutable.
Outflows that are calculated from inflows (e.g. CO_2)	The same as for the corresponding inflow.
Other energy related air emissions	Factor of 10.
Other process-specific emissions	Factor 10-100 or higher if mistakes or very different types of technologies can appear.
Total amount of solid waste.	Factor of 10.
Specific types of solid waste	The variation can be very high partly due to different classification systems in different countries.

b) Streamlined LCA

In practical terms, it can be expensive and time consuming to conduct an LCA according to ISO or SETAC guidelines. Streamlined or abridged LCAs are approaches that are used to obtain timelier and less expensive results. According to SETAC, there are two main kinds of streamlining: approaches within the LCA framework and alternative life-cycle approaches as shown in Figure 4.

Approaches within the LCA framework can be again subdivided into two: process oriented and methodology oriented. Process oriented streamlining methods deal with the actual operation of performing the LCA (e.g. making software with embedded ready-to-use databases, using process templates, etc.). Methodology oriented streamlining methods deal with simplifying the actual stages of the LCA. By limiting the goal and scope of the study, the LCI and LCIA steps can be simplified. Table 4 contains a list of the common streamlining decisions that have been used in the past, along with their advantages and disadvantages. Since all LCAs are streamlined to some extent, the degree of information "lost" by these techniques cannot be fully accounted for.

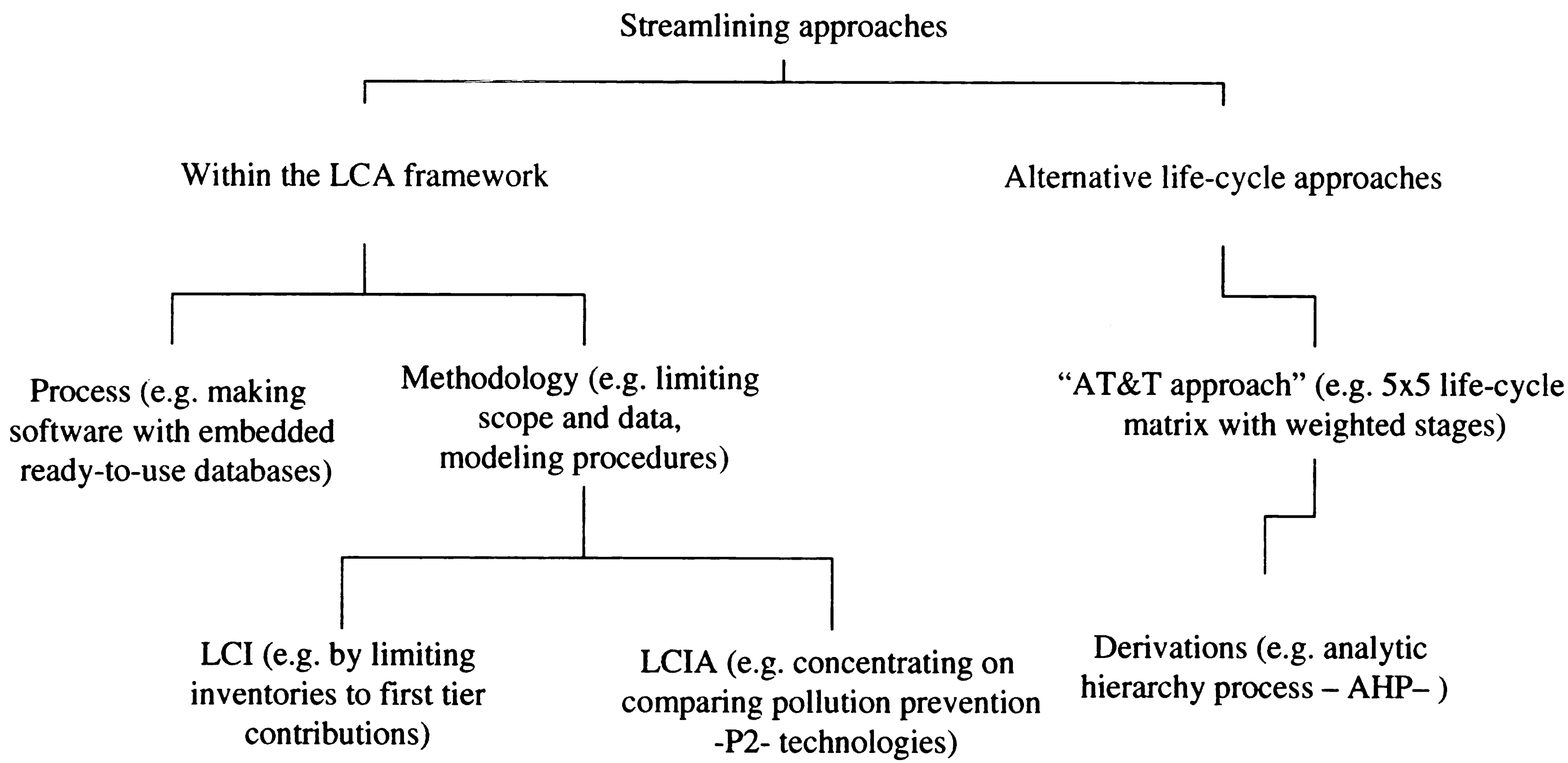


Figure 4. Streamlining approaches

Table 4. Methodology oriented streamlining approaches (based on Todd and Curran (1999), and Hunt et al (1998))

Streamlining approach	Application procedure	Advantages	Cautions
Removing upstream components	All processes prior to final material manufacture are excluded. Includes fabrication into finished product, consumer use, and post-consumer waste management.	Clear boundaries set. All the products and processes directly involved in producing the product are considered. Eliminates proprietary vendor data issue.	Important environmental consequences of raw material extraction or production may be eliminated from consideration, causing a skewed result.
Partially removing upstream components	All processes prior to final material manufacture are excluded, with the exception of the step just preceding final material manufacture. Includes raw materials extraction and precombustion processes for fuels used to extract raw materials.		
Removing downstream components	All processes after final material manufacture are excluded.	Captures some important environmental concerns within the life cycle useful in product improvement. Results can be used to support environmental procurement programs.	Ignores important stages in the life cycle. For example, the use stage for some products (e.g. building materials), final disposal (e.g. packaging).
Removing up- and downstream components	Only primary material manufacture is included, as well as any precombustion processes for fuels used in manufacturing. Sometimes referred to as a "gate-to-gate" analysis.	Data gathered and processes under the study can be directly affected by user. Results likely to be useful to sponsor.	Actual life cycle of material is missed.
Using specific entries to represent impacts	Selected entries are used to approximate results in each of 24 impact categories, based on mass and subjective decisions; other entries within each category are excluded.	Focuses on environmental considerations deemed important by the user. Helpful when regional considerations are of critical importance.	Other important environmental considerations can be excluded, thus results-based decisions may not be the best for the environment or human health.

Table 4. Methodology oriented streamlining approaches (based on Todd and Curran (1999), and Hunt et al (1998)) (cont'd)

Streamlining approach	Application procedure	Advantages	Cautions
Using specific entries to represent LCI	Specific entries from the individual processes comprising the LCI that correlate highly with full LCI results are searched for; other entries are excluded.	Selected surrogate inventory entries may be useful to evaluate potential "what ifs" scenarios.	Surrogates must be carefully chosen to ensure that surrogate truly represents the product, material, or process under study.
Using " showstoppers" or "knockout criteria"	Criteria are established that, if encountered during the study, can result in an immediate decision.	Focuses on specific issues deemed important by the user. No need to explore effects of all constituents.	Other important environmental considerations can be excluded, thus results-based decisions may not be the best for the environment or human health.
Using qualitative or less accurate data	Only dominant values within each of 6 process groups (raw materials acquisition, intermediate material manufacture, primary material and product manufacture, consumer use, waste management, and ancillary materials) are used; other values are excluded, as are areas where data can be qualitative, or otherwise of high uncertainty.	All potential environmental issues are detected at each stage of the life cycle. Some environmental factors not readily amenable to quantification (e.g.biodiversity, habitat issues), can be considered.	Difficulty in assessing importance of each environmental concern in overall life cycle and in comparison to other products.
Using surrogate process data	Selected processes are replaced with apparently similar processes based on physical, chemical, or functional similarity to the datasets being replaced.	Estimates can be developed for data that would otherwise be unavailable.	Surrogates must be carefully chosen to ensure that surrogate truly represents the product, material, or process under the study.
Limiting raw materials	Raw materials comprising less than 10% by mass of the LCI totals are excluded. A 30% limit has also been used.	Limits the number of items and focuses on those that are likely to be most important for the product under study. Easy to define clearly and does not have an inherent bias.	By focusing only on volume and disregarding hazard or toxicity, important environmental effects may be overlooked.

Alternative life-cycle approaches do not involve a complete inventory analysis and do not follow the inventory/impact/improvement analysis path required by ISO. Instead, they attempt to evaluate relative differences among alternatives along their life cycles. One of the best examples of this approach is the one developed in 1993 by Graedel and Allenby at AT&T (2003) called environmental responsibility product assessment (ERPA) matrix. The ERPA method divides the product life cycle into 5 stages: pre-manufacture, product manufacture, product delivery, use, and recycling or disposal; and considers 5 environmental concerns: material choice, energy use, solid, liquid and gaseous residues. These two dimensions are presented in a matrix format in Table 5.

Table 5. Environmentally responsible product assessment matrix (Graedel and Allenby, 2003)

Life cycle stage	**Environmental concern**				
	Material choice	**Energy use**	**Solid residues**	**Liquid residues**	**Gaseous residues**
Premanufacture (i.e., raw material extraction/production)	(1,1)	(1,2)	(1,3)	(1,4)	(1,5)
Product manufacture	(2,1)	(2,2)	(2,3)	(2,4)	(2,5)
Product delivery	(3,1)	(3,2)	(3,3)	(3,4)	(3,5)
Product use	(4,1)	(4,2)	(4,3)	(4,4)	(4,5)
Recycling, disposal	(5,1)	(5,2)	(5,3)	(5,4)	(5,5)

As indicated in Table 5, each cell of the resulting 5x5 matrix is then assigned a score ranging from 0 (highest impact of a stage on an environmental concern item) to 4 (lowest impact of a stage on an environmental concern item). By this scoring technique, the method estimates the results of more formal LCI and LCIA, and also takes into account whether the possibilities of reducing impacts have been utilized or not (Hochschorner and Finnveden, 2003). The scores are assigned by consideration of information from actual life cycle studies, checklists, manufacturing surveys and experience. The ratings in a matrix can be added up (i.e. to sum up to a maximum of 100) or can be plotted in a circumference target plot for more convenient evaluation (e.g. center or zero represents highest impact, circumference represents lowest impact). This technique, which critics say is subjective, was found useful for identifying hot spots and opportunities for environmental improvement and has been used recently for a number of different product categories and applications range from re-refined oil in Japan (Nakaniwa and Graedel, 2002), to evaluating the environmental impact of a residential refrigeration unit with a proposed maintenance and take back service (Bennet and Graedel, 2000).

c) Input-Output LCA

In conventional (SETAC/ISO) LCAs, the system boundary is usually chosen with the assumption that addition of successive upstream production stages has a small effect on the total inventory. However, truncation errors inherent to conventional LCAs have been estimated and in cases can be of the order of 50% (Lenzen, 2001). One way to address the boundary issue in LCA is by using economic input-output methods. Economic input-output LCA (EIO-LCA) is the result of applying economic input-output (EIO) analysis to help perform a life cycle assessment. The EIO analysis is based on using the EIO matrices that are regularly estimated for most developed countries and economies. An EIO matrix is a transaction matrix that shows the relationship between the different sectors that form part of an economy. Table 6 shows the basic structure of such a matrix in which entries are expressed in dollars. For example, a_{12} is the amount of dollars required to (directly and indirectly) input in sector 1 (e.g. electricity) to obtain $1 worth of sector 2 (e.g. aluminum sheeting) output. For the U.S., the EIO matrix has 519 sectors, so it is a 519x519 matrix.

Table 6. Basic structure of an EIO matrix

Output / Input	**Sector 1**	**Sector 2**	......	**Sector n**
Sector 1	a_{11}	a_{12}	......	a_{1n}
Sector 2	a_{21}	a_{22}	......	a_{2n}
⋮	⋮	⋮		⋮
Sector n	a_{n1}	a_{n2}	......	a_{nn}

The basic approach for EIO-LCA then can be compactly summarized using matrix algebra notation in equations 1 and 2 as described by Lave and colleagues (1998; 1995):

$$\mathbf{X} = (\mathbf{I} - \mathbf{D})^{-1}\,\mathbf{F} \qquad (1)$$

$$\mathbf{B} = \mathbf{R}\,\mathbf{X} \qquad (2)$$

In equation 1, X is a vector containing the total output (in dollars) from different sectors of the economy required to meet a desired final demand, I is a identity matrix (i.e. to include the output of the aluminum sheeting sector itself), D is the EIO matrix, and F is a vector representing the desired final demand (e.g. dollars worth of a desired amount of aluminum sheets). In equation 2, B is the vector containing the economywide environmental burdens (e.g. toxic emissions or electricity use), and R is a matrix representing the environmental burden per dollar output of each sector.

By setting the boundary of the LCA on the level of the national economy, EIO-LCA with the EIO matrix attempts to address the boundary issue including the interdependence of different processes. However, this analysis has its own problems including the high level of aggregation (i.e. combination of product and technology information) in industry or commodity classifications, which limits the level of detail of EIO-LCA studies. Moreover, there is incompleteness of sector-based environmental statistics, which in turn limits the accuracy of the EIO-LCA results.

Due to the previous reasons, conventional LCA is often seen as more detail oriented. However, these analyses are more labor- and time-intensive, and suffer from the stated truncation error (i.e. due to omission of contributions outside its finite boundary). In fact, due to the quick and inexpensive nature of the EIO-LCA approach developed by researchers at Carnegie Mellon University, Matthews and Lave (2003) suggested the use of this system to help in corporate benchmarking efforts to evaluate the environmental performance of their operations. Moreover, using input-output techniques, Suh (2001) a researcher at Leiden University has recently developed the MIET, an inventory estimation tool for missing flows.

"Hybrid" analyses combine process-level data with sector-level input-output analysis and thus try to get the best of both approaches. According to Suh et al (2004), hybrid approaches can be grouped into three different categories, which are tiered hybrid analysis, input-output based hybrid analysis, and integrated hybrid analysis. Table 7 summarizes the main aspects of each approach along with their perceived advantages and disadvantages.

Table 7. Main hybrid approaches combining process-based analysis and input-output based analysis (based on Suh et al (2004))

Approach	Characteristics	Advantage	Disadvantage
Tiered hybrid analysis	Direct and downstream requirements (e.g., construction, use, maintenance, and end-of-life) and some important lower order upstream requirements of the product life cycle are examined in a detailed process analysis. Remaining higher order requirements (e.g., materials extraction and manufacturing of raw materials) are covered by input-output analysis. Exact location and comparability of the boundary between the process and input-output analysis part depends on data availability, desired detail and accuracy, and constraints in terms of cost, labor, and time.	Easy to use. May be useful to address dependency upon imports.	Issues with double counting. Issues with recurring flows (e.g. recycling, reuse) in process-based analysis part.
Input-output based hybrid analysis	Major input-output sectors are further disaggregated in case more detailed sectoral monetary data are available..Disaggregation may reach a resolution of the level of process-specific studies.	Consistent method. Avoids double counting.	Use and end-of-life phase are externally added. Issues with recurring flows (e.g. recycling, reuse) between use and end-of-life stages and input-output part.. Should be combined with other methods if national economy is highly dependent upon imports.
Integrated hybrid analysis	The process-based system is represented in a technology matrix by physical units per unit operation time of each process while the input-output system is represented by monetary units. This model links the process-based and the input-output-based systems through flows crossing the border.	Consistent matrix framework for the whole life cycle. Avoids double counting. Easy to apply analytical tools.	Complex to use. Time- and data-intensive.

The connection between process-based and input-output-based LCA is a topic under development (Heijungs and Suh, 2002) and thus much research work still needs to be done to define this relationship. Nevertheless, recent assessment studies have already benefited from the information provided by these hybrid analyses. For example, by using a hybrid approach, Norris et al (2003a) were able to estimate the energy consumption during the factory-to-mall phase of life cycles that has been universally neglected in process-style LCAs; and Nakamura and Kondo (Nakamura and Kondo, 2002) have developed a hybrid approach that expanded the input-output system to include waste flows and showed that the EIO model is in fact a special case of their model.

d) Economic analysis and LCA

While LCA can be useful for evaluating environmental attributes of a system, it is often criticized for not providing monetary information that business managers routinely need to allocate the often scarce capital resources available to minimize the environmental footprint of their business operations. Thus, various approaches have been developed to supplement environmental information with cost information and enhance the decision-making process. The central challenge is estimating of the "environmental cost" of business operations and a whole body of concepts and terms has been developed under the umbrella of "environmental accounting" to address this issue (U.S. EPA, Introduction to environmental accounting, 1995; Shapiro, 2001).

The scope of the present discussion comprises life cycle based approaches to estimate environmental costs of products. One of these approaches is life cycle costing (LCC). LCC is a systematic procedure for identifying environmental consequences along the life cycle of a product (i.e. product line, process, system or facility), and assigning measures of monetary value to those consequences using accounting procedures. This process includes the assessment of material flows (e.g. amount of solid waste generated) through the product system (i.e. materials accounting, essentially a kind of LCI) as well as costs (i.e. cost accounting), including environmental costs (e.g. waste disposal).

Despite the apparent compatibility of approaches, LCA and LCC have important methodological differences. For example, while LCA attempts to evaluate the relative environmental impact (from a broad societal perspective) of alternative product systems that perform the same function, LCC intends to estimate the relative cost effectiveness of alternative investments and business decisions, very often from a private perspective. Thus, LCA and LCC actually consider life cycles with different spans and flows (i.e. physical or energy units vs. monetary units) that are not necessarily compatible. In a succinct table (Table 8), Norris (2001a; 2001b) summarizes the extent of the differences between life cycle assessment and life cycle costing methodologies. Furthermore, the LCC outcome is limited when used at the product design stage (e.g. Design for the Environment programs), since it suffers from greater uncertainty than LCA (Schmidt, 2003). This is because future technological changes have a strong effect on the results, and because of specific additional factors (e.g. interest rate and market dynamics) that are not always stable and are independent from technology changes.

Table 8. Differences between LCA and LCC (Norris, 2001b)

Items \ Tools	LCA	LCC
Objective	Compare relative environmental performance of alternative product systems for meeting the same end-use function, from a broad, societal perspective	Determine cost effectiveness of alternative investment and business decisions, from the perspective of an economic decision maker such as manufacturing firm or a consumer
Scope of life cycle	Supply chain of processes supporting usage phase; entire physical usage	Activities directly causing costs or benefits to the decision maker during the economic life of the investment as a result of the investment
Flows considered	Pollutants, resources, and interprocess flows of materials and energy	Direct costs and benefits to decision maker
Units for tracking flows	Physical and energy units	Monetary units (e.g. dollars)
Time treatment and scope	Timing ignored; all causally linked flows, and some of their impacts collapsed in time and valued equally regardless of timing	Timing is critical; present valuing (discounting) of costs and benefits; specific time-horizon scope, outside of which costs and benefits are ignored.

However, by offering direct opportunities for cost reduction, LCC is perceived to help to promote life cycle based analysis. In turn, economic analysis with a life-cycle perspective has the potential of discovering "hidden" costs (see Table 9, cost types 2, 3, 4 and 5) and revenue impacts that are otherwise neglected in conventional economic analyses. Very often, though, LCC users utilize the pragmatic approach of focusing exclusively on internal or "private costs" (i.e. type 1 and some type 2). Just recently, some comprehensive approaches have been developed to bridge the gap between LCA and LCC and to improve it to allow for easier identification of hidden costs. For instance, Total Cost Assessment (Total cost assessment methodology, 1999), method developed by a joint effort of private companies and the American Institute of Chemical Engineers' Center for Waste Reduction Technologies, is an approach that facilitates the inclusion of environmental costs into a capital budgeting analysis by classifying costs into categories shown in Table 9.

Table 9. Categories of costs according to AICHE/CWRT (Center for Waste Reduction Technologies-AIChe, Total cost assessment methodology, 1999)

Cost Type	Description
Type 1: direct	Direct costs of capital investment, labor, raw material, and waste disposal. May include both recurring and nonrecurring costs. Includes both capital and operations and maintenance (O&M) costs.
Type 2: Indirect	Indirect costs not allocated to the product or process (overhead). May include both recurring and nonrecurring costs. Includes both capital and O&M costs.
Type 3: Contingent	Contingent costs such as fines and penalties, costs of forced cleanup, personal injury liabilities, and property damage liabilities.
Type 4: Intangible	Difficult to measure costs, including consumer acceptance, customer loyalty, worker morale, union relations, worker wellness, corporate image, and community relations.
Type 5: External	Costs borne by parties other than the company (e.g., society).

Several alternative versions of life cycle costing methodologies have also been developed, mostly by interested companies, and confusion still exists about the concepts, scope and terminology involved (Total cost assessment methodology, 1999; Sonnemann et al., 2001).

VII. LCA and packaging

Packaging situations were one of the earliest applications of LCA. Harry E. Teasley, Jr., manager of the packaging operations at the Coca Cola Company was credited for first devising the analytical scheme of quantification of material, energy and the environmental burdens of a package over its complete life cycle from raw material to disposal in 1969 (Hunt and Franklin, 1996). About three decades ago, public concern over increasing volumes of solid waste due to the use of plastic in packaging and later concerns about energy consumption became the major driving forces to study the effects of packaging on the environment (Sonneveld, 2000). A representative study is the comprehensive energy analysis of production and use of packaging systems published by Boustead and Hancock (1981). Eventually, these studies evolved into the comprehensive tool that LCA is today and example studies are those such as that published by the Swiss Agency for the Environment, Forests and Landscape (SAEFL) (LCI for packaging Vol. I, 1998a; LCI for packagings. Vol. II., 1998b) and the series of "eco-profiles" of plastic resins and intermediates, conversion processes, and packaging published by the Association of Plastics Manufacturers in Europe (APME) (2002; 2003).

Applications of LCA in packaging situations can be divided into two main categories depending on which constituencies use it, namely: a) stakeholder, and b) third-party organizations.

a) Stakeholder use of LCA

A stakeholder, is any interested (often private) body that might use LCA, or some form of LCA, in decision making regarding product design, product improvement, product comparison, strategic planning, compliance with regulatory policy, marketing, academic purposes, etc.

Recent examples of the use of LCA for product development and improvement purposes can be drawn from several industries. Most of them resulted from corporate environmental stewardship programs that generally involve proactive premises such as design-for-the environment (DfE). DfE intends to integrate health and environmental considerations into business decisions and it has been applied in Europe as well as in the U.S. While in Europe most of these DfE programs are voluntary and often internally adopted by companies that want to comply with the strict disposal regulations in place, in the U.S. DfE is mostly known as a voluntary partnership between the U.S. Environmental Protection Agency and industries to attempt to help with pollution prevention (Hart et al., 1995). DfE principles, along with integrating health risks, aim to use "accepted" results from LCA studies to create the attractive but unproven concept of an "environmental preference ranking" or "environmental indices" for the selection of materials to be used when designing new products (Industrial Designers Society of America, EPA partners with IDSA, 2005; Toloken, 2004). The Cleaner Technology Substitute Assessment (CTSA) methodology developed by the EPA, which involves comparative evaluation of substitute technologies, processes, products or materials, regarding human health, environmental risk, performance, cost and resource conservation, is another tool (along with LCA) that is used under the DfE premise (Kinkaid et al., 1996) to try to help with process selection. Many DfE programs are internally developed by companies that often claim substantial economic benefits after implementing DfE-recommended improvements. For instance, Xerox Europe using DfE under its "waste-free initiative", reported utility savings by pushing towards the reuse and recycling of equipment components through appropriate labeling and improving disassembly, as well as the reuse of packaging components by reducing the number of pallet styles and boxes used for new equipment, and by switching from conventional single-use corrugated boxes to wooden and steel totes for the collection of used equipment (Maslennikova and Foley, 2000). Likewise, U.S. examples exist on the use of LCA for product improvement purposes by studying packaging options. For instance, an LCA conducted to evaluate the environmental performance of the yogurt product delivery system used by Stonyfield Farm Inc. (Keoleian, 2001; Keoleian et al., 2004) analyzing different packaging formats (i.e. 4, 6, 8 and 32 oz polypropylene cups and 2 oz linear low-density polyethylene), estimated that the greatest potential improvements were the redesigning of the primary packaging and the use of alternative manufacturing techniques for the yogurt cups. The study indicated that in this case, shifting from injection molding to thermoforming of 32 oz containers reduced the life cycle energy by 18.6% and solid waste by 19.5%, primarily due to light-weighting. The authors claimed that elimination of overcaps for 6 oz and 8 oz containers provided similar advantages, and indicated that the effect of container size was significant when it was found that delivering yogurt in 32 oz instead of 6 oz containers could save 14.5% of the life cycle energy and decrease solid waste by 27.2%.

Along with product improvement applications, partnership of industry with research institutions using packaging related "case studies" attempted to evaluate the environmental aspects of a number of industry operations (Eide, 2002; Keoleian, 2001; Keoleian et al., 2004; Keoleian and Spitzley, 1999; Ross and Evans, 2002) as well as the LCA methodology itself. For instance, researchers from the University of Melbourne used a case study involving the utilization of different packaging formats by an Australian-based maker of refrigerators, to estimate the effects of excluding and including site-specific data (Ross and Evans, 2002). By limiting their analysis to a single non-global cumulative impact category such as the presence of significant photochemical precursors in the atmosphere, they reported the ability to assess whether an improvement in protective packaging produced any noticeable change in this impact category under two scenarios (i.e. when aggregated into a single global parameter or when spatial and temporal factors were taken into account).

LCA has been often used for product comparison purposes as well, but due to its nature, the assessment shows the environmental implications of different choices and the trade-offs that need to be made, instead of a clear answer. Nevertheless, oftentimes the complexity in the interpretation of the results is overlooked in many LCA comparison reports that, deliberately or not, portray one alternative as more environmentally sound than the other.

A survey of the literature in order to attempt an analysis of the use of LCA in packaging comparisons was presented by Martino (2005) and is shown in Table 10. The table presents a list of some packaging oriented LCA and LCI involving comparison studies which have been released or published in peer-reviewed journals, summary reports or books, in the last fifteen years. It can be seen from the Table that, regardless of the packaging formats evaluated in the studies, their scope is relative consistent comprising processes from raw material extraction to, in most cases, disposal. Likewise, in most cases, the functional units have been selected comprising the containment of the product and in some instances including its delivery to the consumer.

Consistency is also noticed with regards to the inventory parameters/impact assessment indicators since all of them include energy used (though many don't indicate whether it is a gross or net value), and some include warming emissions. In some instances the impact assessment indicators belong to a pre-established set based on a particular method (e.g. SimaPro).

Not surprisingly, the consistency diminishes when analyzing, the data sources and the types of scenarios considered, since they are directly related to the systems studied and the selected functional units. In general, in only one instance an uncertainty analysis has been included and its results included in the conclusion. It can also be noticed that when such studies are used for marketing purposes, their conclusions seem less qualified and more absolute and there is no critical review included.

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Hot drink	Single use polystyrene (PS) and paper cups	Process based/Public databases date around 1981	Case study	Extraction of raw materials, paper and plastic manufacture, use and disposal (i.e. landfilling and incineration). Recyling of packaging that is not disposed.	N/a	Same size of hot drink cup for fast food applications	Raw material and utility (i.e. steam, cooling water, power) use, air and water emissions.	PS cups consume less raw materials, utilities. Produce less air and water emissions with the exception of some alkane emission.	Yes	Hocking; Wells; McCubbin; Cavaney; Cammo, 1991 (Hocking, 1991; Wells et al., 1991)
Pallet load	Polyethylene (PE) stretch wrap and a glue-like unitizing system (i.e. Lock'n Pop®).	Process based/private and public databases dated around...	Marketing	Extraction of raw materials, processing, manufacturing, use and disposal (i.e. incineration and landfill). Incineration is credited.	N/a	Unitization of 1625 model pallet loads	Energy, oil and landfill use, carbon dioxide and water emissions.	Lock'n Pop® use less energy and oil, produce less emissions and wastes.		Dumbleton Consulting, 1992 (Plastic stretch wrap vs. Lock'n pop unitization system, 1992)2)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Soft drink	Like-sized containers: polyethylene terephthalate (PET) (16-ounce), glass (16-ounce) and aluminum (12-ounce) soft drink containers.	Process based/Private database	Marketing	Extraction of raw materials, processing, manufacturing and filling of primary containers to secondary packaging and distribution. Disposal burdens of material that is not recycled. Incineration is credited.	1995 reclycing rates	1000 gallons of soft drink purchased by consumer	Energy, Waterbone and airborne emissions, solid waste	PET consumes as less energy than one-way glass and as much energy as aluminum. PET produces less waterborne and airborne emissions, less waste		Franklin Associates for the National Association for Plastic Container Recovery, 1995 (The environmental impact of softdrink delivery systems, 1995)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Beer	One way and returnable glass bottles	Process based/ Private database (Belgium brewery)	Case study	Production and transport of bottles to filling, production of secondary and tertiary packaging and transport to distributor. Energy for washing and transporting returned bottles. Transport and replacement by new bottles plus transport and recycling of the bottles that are removed from the system.	15% glass recycling rate. Different break rates, transport distances, truck tonnage	Packaging and delivery of 1000 l of beer in bottles of 25 cl	Energy	Regardless of transport distance, when break rate is <5% returnable glass consumes less energy than one way glass bottles	No	Van Doorselaer and Lox, 1999 (Van Doorsselaer, 1999)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	**Packaging formats and materials compared**	**LCA type/Data source**	**Main Application**	**Scope highlights**	**Scenarios considered**	**Functional unit/basis for comparison**	**Inventory/ Impact indicators considered**	**Summary of conclusions**	**Critical review**	**Year and reference information**
N/a	Nylon 66, Nylon 6, polycarbonate (PC), low density polyethylene (LDPE), polypropylene (PP), high impact polystyrene (HIPS), general purpose polystyrene (GPPS), polyethylene terephthalate (PET) and polylactide (PLA) resin material	Process based/Private and APME databases, own estimations,	Product development/ improvement	Craddle to gate study. Raw material extraction and resin manufacture of PLA. Values for petrochemical polymers were obtained from related APME studies.	PLA generation I and II polymerization processes, combined with energy source alternatives such as biorefinery public electricity grid and wind power	1 Kg of polymer	Gross energy, global warming emissions, fossil fuel and water use	PLA resin uses less gross energy and produces less GWP emissions than the rest. PLA is surpassed only by PET in terms of least water consumption.	No	Vink et al, 2003 (Vink et al., 2003)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Yogurt	8oz polylactide (PLA) and polypropylene (PP) thermoformed cups	Process based/Private database	Case study	Extraction of raw materials, container manufacturing, use, disposal (i.e. landfilling). Recyling of product that is not disposed.	Double and triple effect evaporation for aquous lactic acid distillation. Landfill biodegradation of PLA to methane with methane collection and combustion. No landfill biodegradation of PLA.	1000 kg of yogurt purchased by consumer	Global warming emissions and gross energy	Thermoformed PLA cups consume less energy than PP cups as long as triple effect evaporation is used for lactic acid recovery. Difference is within margin of error when double effect evaporation is used. PLA and PP greenhouse gas emissions from landfill are equivalent provided PLA does not biodegrade. Otherwise PLA greenhouse emissions are higher than PP's.		Bohlman. 2004 (Bohlman. 2004)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Yogurt containers	Conventional wooden pallet and a specific-purpose bulk packaging system (i.e. Enviropak® T760).	Process based/APME, RMIT and SimaPro software	Case study	Manufacture of raw materials into primary materials (e.g. resins, timber, etc), packaging manufacturing, transport, use, recycling and disposal of the two packaging systems in New Zealand, Excluded upstream processes, (i.e. extraction of raw materials, manufacture and maintenance of equipment).	Uncertainty information about weighting factors included in analysis.	A unit of the plastic packaging system (i.e. Enviropak® T760) and a wooden pallet.	Impacts included in the Environmental Priority Strategy 2000 Default method.	The Enviropak® T760 obtained a better score than the wooden pallet across the impacts considered in the EPS 2000 Default method.		Lee and Xu, 2004 (Lee and Xu, 2004)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Soft drinks and mineral water	1.5 l one-way polyethylene terephthalate (PET) and 0.7 l refillable glass bottles. Other sizes are also included.	Process based/Not specified in source surveyed	Marketing	Extraction or raw materials, container manufacturing, use, disposal (i.e. incineration and landfill). Recycling of product that is not disposed using the expanded boundaries approach.	German kerbside collection and recycling system (Germany) and deposit based recycling system (Far East).	1000 l of beverage	Global warming emissions, fossil resource, acidification, terrestrial and acuatic eutrophication, smog, use of nature.	Environmental impact differences between the 1.5 l one-way PET bottles and the reusable 0.7 l glass bottles are within margin of error of the study when both are recycled within the German kerbside system. When PET bottles are recycled outside German system, the difference is notable.		Institute for Energy and Environmental Research (IFEU) in Heidelberg, Germany for the PET Container Recycling Europe, 2004 (World largest PET LCA, 2004)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Mail-in-order soft goods	Corrugated boxes with various types of dunnage and shipping bags composed of paper and/or plastic.	Process based	Purchasing analysis	Raw material extraction, packaging manufacture, transportation to order preparing facility, transportation to customer, disposal. Recycling and reuse also considered.	Specific packaging systems included more than one material (plastic, paperboard, paper, etc). Most packaging components were analyzed under two levels of recycled content.	10,000 arbitrary (i.e. 17.5" x 12" x 2.5" - uncompressed height- and a weight of 1.28 pounds) packages of soft goods items to customers	Energy use, air and water emissions, solid waste.	Weight of packaging is the most critical factor influencing environmental indicators. For example, box systems which were more than four times heavier than bags required more production energy. Boxes twice as heavy as bags produced more waste and greenhouse gas emissions.	Yes	Franklin Associates for the Oregon Department of Environmental Quality and the U.S. EPA, 2004 (Life cycle inventory of packaging options for shipment of retail mail-in-order softgoods, 2004)

Table 10. List of selected packaging oriented LCA and LCI involving comparison studies (cont'd)

Product	Packaging formats and materials compared	LCA type/Data source	Main Application	Scope highlights	Scenarios considered	Functional unit/basis for comparison	Inventory/ Impact indicators considered	Summary of conclusions	Critical review	Year and reference information
Tape recorder	Expanded polystyrene and corrugated paperboard inserts	Process based/European Manufacturers of EPS packaging (EUMEPS), European Database for Corrugated Paperboard Life Cycle Studies, APME and journal publications.	Case study	Raw material extraction, insert manufacturing, assembly, transportation, use and transportation and disposal of packaging materials.	Redesigns of inserts using less material weights. Different end-of-life scenarios (i.e. landfilling and incineration rates) in Siingapore	Same internal protective function of holding a tape recorder securely in a box.	Impacts included in SimaPro LCA Version 5.0 software's Eco-indicator 99 method: climate change (global warming emissions), acidification /eutrophication, ecotoxicity, fossil fuels and respiratory inorganics.	Both redesigned inserts obtained better scores than original ones across the impacts considered by Eco-indicator 99. Higher rates of incineration, as opposed to landfilling, resulted in better Eco-indicator 99 scores.		Tan and Khoo, 2005 (Tan and Khoo, 2005)

Another popular use of LCA in packaging is when LCA has been used from a waste management perspective to attempt to identify environmental burdens of a certain waste management operation, or to determine what is the environmentally better waste treatment system for packaging materials (Arena et al., 2003; James et al., 2002; Neumayer, 2000; Wollny et al., 2002). Within the waste management programs often studied by LCA, often included are take-back programs, which are taking root in a number of industries (i.e. automobiles, computers and other electronic devices, adhesives and garments) for dealing with the disposal of their products. Product take-back programs and "extended producer responsibility" (EPR), have become popular in many countries (Scarlett, 1999; Schiffler, 2002a; b). The take-back systems are founded on the idea of product "recovery" by the manufacturer or "reverse logistics" and by involving appropriate planning, managing, and optimizing of the forward as well as reverse distribution streams of both new and used products. They have to be not only cost-effective, but also to reduce the environmental footprint of an operation, but not necessarily by packaging reuse. While a number of these programs have developed as the result of legislation, some of these programs are voluntary, set by industries. For instance, the electronic components industry, due to its high volume of bulk shipments to a relatively small number of globally distributed locations and the near pristine condition of the packaging used after shipment, seems to be especially suitable for instituting packaging take-back systems (Matthews and Axelrod, 2004; Matthews, 2004). Furthermore, these take-back systems may be part of broader initiatives such as DfE.

b) Third party use of LCA

A third party, in this case, is any independent (governmental or private) body that might use LCA, or some form of LCA, for either of two purposes: 1) help with environmental labeling programs, or 2) help with policy-making.

Environmental labeling programs

By a third party, LCA can be used to help obtain information required by some environmental labeling schemes. For most developed (and some developing) countries, environmental labeling in general comprises more or less the types of environmental labels listed in Figure 5.

In recent years, steps for harmonizing and standardizing environmental labeling programs worldwide have been taken by ISO, with the release of a set of standards in which it recognizes three types of voluntary environmental labels: Type I (Environmental labels and declarations), Type II (Self-declared environmental claims) and Type III (Technical report-environmental labels and declarations). ISO Type II labels, also called "green claims" or "green symbols" are issued by the interested party which itself creates the label, applies the label, and establishes controls to ensure that its product meets the claims on the label (Weber Marin and Tobler, 2003). In some cases, these claims may be certified under single-attribute third-party certification programs. Green claims are the most widely used environmental labels (Lavallée and Plouffe, 2004) and are not based on product life-cycle concepts. Instead, they are general statements about whether a product is recyclable, contains recyclable material, is degradable/biodegradable/photodegradable, or compostable or source reduced, or refillable, or ozone safe, etc. In countries with regulations that allow the use of this type of labels, claims are required to be accurate and not misleading in order to comply with national legislation and trade regulations. For example, in the U.S., federal (Federal Trade Commission) as well as state bodies have developed guidelines to regulate such claims (Levy, 2000).

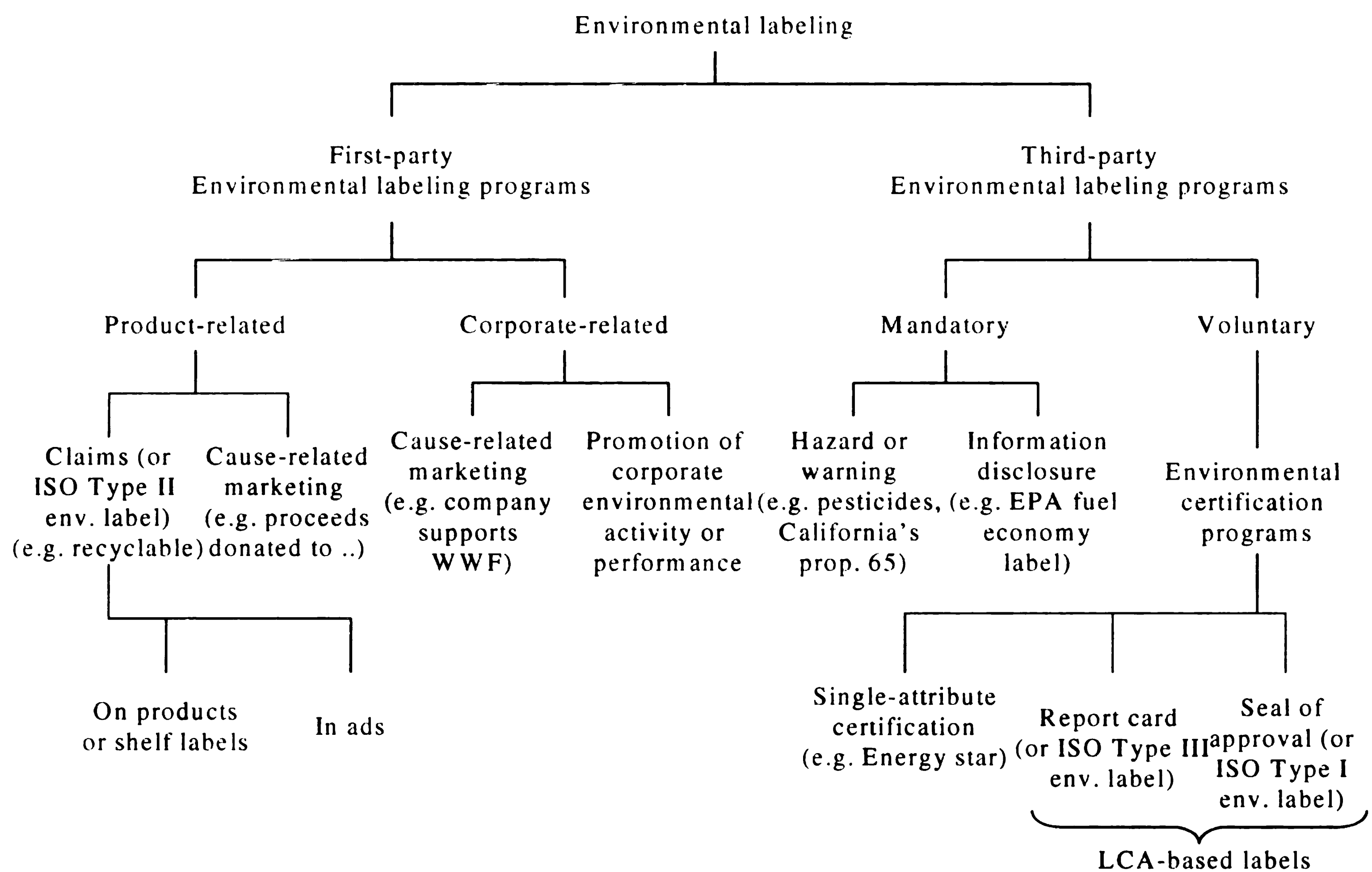


Figure 5. Environmental label classification (based on Environmental Labeling Issues, Policies and Practices Worldwide, United States Environmental Protection Agency (1998))

Though the purpose of these claims is to provide the consumer with accurate information about positive environmental attributes of products and to help with international trade, a couple of issues arise with their use. First, due to economic limitations it is very difficult for an average consumer to challenge or question these statements since tests are often expensive and time consuming. Second, these claims often focus on one stage of the product life cycle, disregarding other stages that potentially may be more harmful to the environment, and thus providing misleading information. In fact, Lavallée and Plouffe (Lavallée and Plouffe, 2004) argue that the widespread use of such labels has hurt the development of LCA-based labels.

Type I and Type III labels are issued by a third party and involve LCA-based analysis for the certification. Type I labels, also known as seal-of-approval, are the outcome of what are usually called eco-labeling programs in which a product, process, or management system is certified to meet specific environmental criteria as established by a third party organization. Oftentimes, manufacturers make prior use of LCA under programs such as DfE that help environmental stewardship, to "self-certify" their processes before third-party validation. The third party can be the owner or administrator of the label program, which usually has three basic steps: (a) selection of product category (e.g. by similar function, and/or similar environmental impacts, and/or importance of product in marketplace); (b) development of requirements to be met (e.g. by using some form of LCA along with peer-review process, selection of the system's most relevant contributions to environmental impacts and guidelines for their reduction are set); and (c) certification and licensing (i.e. compliance verification and testing, applicant licensing and monitoring). Worldwide, eco-label programs vary on how they are run or sponsored. They can be administered by governments, private companies (for profit and non-profit), non-governmental organizations, or some combination of the above.

Though both ISO Type I and Type III environmental labels involve LCA concepts, they differ with respect to the way they convey the information. By assuming that the information from LCA results is too complex and too extensive to present on a label, Type I label programs first decide which stages of the LCA are the most significant for the determination and weighting of the certification criteria, and finally evaluate and issue the seal-of-approval for qualifying products. Alternatively, ISO Type III labels aim at presenting to the consumer much more detailed environmental information, including items such as energy use and environmental impacts in a report-card format, and assume that consumers can themselves prioritize across environmental burden categories and thus themselves do the judgment.

Several issues stir debate and complicate the use and implementation of these environmental labeling programs. For example, while ISO standards require an LCA compliant with ISO 14040, in-depth LCAs are seldom used for awarding these labels because they are cost- and time-intensive. Instead, these programs end up considering only certain stages of the life cycle (Davis et al., 1998), usually by extrapolating from environmental performance results of similar products offered on the market. Furthermore, regarding type III labels, since the selection of labeling criteria is not based on the same LCA methodology, product comparisons cannot be made, thus confusing the consumer at the moment of judging the preference.

Issues may also occur due to the complexities added by global economic trends, trade agreements and logistical practices when considering imported goods and the environmental assessment of their life cycle (Appleton, 1997). Further, due to the nature of the ecolabeling programs and their LCA-based approach, often the programs end up awarding preferability seals to products made with state-of-the-art technologies that are difficult to obtain in less developed regions or countries, thus creating animosity towards the results of these studies (World Trade Organization, Environment: Trade and environment news bulletin, 1996).

Standardization efforts, though very costly and/or technically challenging in some cases, have been made in order to achieve harmonization and/or mutual recognition among programs. In fact in 1994, national and multinational ecolabel licensing programs founded the Global Ecolabeling Network (GEN) with the objective to "improve, promote and develop the ecolabeling of products" (Global Ecolabeling Network, Introduction to ecolabeling, 2004). Currently GEN has twenty-six members with programs such as the well-established Green Seal (U.S.), Blue Angel (Germany), TerraChoice (Canada), European eco-label (E.U.) and Eco Mark (Japan).

Policymaking

The use of LCA for policy making is a practice that sometimes faces strong opposition from trade and industry organizations and even from countries. So far, the governments within the European Union have the most experience in using LCA concepts for developing policies.

The Integrated Product Policy (IPP) developed in 1999 by the European Commission, is a product-oriented approach to government policy that attempts to reduce environmental degradation by addressing all phases of a product's life cycle. The IPP approach uses a number of instruments, such as economic assessments, product stewardship programs, substance bans, voluntary agreements, environmental labeling and product design guidelines, to address the system life cycle impacts of products and processes.

Not surprisingly, though, actual LCA-based policy making, as for any other type of environmental policy, has many critics, and may have economically sensitive consequences for many industries when policymakers consider taxing or restricting what are found to be environmentally unsound products (Europen, Use of LCA as a policy tool, 1999; Europen, Economic instruments in packaging, 2000). For example, many European countries have used some form of LCA to develop federal packaging mandates that require manufacturers to take back packaging discards or pay for their recycling. Germany requires companies that do not participate in its Green Dot program to take back their packaging and pay the cost of recycling it themselves, with no exceptions for foreign companies. This measure has broad implications since the take-back burden is far greater for those companies that ship their products long distances to Germany. Thus several manufacturers exporting to Germany from within the EU and beyond argue that, due to its nature, the Green Dot label program places imported goods at a market disadvantage. Moreover, industry and trade organizations within Europe argue that the degree of diversity between countries and even regions within the same country is so large that the preferred waste management method in one area may not be appropriate for other areas. Thus, these constituencies claim that waste management decisions should be made on a case-by-case basis (Europen, Use of LCA as a policy tool, 1999).

There are no federal packaging mandates of a similar kind in the United States (U.S. EPA, Environmentally Preferable Purchasing, 2005). However, there are a number of federal and state initiatives that involve the use of LCA based tools. For instance, since 1997, the U.S. EPA has been promoting the concept of extended product responsibility (EPR) which is a product-oriented instead of a facility-oriented approach to pollution prevention by using product life cycle concepts (Davis et al., 1997). Within this principle, programs such as Environmentally Preferable Purchasing (EPP) promote the use of LCA-based tools. In fact, originating from executive Order 13101 on "Greening the Government through Waste Prevention, Recycling, and Federal Acquisition", EPP is a nationwide program that uses the leveraging strength of federal buying power as an incentive for industry to develop environmentally preferable products. Guidelines for EPP are developed by the U.S. EPA for use by other federal agencies; however, the program encourages state and local government and the private sector to incorporate environmental considerations into their purchasing processes as well.

Another kind of initiative is the U.S. EPA Design for the Environment Program (U.S. EPA-DfE) (Environmentally Preferable Purchasing, 2005) which is a voluntary government-industry partnership that seeks to incorporate environmental considerations into the design and redesign of products, processes, and technical and management systems.

VIII Outlook and conclusions

The future of LCA for packaging, as for any other product category (e.g. energy, automobiles, appliances), is necessarily linked to the future of LCA and its maturation into a more reliable tool.

Thus, with regard to LCA in general, challenges remain due to the uneven pace at which it has been embraced around the world. In fact, though developed more than thirty years ago in the U.S., the European willingness to incorporate it as a part of their environmental regulatory process is often cited as the reason why Europe leads the way in LCA research (Curran, 1999). On the other end, it is only since the 1990's with ISO's release of its 14000 series of standards on Environmental Management that many developing countries have started to learn about this concept. The delay in coordination (i.e. internationally and nationally) is one reason why a common terminology has been slow to develop, and terms and approaches such as life cycle management or life cycle costing generate confusion. Further, since many LCA studies still remain unpublished or inaccessible, the assimilation of common methodologies is even more difficult. On the other hand, the increasing tendency of the private sector to look at product life cycle concepts and embrace them at the product design phase in order to respond to consumer expectations may be a sign of what is next. As multinational firms extend their operations around the world, along are spread their philosophies and their understanding of LCA concepts. This is why efforts on harmonizing private life-cycle initiatives have started to occur. For example, under the UNEP/SETAC Life Cycle Initiative various workgroups on inventory, impact, and Life Cycle Management (LCM) are trying to achieve this international coordination and discussions have been proposed to adopt LCM as the platform from which to build and execute private environmental stewardship programs (Hunkeler et al., 2001), with an "LCM toolbox" with LCA and LCC as components (Hunkeler et al., 2002; Sonnemann et al., 2001). The open partnership of the private sector with environmental research institutes and regulatory bodies has often been cited to as one of the reasons why many European countries have a healthy LCA activity (Hansen, 1999), and this is a reason why international harmonization measures in the private sector are important. Thus, coordination efforts reflected by the numerous guidelines from SETAC and ISO, and working groups and workshops will need to continue to catalyze the harmonization process and to address current limitations. Furthermore, private and governmental agencies of the U.S. and of European countries will need to continue their development of frameworks and partnerships with industry to help with the goal of making LCA a more useful tool for decision-makers. Current efforts by environmental certification third party organizations in these countries will need to focus on continuing their homologation steps as well as reaching out to similar bodies in developing countries. Likewise, work remains to be done on controversial matters such as data quality and harmonization of inventory procedures and impact assessment methods as well. LCIA improvement will also depend on the success of efforts on modeling the fate of chemicals released into the environment and development of weighting procedures. But perhaps, due to the nature and subjectivity of many of the components of an LCA, several issues will still remain unresolved.

For LCA for packaging, future challenges exist. For example, as packaging remains a necessary item in the market and one of the preferred fields to which LCA is often applied, world population growth and a higher overall quality of living are indicators that packaging waste management options (e.g. reuse, incineration, recycling) will remain issues for further LCAs. Furthermore, optimization in packaging design with regard to the environment will necessarily use life cycle-based methods as expressed by emerging industry groups such as the Sustainable Packaging Council (Johnson, 2005). The consequences of trade agreements and a "global economy" with regard to production and movement of products and their packaging from one region to another and the consequent implications for resource (e.g. material and energy) use and emissions release will also have to be investigated with a "life cycle thinking" philosophy. Lastly, as ever more complex packaging concepts are designed in order to meet increasingly higher consumer expectations (Ver-Bruggen, 2004), complexities in their LCA will require further research. Improvements in radio frequency identification (RFID) technology for product tracking, new dynamic promotion/information capabilities through active labels, product quality sensors, nanoscale optimization of material properties, development of fully biodegradable as well as biodegradable composite materials are some examples of scientific breakthroughs that are starting to be used (Biodegradable Plastic Society, Inventory of GreenPla products, 2004; Lingle, 2004) or are being investigated to build packaging materials and packaging components (Anyadike, 2004; Harrop, 2002) in the future. In turn, inventory databases will need to be developed not only for "conventional" packaging materials and components but for these new packaging concepts as well. Furthermore, since with the use of these new capabilities, new functions would be added to packaging (e.g. degradability, traceability, sensing) besides the traditional containment, the definition of the functional unit for an LCA will be a subject for future debate. All these challenges are indications that LCA research for packaging options should remain a high priority in the future.

CHAPTER 3 COMPARATIVE LCA OF THREE DRINK DELIVERY SYSTEMS

I. Introduction

This chapter presents information regarding a comparative life cycle assessment (LCA) based on a hypothetical drink product for which we evaluate three main container material alternatives: (1) a polyethylene terephthalate (PET) bottle; (2) an aluminum can; and (3) a polylactide (PLA) bottle.

Following ISO's LCA steps, details and information sources used for this comparative LCA are presented. In order to represent realistic implications of the use of the three main material/container alternatives and their actual packaging operations, production and use of distribution packaging (i.e. corrugated trays, stretch wrap, and pallets), as well as transportation steps through several parts of the packaging life cycle were included. Relevant end-of-life scenarios such as landfilling, incineration and recycling were also taken into account in this analysis. Each of the main material alternatives (i.e. PET, PLA, aluminum) along with its whole set of operations was defined as a product delivery system (PDS) and environmental burdens associated with each of the three PDSs were included in the study.

The calculation model of the present study was based on previous comparative packaging based LCAs published elsewhere (Keoleian, 2001; Keoleian et al., 2004; Sellers and Sellers, 1989). The data used in this study come from a number of publicly available sources and can be divided into background and foreground categories. Background data comprise the life cycle inventory data associated with all the activities and processes involved in the drink delivery systems. This type of information was obtained from the Data for Environmental Analysis and Management - DEAM™ - database developed by the Ecobilan Consulting Group (Paris, France) (DEAM module databases and manuals, 1999). The DEAM™ database represents industry activities by "modules" whose inventory data in turn were gathered from a number of different sources (e.g. official reports, surveys, engineering estimates, etc.). Each module is, in fact, the result of a "cradle to gate" inventory analysis. With this convention, then, a complete product life cycle (i.e. cradle to grave) inventory analysis can be modeled by appropriately linking several of these modules. The linking of these modules requires the knowledge of foreground data. Foreground data comprise data specific to a certain product and its requirements (electricity requirement for PET injection stretch blow molding, etc.). Foreground data were obtained from a number of different sources, and for organization purposes their details are provided in later sections of this chapter.

Regarding this study's impact assessment component, energy, water use and two characterized impacts are included: global warming potential (GWP) and ozone depletion potential (ODP). Other impacts were not included since we restricted the focus of our analysis of the comparative LCA results to packaging-related causes and effects of uncertainty in the inventory data. Thus, as explained in Chapter Two and aware of further uncertainty considerations in the impact assessment methodologies of other impact indicators (e.g. natural resource use), we opted for using GWP and ODP as the two most widely accepted, least uncertain impact indicators, according to the LCA community.

For organization purposes, this whole chapter is devoted to presenting the information used in the comparative LCA study as transparently and openly as possible and clearly giving specifics on the LCA scope, data origin, and the systems considered.

Actual results of the LCA are in the next chapter, Chapter Four, along with the actual uncertainty estimation and the discussion of its results.

The following sections of this chapter explain the methods and present the details of the LCA. In particular, Section II discusses the methodology for the study, including the goal and scope of the analysis. Details on the definition of the functional unit and product delivery systems are provided. Specifics about the system model and its boundaries, data categories and impact categories are also included. Later in this section, information regarding the life cycle inventory analysis itself is presented, including specifics on material allocations, system diagrams, data sources, calculation procedures, assumptions and limitations of the analysis.

II. Methodology

a) Goal and scope definition

In this section, the goal and scope of the analysis is presented. Details on the purpose of the study, definition of the functional unit, and definition of the product delivery systems are provided. An overview of the basic PDS model and its boundaries, accounted data categories and impact categories is also included.

Purpose of the study

The primary purpose of this study is to provide an LCA model to evaluate robustness of the results when subject to scenario and parameter uncertainty. This study also provides information to increase the knowledge of some of the potential environmental impacts associated with the life cycle of a hypothetical product delivery system considering three packaging material alternatives.

The alternative product delivery systems modeled in this study are:

- PDS 1: utilizing PET bottles and PP caps as one choice for drink delivery.
- PDS 2: utilizing aluminum cans as another choice for drink delivery.
- PDS 3: utilizing PLA bottles and PP caps to investigate the use of renewable material feedstocks.

The findings in this study are intended for research purposes. In particular, the information in this study is expected to give insights on the behavior of a packaging-based LCA outcome. Though the findings of this work may be useful, due to the streamlined nature of this study in particular, and the intrinsic limitations of LCA in general, it should not be considered as the only source of information regarding the environmental performance of the materials investigated or the product delivery system considered.

Function and functional unit

The function of the system was the delivery of a hypothetical drink to the market. In this study, market is defined as the first destination after the product filling stage. The functional unit was 1000 liters of drink delivered to the market (distributor or retailer).

Product delivery systems (PDS)

The PDSs were defined as the systems employed for the distribution of the hypothetical product to customers. They included the primary and distribution packaging, as well as all manufacturing, and transportation associated with the function of the system.

- Primary packaging: The hypothetical container formats, materials and sizes considered in this study are listed in Table 11. The plastic bottles under consideration were assumed to be injection stretch blow molded and the aluminum can was assumed to be a draw and iron two-piece container.

Table 11. Primary packaging characteristics

Format	Materials	Individual size, fl. Oz.	Number of containers per tray
Bottle/sleeve/cap/carrier ring	PET/LDPE/PP/LDPE	16	24
Bottle/sleeve/cap/carrier ring	PLA/PLA/PP/LDPE	16	24
Two-piece can/carrier ring	Aluminum/LDPE	12	24

As is a common practice, all formats use LDPE ring carriers and paperboard trays. Inks used in primary packaging or related printing processes were not included in the study.

Distribution packaging: This is defined as the components of a PDS used for the delivery of materials for the manufacture of containers, and the materials used to protect the primary packaging during the transportation to and from the drink filling facility. Specifically, the distribution packaging comprised corrugated boxes, stretch wrap and pallets.

Transportation: The shipping mode considered in the PDSs is truck.

Location and time considerations: The operations involved in the product delivery process are assumed to be current and to happen in the U.S.

System model

The system model of the LCA divides the PDS into phases (see Figure 6). Each PDS was represented as a nine-phase system with phases labeled as follows: material production, distribution 1, container component manufacturing, distribution 2, filling, distribution 3, wholesale retailer, product consumption and end-of-life.

Though identified in the PDS model, the wholesale retailer and product consumption phases were not included in the life cycle analysis (i.e. their associated environmental burdens were not compiled) because emissions and energy use data about these operations were not readily available. Furthermore, it was reasoned that the omission of these operations would not impact or change the characteristics of the LCA since its contribution was assumed negligible (e.g. when compared to the material production phase).

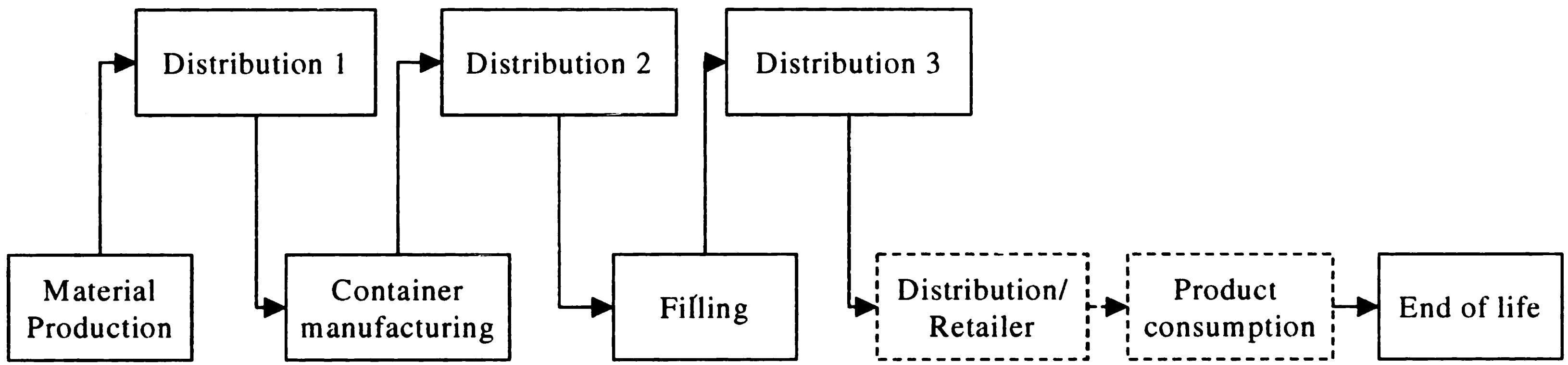


Figure 6. Life cycle phases considered in the study

System boundaries

The boundaries of the system are presented next, following the order of the phases of the PDS according to Figure 6.

- Material Production: The Material Production phase includes the extraction of the raw materials, the materials manufacture (i.e. the processing of the raw materials into intermediate materials), and the corresponding transportation of raw materials to a location where they are processed into intermediate materials. Material Production includes only production of materials used in the primary packaging. Material production of distribution packaging is included in the Distribution phases. Additives (e.g. plasticizers, antioxidants) and color concentrates (i.e. pigments and carrier resin) that the PET, PP, LDPE and PLA materials used for the primary packaging may contain usually represent a very small percentage of the plastic mass and thus their associated production burdens were excluded from the system model. As a result, the materials were modeled as 100% plastic resin. The aluminum for the aluminum can was also assumed 100% pure.
- Distribution 1: Distribution 1 includes the transportation of primary packaging input materials from the location where materials are produced to the container component manufacturing location. Material production and manufacturing of the distribution packaging used to ship materials between the Material Production phase and the Manufacturing phase were considered to be outside the boundaries of this study. Likewise, since most of the shipments in this phase utilize bulk trucks, distribution packaging weight was not included.

- Container component manufacture: this phase includes only the manufacture of the primary packaging. Manufacturing of distribution packaging was included in the Distribution 2 and 3 phases. Burdens associated with the production of the manufacturing equipment are excluded from the system model. These burdens were expected to be small relative to the burdens associated with the processes that were inventoried. The one exception was that road transport in many of the published data modules contained the burdens associated with the construction of the vehicles and replacement parts such as tires and batteries. Burdens associated with the construction of the manufacturing facilities are excluded from the system model. Burdens associated with human activities, including driving to and from work, are excluded from the system model. Printing inks are excluded from the system model due to lack of data.
- Distribution 2: this phase includes the transportation of primary packaging materials from the manufacturing location to the filling facility. The environmental burdens associated with material production and manufacture of the distribution packaging used during this transportation were included. The burdens associated with the transportation of the distribution packaging from the supplier to the manufacturing facility were also included. Materials required for the shipment of the distribution packaging were considered to be outside the boundaries of this study because emissions and energy use data about these operations were not readily available. Furthermore, it was reasoned that the omission of these operations would not impact or change the characteristics of the

LCA since its contribution was assumed negligible (e.g. when compared to the material production phase)

- Filling: Energy and emissions burdens associated with the filling process were excluded from the system model. It was assumed that the facility in which the filling occurs is also used for the drink production and therefore burdens associated with the PDS are difficult to distinguish from the burdens associated with drink production. For this reason, Filling was determined to be outside the scope of the study. The only exceptions were two: a) the inclusion of the injection stretch blow molding energy (i.e. in the form electricity) prior to the filling in the case of bottles; and b) the solid waste generated from the losses of primary packaging during the filling/packing processes, which were directly related to the PDS and therefore included.
- Distribution 3: This phase includes the transportation of the drink and its primary and distribution packaging from the filling location to the distributors/retailers. The environmental burdens associated with material production and manufacturing of the distribution packaging used during this transportation are included, as well as the burdens associated with the transportation of the distribution packaging from the supplier to the filling facility. Materials required for the shipment of the distribution packaging were considered to be outside the boundaries of this study. Distribution 3 includes only the burdens associated with transportation to the first destination.
- Wholesale retailer: Environmental burdens associated with activities in the Distributor/Retailer phase were excluded from the system model. Disposal of

distribution packaging used in the Distribution 3 phase is counted in the Distribution 3 phase.

- Drink consumption: Environmental burdens associated with the transportation of the product from the retailer to the consumer and consumption of the drink were excluded from the system model.
- End-of-life: Environmental burdens associated with the End-of-Life phase include burdens of landfilling, incineration and recycling operations associated with the main material alternatives (i.e. PET, PLA and aluminum).

Data categories

Four data categories were used to classify the items inventoried throughout the study. The data categories were intended to reflect the emissions or resource use for each area of interest. The data categories and their components can help make overview statements pertaining to the environmental impacts of the PDS.

- Energy: The study tracks total primary energy consumed at each life cycle phase. The DEAM™ database further decomposes primary energy into Renewable, Non-Renewable, and Feedstock energy. Renewable and Non-Renewable refer to energy generated from renewable fuels (hydroelectricity, wood and biomass) and non-renewable fuels (coal, lignite, oil, natural gas or uranium) respectively. Feedstock energy is the part of total primary energy that is embedded within used material such as combustible fuel material. This level of detail was maintained in this study and will be referred to in the discussion of results section.
- Water use: The data category for total water use referred to the tracking and aggregation of water used at each phase of the life cycle, measured by volume.

Impact categories

Two characterized impact categories were analyzed:

- Global warming potential: Global Warming Potential is the category indicator measuring possible contribution by the PDS to the "greenhouse effect." The greenhouse effect refers to the ability of some atmospheric gases to retain heat that is radiating from the earth. The measurement of GWP employed in this report is based on the model compiled by The Intergovernmental Panel on Climate Change (IPCC) (1995). The GWP index is defined as the cumulative radiative effect between the present and a chosen time horizon (this study arbitrarily chose 20 years) caused by a unit mass of emitted gas, expressed relative to that for some reference gas (IPCC and this study use CO_2). A single indicator is produced for the greenhouse effect, in which:

$$E = \sum GWP_i * m_i$$

where, for a greenhouse gas "i", m_i is the mass of the gas released (in grams), and GWP_i is its Global Warming Potential, expressed in CO_2 equivalents. Although the GWP calculation is one of the most accepted of all life cycle impact assessment index methodologies (Udo de Haes, 2002), it is limited since GWP assumes an even distribution of the gases being tracked, and indirect GWP effects, such as the emission of one greenhouse gas leading to the formation of another greenhouse gas, are not considered.

- Ozone depletion potential: Deterioration of the ozone layer allows more radiation to reach the Earth's surface, potentially destabilizing ecosystems as well as causing adverse effects on agricultural productivity, human health and climate.

ODP is an impact category measuring possible contribution by the PDS to deterioration of the stratospheric ozone layer. The most accepted ODP model, developed by the World Meteorological Organization (Scientific assessment of ozone depletion, 1998), is employed in this study. The impact category is evaluated as follows:

$$\text{Ozone Depletion} = \sum ODP_i * m_i.$$

where, for an ozone depleting gas "i", m_i is the mass of the gas released (in milligrams), and ODP_i is its Ozone Depletion Potential. Ozone depletion is expressed in milligrams of CFC-11.

b) Life cycle inventory

In this section information and data sources associated with the life cycle inventory analysis of the three drink delivery systems is presented including detailed information on the systems components, material allocations, systems diagrams, calculation procedures and assumptions and limitations. The data used in this study come from a number of publicly available sources and can be divided into background and foreground categories. Background data comprise the life cycle inventory data associated with all the activities and processes involved in the drink delivery systems. This type of information was obtained from the DEAM™ database developed by the Ecobilan Consulting Group (Paris, France)(DEAM module databases and manuals, 1999). The DEAM™ database represents industry activities by "modules" whose inventory data in turn were gathered from a number of different sources (e.g. official reports, surveys, engineering estimates, etc.). Each module is in fact the result of a "cradle to gate" inventory analysis and includes inputs (e.g. bauxite from ground, water used, etc.) and outputs (e.g. air, water and soil emissions) associated to the provision of a certain intermediate product (e.g. PET resin production, aluminum ingot production, etc.) or service (e.g. energy provision by the U.S electricity grid, transportation service of a 40-ton truck, etc.). A product life cycle (i.e. cradle to grave) inventory analysis was modeled by appropriately linking several of these modules. The linking of these modules was made by using foreground data. Foreground data comprise data specific to a certain product and its requirements (e.g. weight of a PET bottle, electricity requirement for PET injection stretch blow molding, materials required for corn farming in PLA production, etc.). Foreground data were obtained from a number of different sources and are detailed in every case when they were used. Foreground data in this study were used provided they were readily available (e.g. literature estimates, previous LCAs, etc.) and from a reference source.

Additionally, besides foreground and background data, there are input data. Input data were measured (e.g. weight of a PET and PLA bottles, aluminum can), estimated using software and/or literature data for average operations (e.g. distribution packaging formats and weights), or arbitrarily set (for all PDSs equal) with the spirit to represent a relatively average operation involving drink products (e.g. transport distances).

DEAM™ modules are designed to be viewed with a spreadsheet like interface. In this study all calculations and system representations were done by appropriately linking and formatting Microsoft Excel spreadsheets (Microsoft Excel, 2000) containing the modules.

System

Material Production: The Material Production phase represents the material production activities for primary packaging. The primary packaging materials were detailed in Table 1.

- Polyolefins (PP, LDPE): The life cycle processes of the production of polyolefin resins are illustrated in Figure 7 below. The production of these materials includes Oil and Natural Gas Extraction, Petroleum Refining, Cracking of naphtha, and Polymerization. The difference between PP production and LDPE production is the intermediate produced in the cracking process. The interim monomer ethylene (C_2H_4) is polymerized into polyethylene and propylene C_3H_6) is polymerized into polypropylene.

  The detailed system model of the material production process depends on the data modules obtained from secondary data sources. The following Table 12 contains the sources and brief explanation about the data modules used in this study.

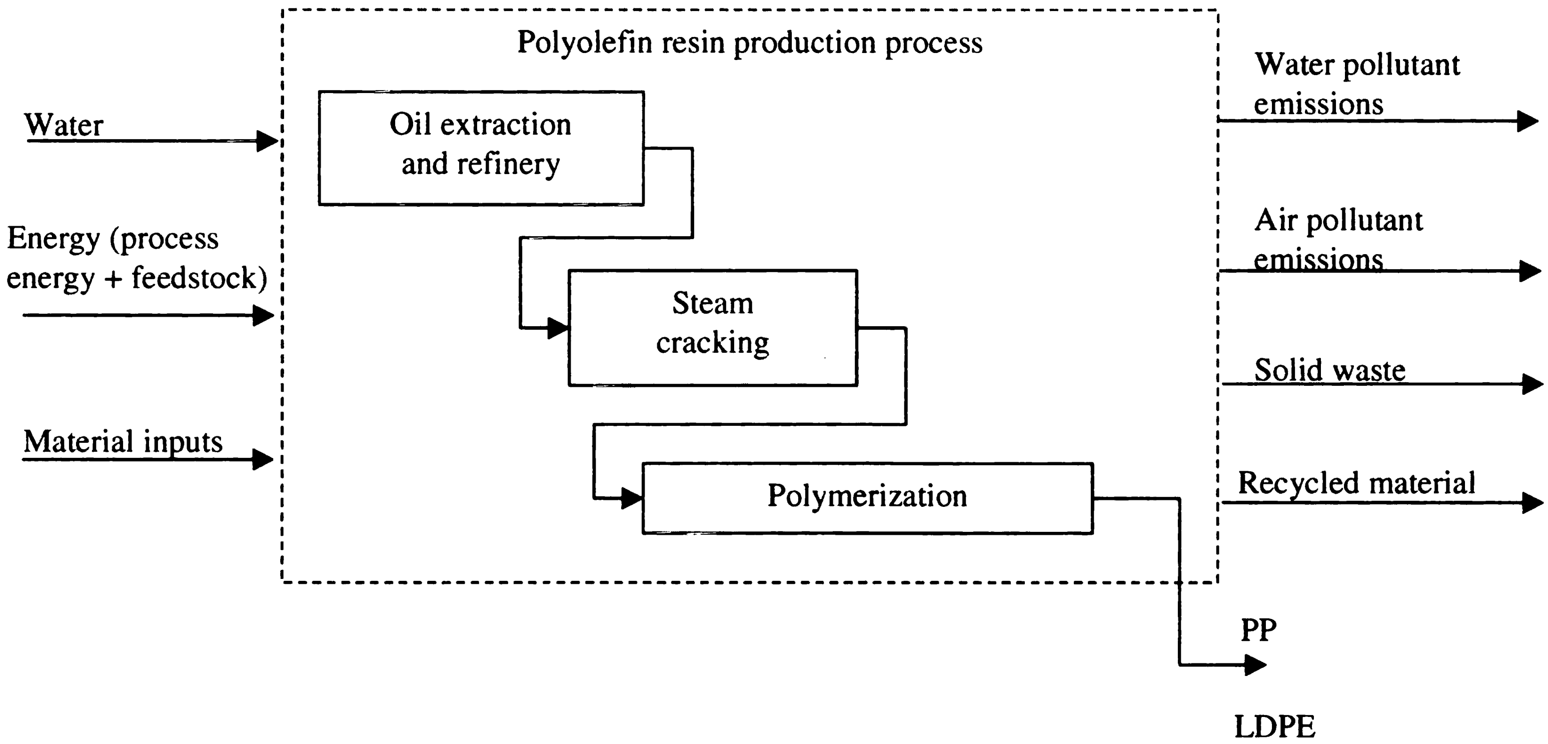


Figure 7. Polyolefin resin material production system

Table 12. Polyolefin resin production system data sources

Module	Source
PP resin production	Background data: DEAM^{TM} module name: Polypropylene: Production.1
LDPE resin production	Background data: DEAM^{TM} module name: Low Density Polyethylene (LDPE): Production.1

- Polyethylene terephthalate (PET): As shown in Figure 8, the production subsystem of PET includes an oxidization and hydration processes and a poly-condensation process, instead of the polymerization process shown in the PP and PE production process. Table 13 contains the sources and explanation about the data modules used in this study.

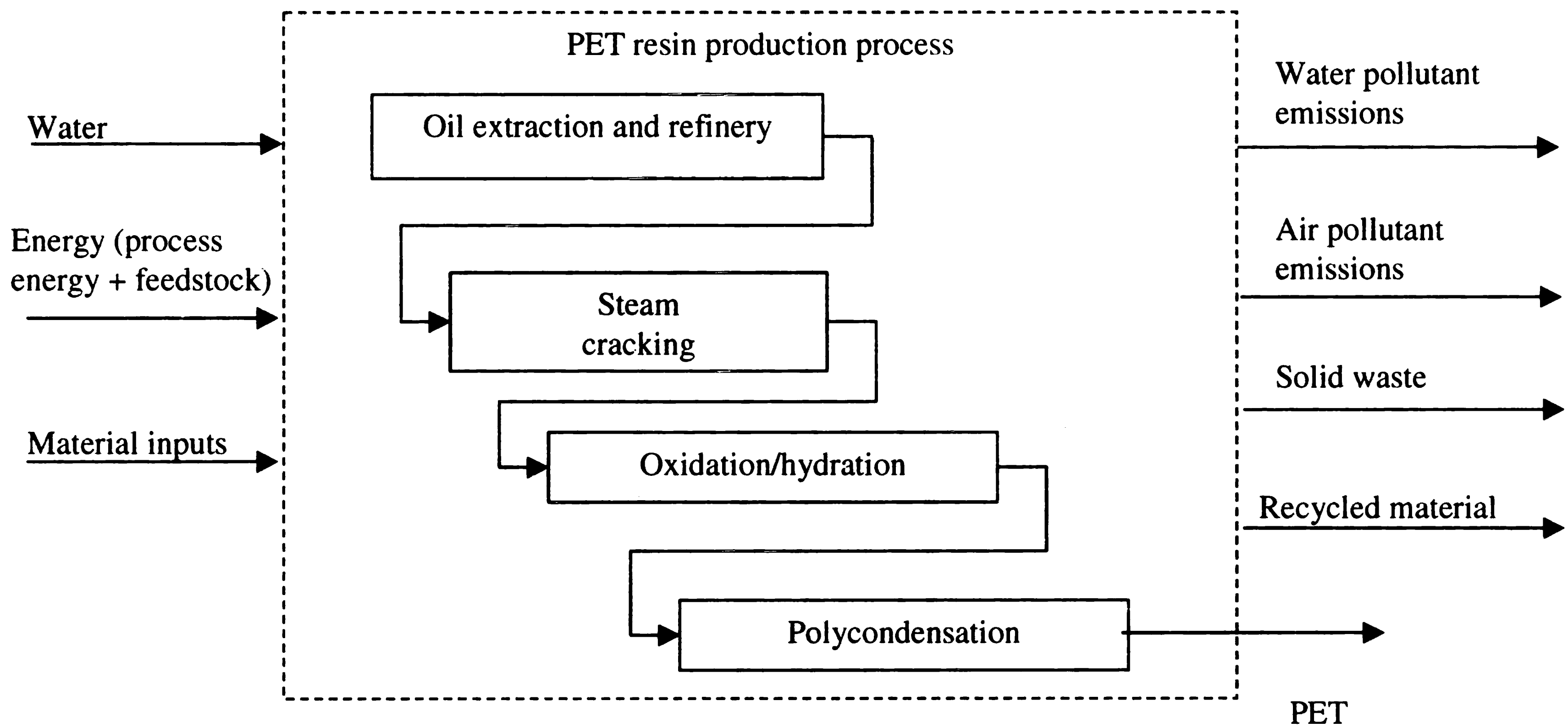


Figure 8. PET resin material production system

Table 13. PET resin material production data source

Module	Source
PET resin production	Background data: DEAM™ module name: Polyethylene Terephthalate (PET, Resin): Production.1

- Polylactide (PLA): As shown in Figure 9, the production system of PLA includes corn production, wet milling, dextrose conversion into lactic acid, purification and polymerization processes. Table 14 contains the sources and explanation about the data used for this module.

Table 14. PLA resin material production data source

Module	Source
PLA resin production	Foreground data: Corn farming and corn wet milling: (Gerngross, 1999; Shapouri et al., 2002) Polymerization: (Gerngross, 2000; Vink et al., 2003). Background data: Corn farming and corn wet milling: DEAM™ module names 1. Limestone (US, $CaCO_3$): Quarrying.1 2. Nitrogen (US, N_2): Production.2 3. Potash (KCl): Production.1 4. Superphosphate (Normal): Production.1 5. Natural Gas (US): Production.2 6. Natural Gas (US, Industrial Boiler): Combustion.2 7. Petrochemical Feedstocks (US): Production.2 8. Electricity (US, average): Production.2 9. Diesel Oil (US): Production.2 10. Diesel Oil (US, tractor): Engine combustion.2

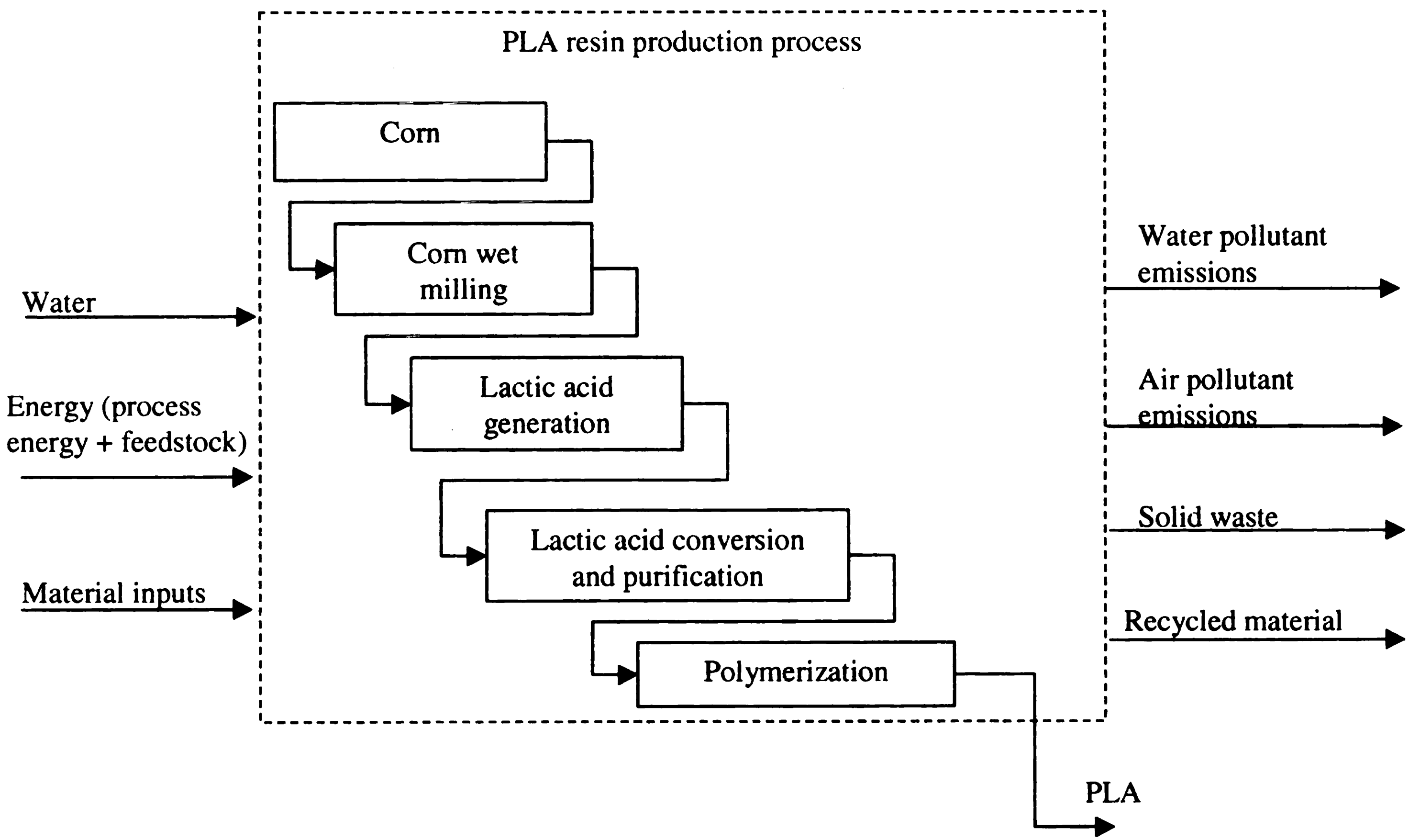


Figure 9. PLA resin material production system

- Aluminum: As shown in Figure 10, a standard practice to represent (in prepared databases) the production system of aluminum cans is by "mixing" in a linear fashion virgin and recycled-based aluminum production systems (Norris, 2003). For the virgin system, it was assumed that the bauxite mining and alumina production (which often happen near the mines) was done outside the U.S. since in the U.S. almost one half of aluminum products are made from imported alumina (Australia and Jamaica account for most of the imported alumina). Under these assumptions, then, the electricity requirements of the DEAM™ alumina production module was linked to the corresponding burdens related to the Australian electricity grid (Ecobilan Group Inc., DEAM module databases and manuals, 1999; Norris, 2003). However, transportation of alumina from Australia was considered outside the scope of this study. The smelting of the alumina to produce aluminum metal was assumed to happen in the US, so the DEAM™ U.S.-based aluminum ingot production module was used. To represent the burdens of the production based on recycled aluminum the DEAM™ U.S.-based recycled ingot production module was used. The aluminum ingots are next transformed into sheet material for later container manufacture process. Table 15 contains the sources and brief explanation about the data used for this module. The inventory data for ingot casting in the U.S. was calculated by subtracting the burdens from alumina production and the electricity requirement from the U.S. electricity grid from the DEAM™ U.S. ingot-production module. The burden data for aluminum for the manufacture of aluminum sheets was obtained by subtracting the DEAM™ aluminum ingot production module from the DEAM™ aluminum sheet production module.

Table 15. Aluminum material production system data source

Module	Source
Aluminum roll production	Foreground data: Alumina requirements for aluminum production (Life cycle assessment of aluminum, LCA of aluminum, 2003; Norris, 2003) Background data: DEAM™ module name 1. Aluminum (Al, ingot) production 2. Aluminum (Al, sheet) production 3. Aluminum (US, 100% primary, ingot) production 4. Aluminum (US, 100% recycled, ingot) production

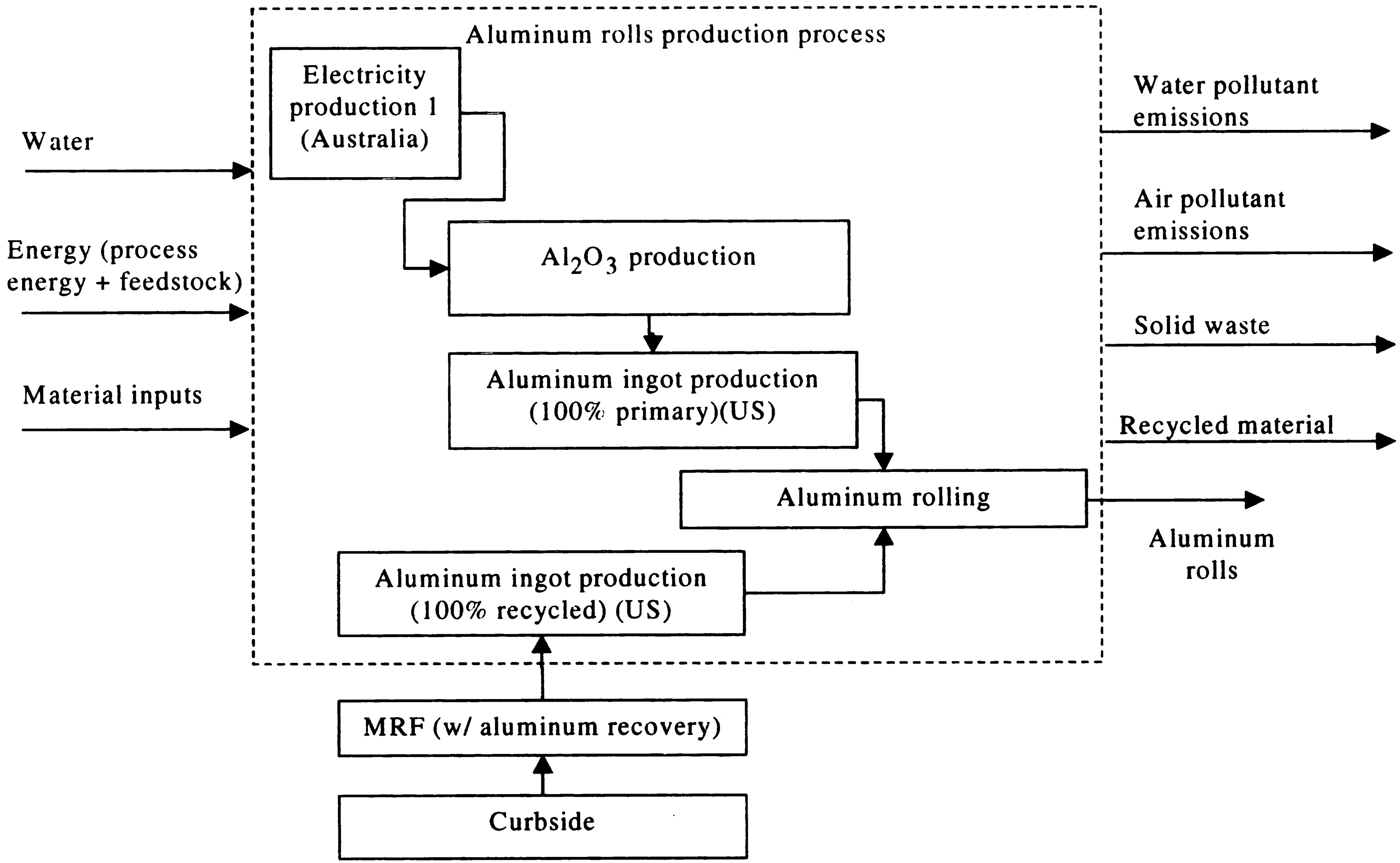


Figure 10. Aluminum material production system

Distribution 1: the Distribution 1 phase includes environmental burdens associated with transportation of materials used for the manufacturing of the primary packaging. The extent of Distribution 1 was the transportation between material producers and the respective container component manufacturers. The burdens of fuel production and fuel use were taken into consideration. The transport load is the sum of the weight of materials for manufacturing of primary packaging (drink container) and the weight of distribution packaging used for the transportation of these materials. Distribution 1 does not include the production of the distribution packaging, since it is a common practice to ship the materials for manufacturing in bulk and thus the use of non-reusable distribution packaging is avoided. Each material was assumed to be transported from the producer to the container manufacturer by truck. The same arbitrary distance was used for all PDSs and the value are presented in the Assumptions section of this Chapter. Any waste associated with the losses of carried materials during transportation was assumed to be landfilled at a rate of 100%. The inventory model for truck transportation includes both the production of fuel and the use of fuel for transportation. Table 16 includes descriptions of the truck model used for the distribution 1 phase in this study.

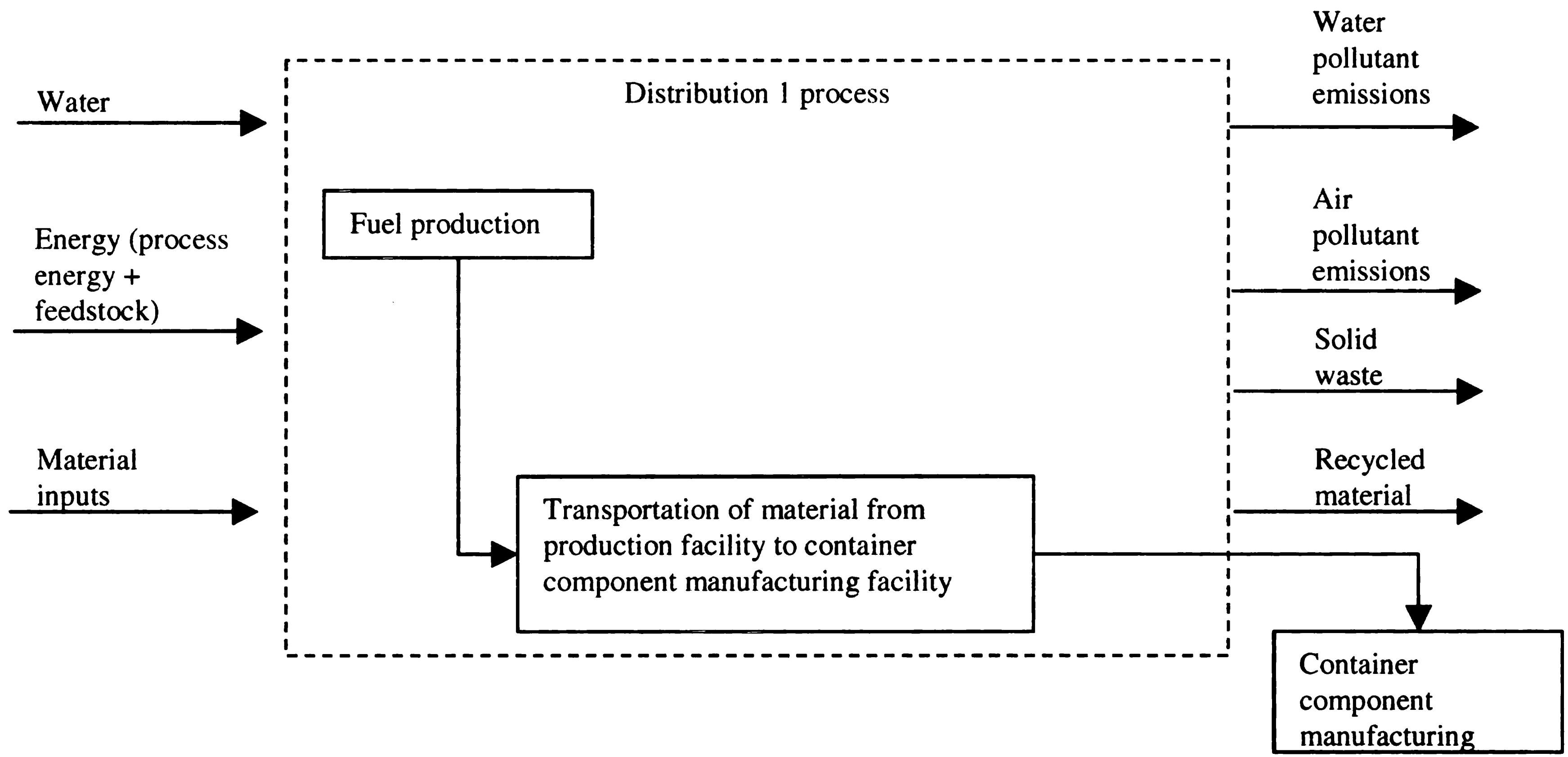


Figure 11. Distribution 1 system

Table 16. Distribution 1 system data sources

Module	Source
Truck transportation	Background data: DEAM^{TM} module names 1. Road Transport (Truck 40 t, Diesel Oil, kg.km).1 2. Diesel Oil (US): Production.2

Manufacturing: Energy requirements for the different manufacturing processes were taken into account. The energy requirements were met by the US electricity grid. Four models for the Manufacturing phase were created. The first three models are for the production of the plastic container components: injection molding, which is used to manufacture caps, injection stretch blow molding, which is used to manufacture bottles and extrusion/lamination which is used to manufacture sleeves and carrier rings. The other model is for the manufacture of the aluminum can body and end. Figure 12 and Figure 13 show the manufacturing operations involved in the different PDS. Following the illustrations, there is a detailed description of each model.

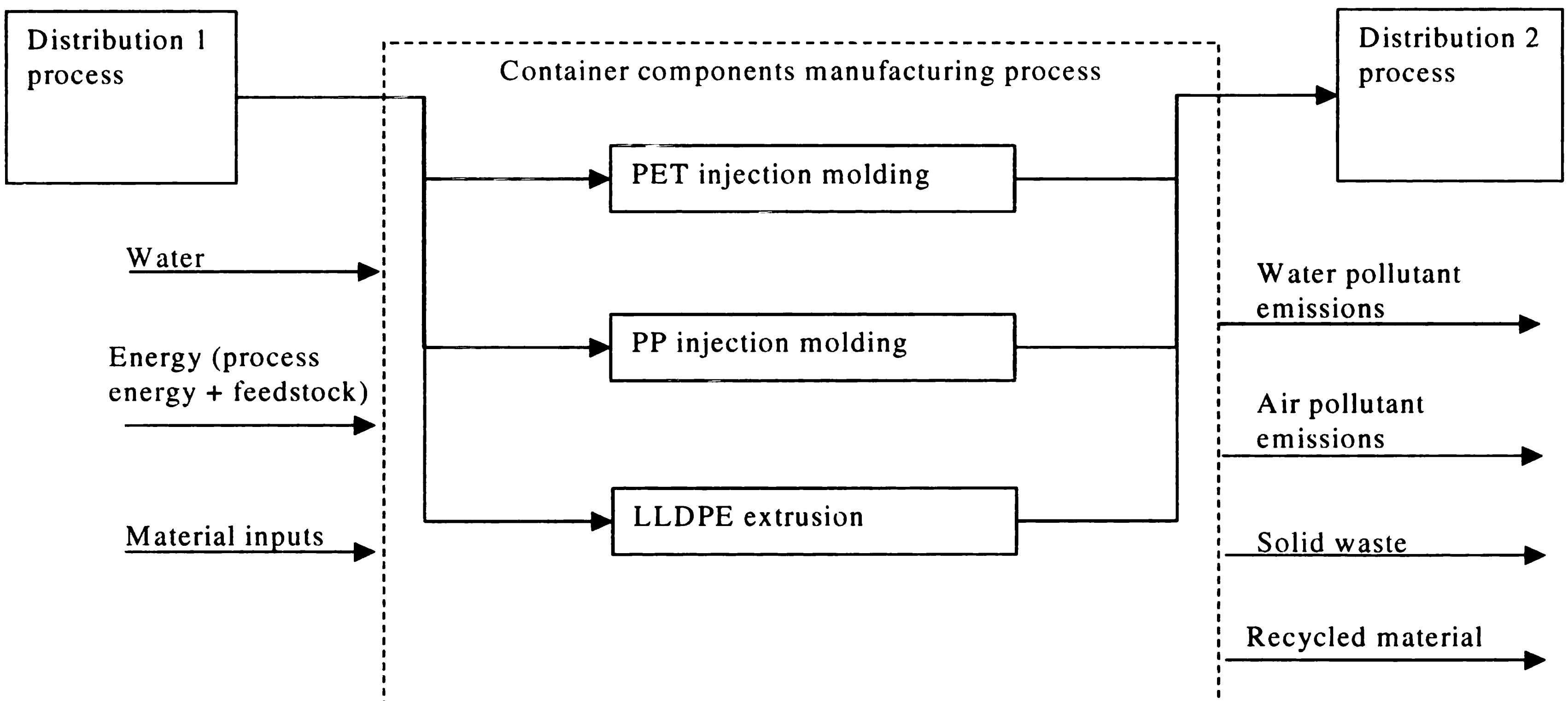


Figure 12 Container component manufacture process of PDS 1 and PDS 3

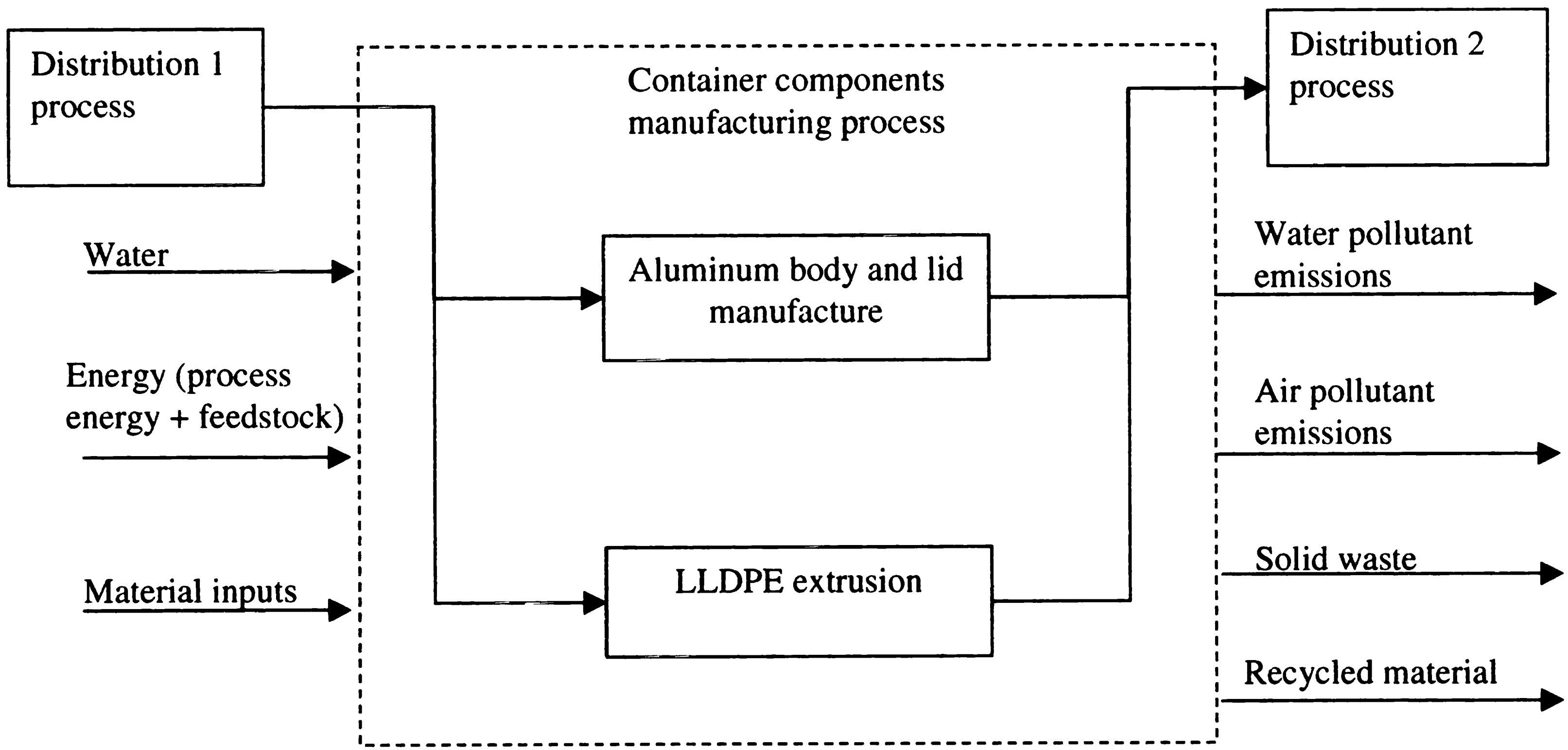


Figure 13. Container component manufacture processes of PDS 2

- Film Extrusion: as shown in Table 17, the process requirements for blown film extrusion process were obtained from the SAEFL study (LCI for packagings. Vol. II., 1998b). The published energy requirements (i.e. electricity) for Swiss plants was converted into U.S. environmental burdens using U.S. specific DEAM™ data modules.

Table 17. Blown film extrusion process data source

Module	Source
Blown film extrusion	Foreground data: SAEFL report: Blown film extrusion energy requirements (LCI for packagings. Vol. II., 1998b) Background data: DEAM™ module name: Electricity (US, average)

- Injection molding: As shown in Table 18, process requirements for the film extrusion process were obtained from the SAEFL study (LCI for packagings. Vol. II., 1998b). The published energy requirements (i.e. electricity) for Swiss plants were converted into U.S. environmental burdens using U.S. specific DEAM™ data modules.

Table 18. Injection molding process data source

Module	Source
Injection molding	Foreground data: SAEFL report: Injection molding energy requirement (LCI for packagings. Vol. II., 1998b) Background data: DEAM™ module name: Electricity (US, average)

- Can body and lid manufacture: process requirements were obtained from a Mitsubishi Materials Corporation LCA study. The published energy requirements (i.e. electricity) for a Japanese aluminum container plant (Mitsubishi Materials Co., LCA of aluminum can, 2003) were converted into U.S. environmental burdens using U.S. specific DEAM™ data modules. More details are contained in Table 19.

Table 19. Can body and lid making processes data source

Module	Source
Can body and lid making	Foreground data: Mitsubishi Materials Corporation – Registration number: S-EP00022 – 1 to 12, Energy requirements for can making (no. 8) (LCA of aluminum can, 2003) Background data: DEAM™ module name: Electricity (US, average)

Distribution 2: the Distribution 2 model includes environmental burdens associated with transportation of manufactured container components to the filling facility. The burdens of fuel production and fuel use were taken into consideration. The transport load is the sum of the weight of components used in the filling process and the weight of distribution packaging used for the transportation of these materials. Distribution 2 and 3 include the production of the distribution packaging. Each component was assumed to be transported from the producer to the container manufacturer by truck. Any waste associated with the losses of carried materials during transportation was assumed to be landfilled at a rate of 100%. The inventory model for truck transportation includes both the production of fuel and the use of fuel for

transportation. Table 20 and Table 21 include descriptions of the truck and distribution packaging model used for the distribution 1 phase in this study.

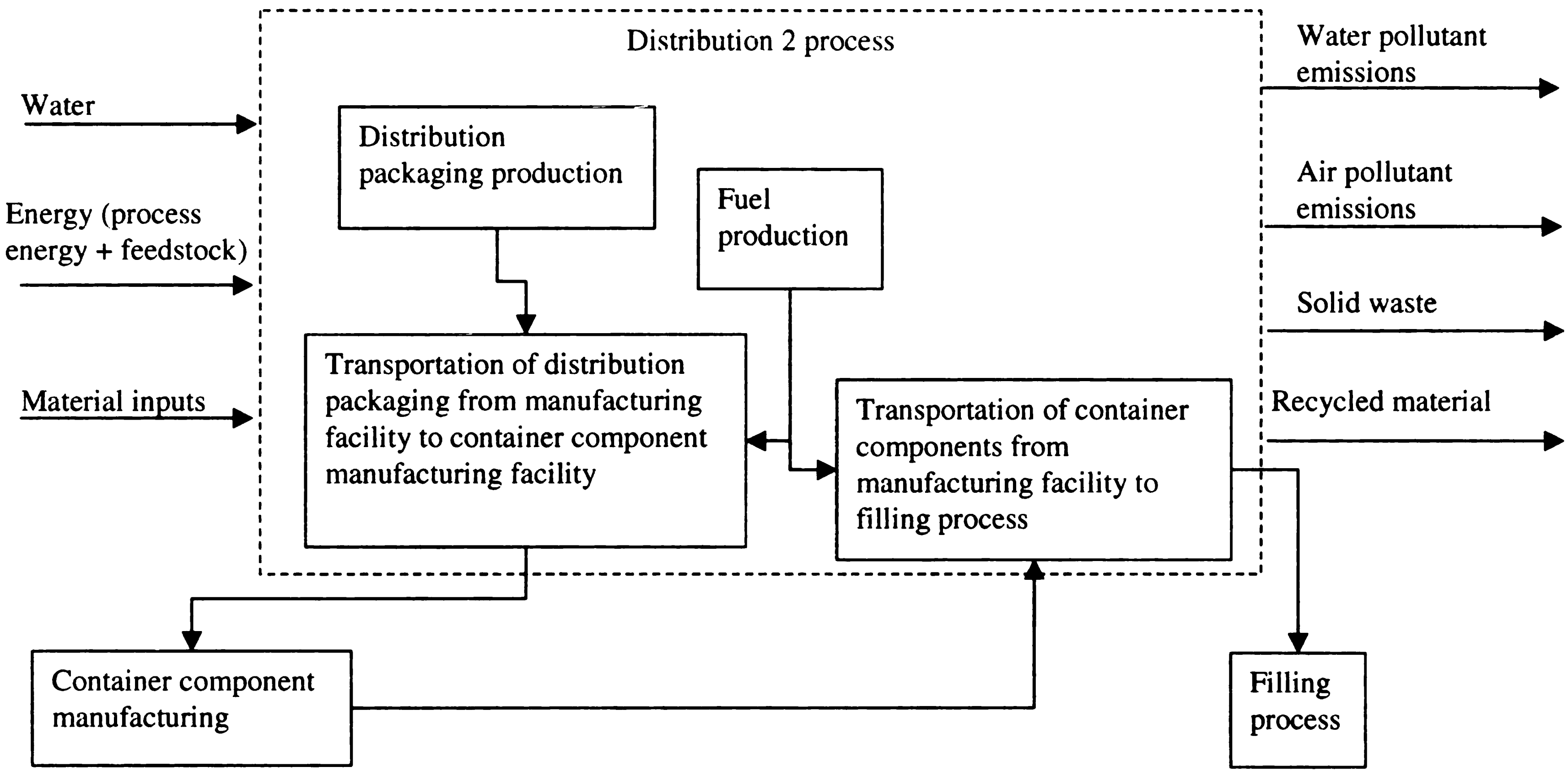


Figure 14. Distribution 2 process

Table 20. Distribution 2 process data source

Module	Source
Truck transportation	Background data: DEAM^{TM} module names 1. Road Transport (Truck 40 t, Diesel Oil, kg.km).1 2. Diesel Oil (US): Production.2

Table 21. Distribution 2 process data source - Distribution packaging modules

Module	Source
Corrugated board	Background data: DEAM^{TM} module name: Corrugated Cardboard (Recycled Fibers): Production.1
Stretch wrap	Background data: DEAM^{TM} module name: Low Density Polyethylene (LDPE, Film): Production.1
Wood	Background data: LCI of kiln-dried lumber (WWPA, LCI of Western Wood Prod. Assoc., 1995)

Filling: Injection stretch blow molding: process requirements for injection stretch molding process were obtained from the SAEFL study (LCI for packagings. Vol. II., 1998b). The published energy requirement (i.e. electricity) for Swiss plants was converted into U.S. environmental burdens using U.S. specific DEAM^{TM} data modules. More details are in Table 22.

Table 22. Injection stretch blow molding process data source

Module	Source
Injection stretch blow molding	Foreground data: SAEFL report: Injection stretch blow molding energy requirement (LCI for packagings. Vol. II., 1998b) Background data: DEAM™ module name: Electricity (US, average)

Distribution 3: Distribution 3 model includes environmental burdens associated with transportation of the drink container from the filling facility to the distribution/retailer. The burdens of fuel production and fuel use were taken into consideration. The transport load is the sum of the weight of components used in the filling process and the weight of distribution packaging used for the transportation of these materials. Distribution 3 includes the production of the distribution packaging. Each component was assumed to be transported from the producer to the container manufacturer by truck. The same arbitrary distance was used for all PDSs and the value are presented in the Assumptions section of this Chapter. Any waste associated with the losses of carried materials during transportation was assumed to be 100% landfilled. The inventory model for truck transportation includes both the production of fuel and the use of fuel for transportation. Table 23 and Table 24 include descriptions of the truck and distribution packaging model used for the distribution 1 phase in this study.

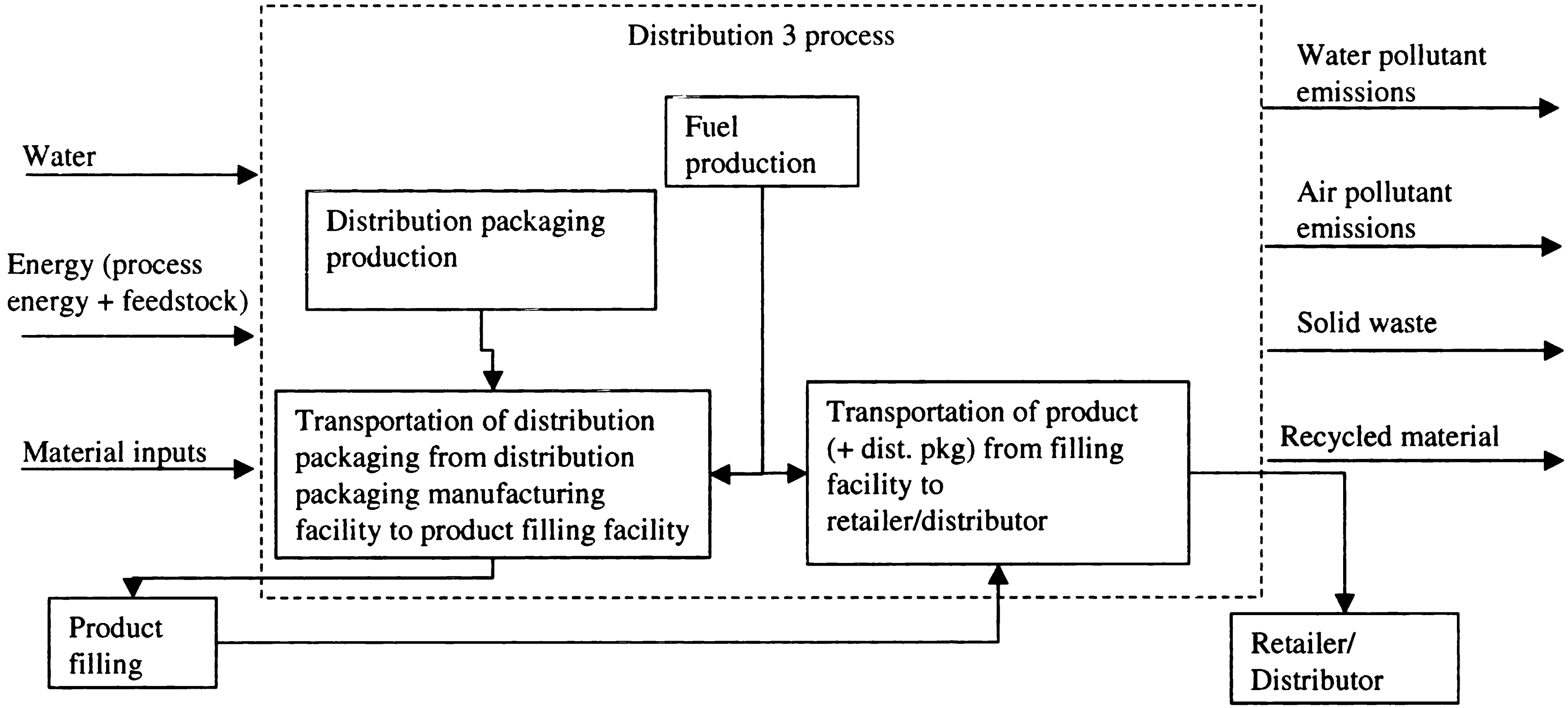


Figure 15. Distribution 3 process

Table 23. Distribution 3 process data source – Transportation modules

Module	Source
Truck transportation	Background data: DEAM™ module names 1. Road Transport (Truck 40 t, Diesel Oil, kg.km).1 2. Diesel Oil (US): Production.2

Table 24. Distribution 3 process data source - Distribution packaging modules

Module	Source
Corrugated board	Background data: DEAM™ module name: Corrugated Cardboard (Recycled Fibers): Production.1
Stretch wrap	Background data: DEAM™ module name: Low Density Polyethylene (LDPE, Film): Production.1
Wood	Background data: LCI of kiln-dried lumber (WWPA, LCI of Western Wood Prod. Assoc., 1995)

End-of-life: End-of-Life process modeling accounted for the environmental burdens that stemmed from waste treatment of used containers. It was assumed that all of drink products produced in the drink Filling phase were consumed at market and all the containers were sent to a recycling site and/or a waste treatment site for incineration and/or landfilling. Table 25 contains details of the modules used for the end-of-life processes.

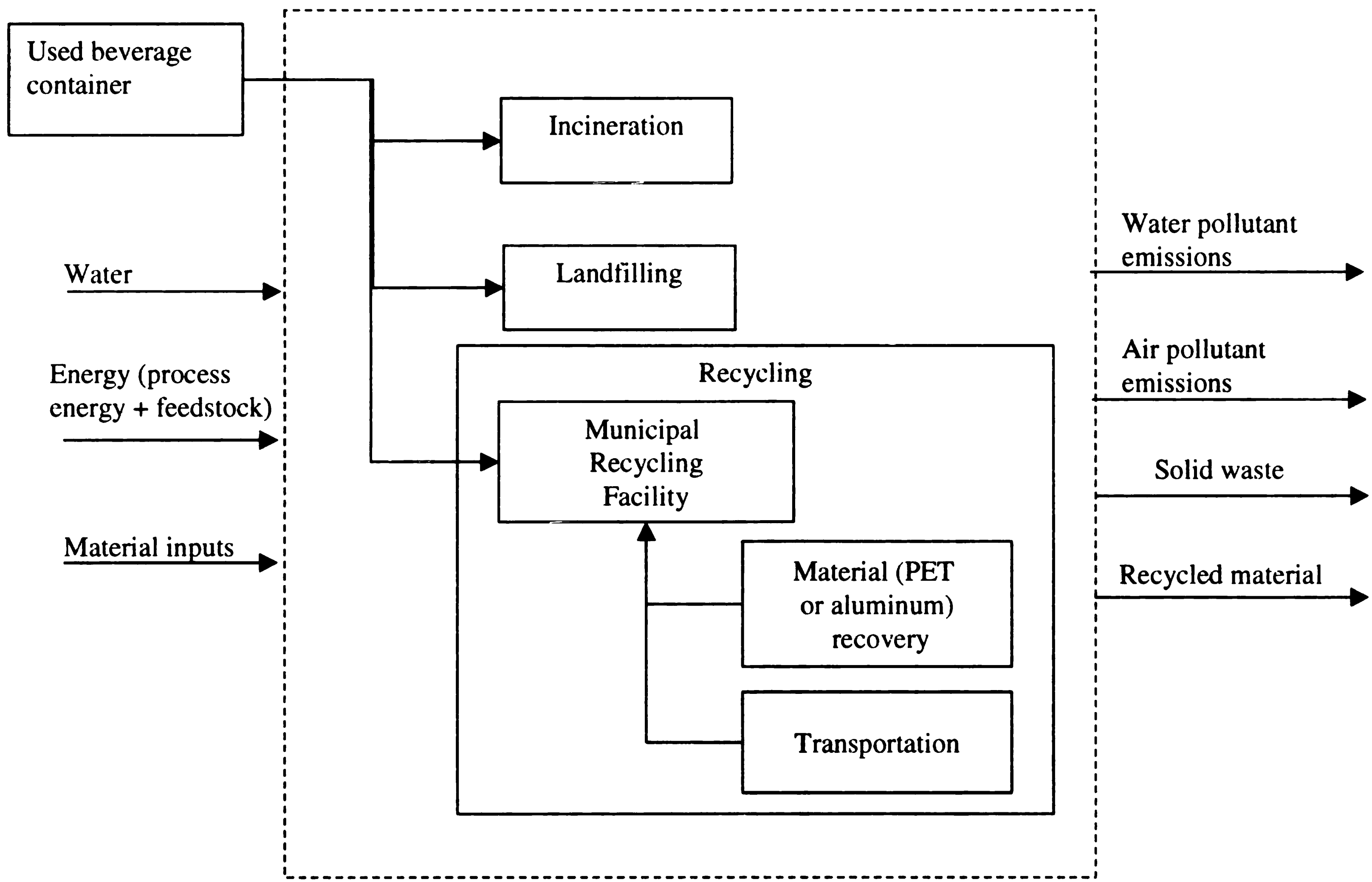


Figure 16. End-of-life processes

Table 25. End-of-life process data source

Module	Source
Incineration (plastic)	Background data: DEAM^{TM} module name Plastic (US, with energy recovery): Incineration.2
Incineration (aluminum)	Background data: DEAM^{TM} module name: Aluminum (US): Incineration.
Landfilling	Background data: DEAM^{TM} module name: Landfilling without energy recovery (US, MSW average).2
Recycling	Background data: DEAM^{TM} module names 1. Commingled recyclables with plastic recovery 2. Commingled recyclables with aluminum recovery 3. Road Transport (Truck 16 t, Diesel Oil, kg.km).1 4. Diesel Oil (US): Production.2

Material flow diagrams

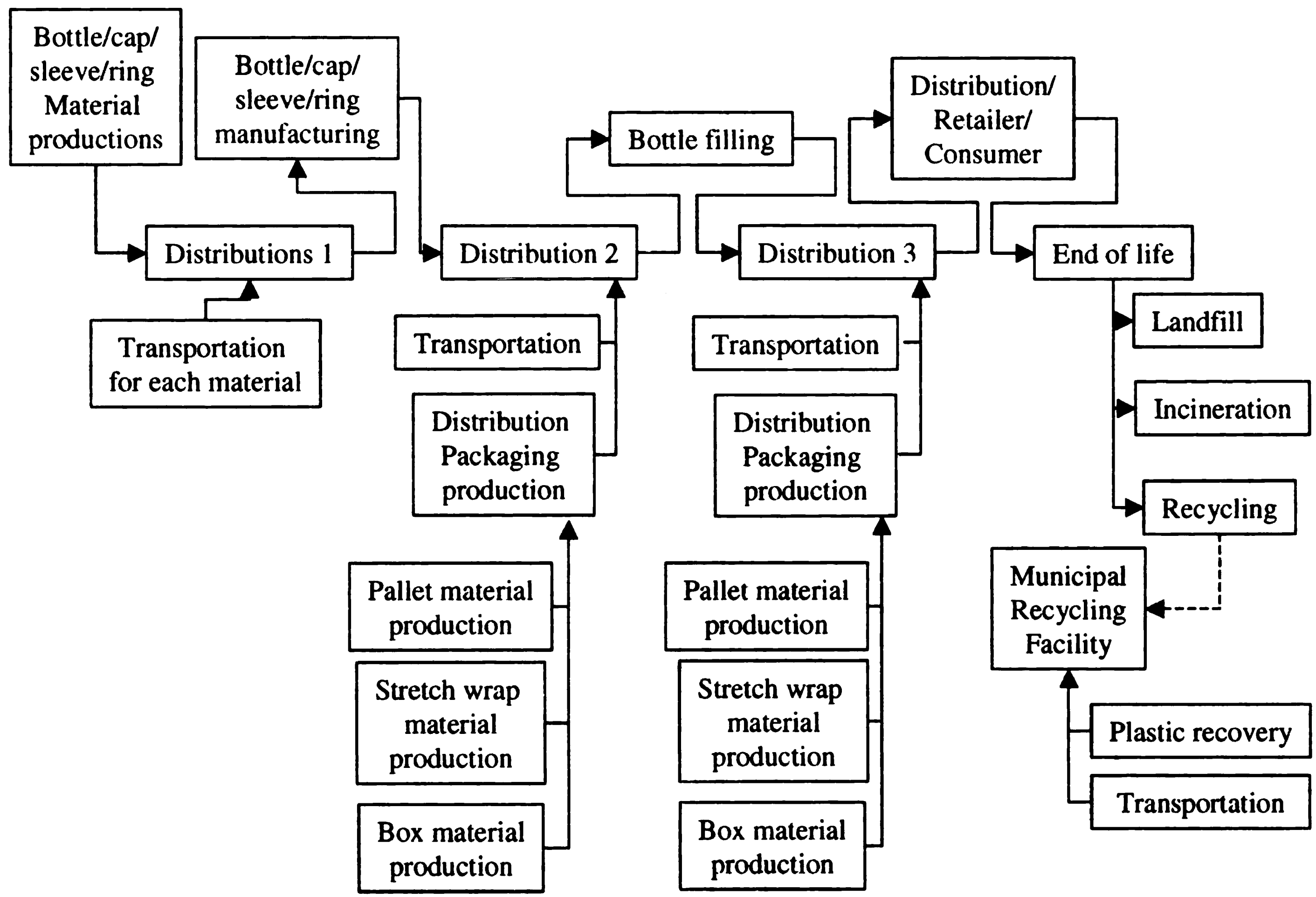


Figure 17. Material flow diagram of PDS 1 (PET based) and PDS 3 (PLA based)

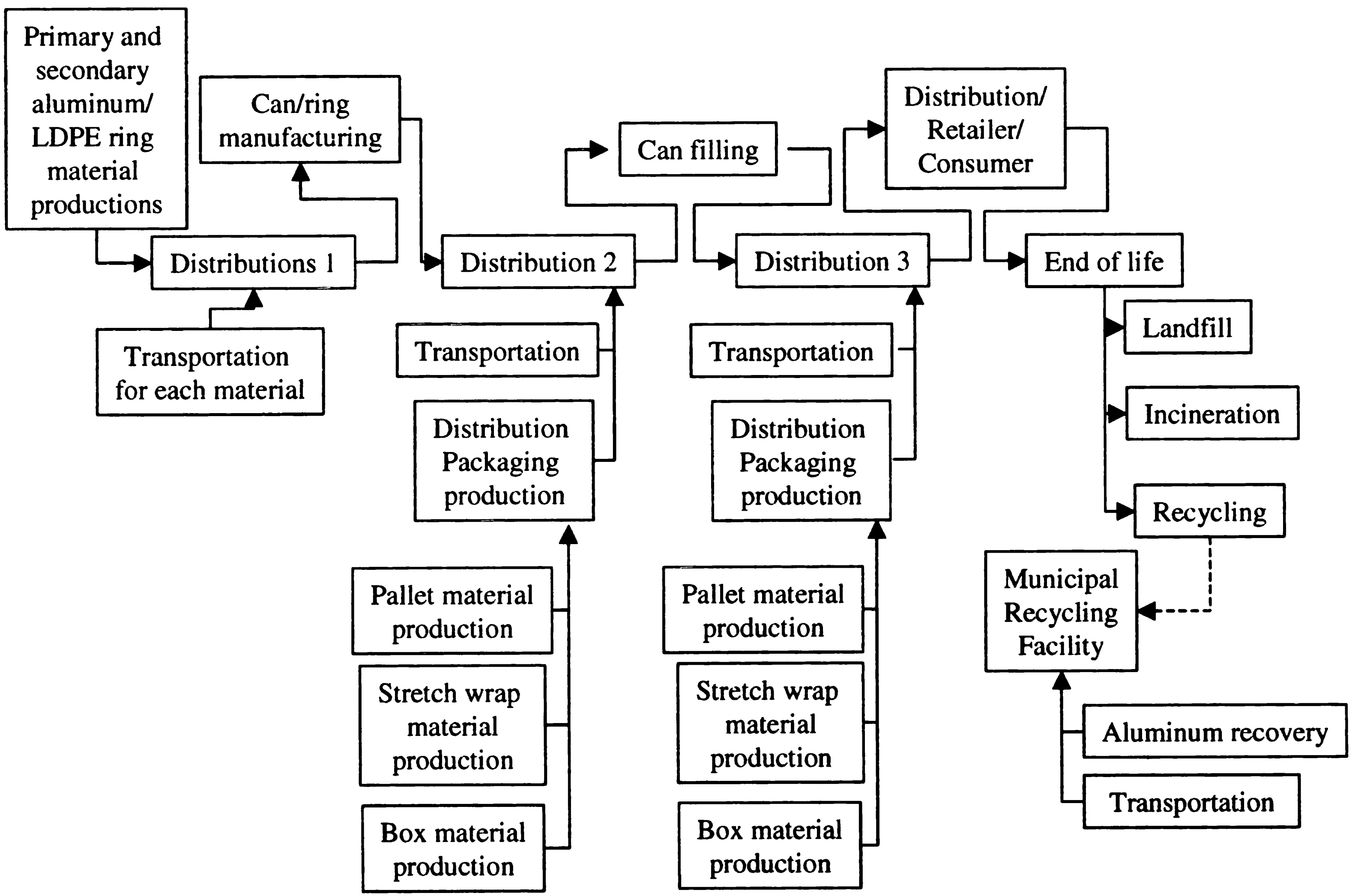


Figure 18. Material flow diagram of PDS 2 (aluminum based).

Allocation procedures

Allocation procedures are used to partition inputs and/or outputs within a product system. An allocation procedure is required when a unit process within a system shares a common pollution treatment infrastructure or where multiple products or co-products are produced in a common unit process. The system on which this research focused was a product delivery system so it did not produce any co-product, except for the electricity generated at municipal solid waste management sites. However, several allocation rules were adopted in each phase; the following sections discuss them phase by phase.

- Material Production:

Co-Product Allocation: Since all the input/output data in the Material Production phase were obtained from secondary data sources, the allocation rules were defined for these data modules in the sections above.

Recycling Allocation: As described above, recycling allocation rules also depended on the data modules from secondary data sources whose information was unavailable and out of the scope of this study

- Distribution 1:

Co-Product Allocation: There was no co-product from the Distribution 1 phase.

Recycling Allocation: There was no recycling allocation in the Distribution 1 phase.

- Manufacturing:

Co-Product Allocation: Since all the input/output data in the Manufacturing phase were obtained from secondary data sources, the allocation rules were defined for these data modules in the sections above.

Recycling Allocation: As described above, recycling allocation rules also depended on the data modules from secondary data sources.

- Distribution 2:

Co-Product Allocation: There was no co-product from the Distribution 2 phase.

Recycling Allocation: 100% of the corrugated cardboard boxes used in Distribution 2 was assumed to be sent to a recycling facility to be reused. However, since such boxes are reused in an open loop process (i.e. outside the PDS), the mass of the used boxes was classified in separately as recycled material. Pallets instead were assumed to be reused and some damaged. Expert estimations (White, 2005) assume that wooden pallets are used for about 16 roundtrips on average, with a reuse rate of about 94%. Similar operations (Keoleian, 2001) make the assumption that the remaining 6% of pallets are damaged. Then, it was assumed that these damaged 6% of total pallet were replaced with new pallets in each delivery cycle, and this is what finally was counted as pallet material (wood) input to the distribution phase under study.

- Filling:

Co-Product Allocation: Since all the input/output data in the Filling phase were obtained from secondary data sources, the allocation rules were defined for these data modules in the sections above.

Recycling Allocation: As described above, recycling allocation rules also depended on the data modules from secondary data sources.

- Distribution 3:

Co-Product Allocation: There is no co-product from the Distribution 3 phase.

Recycling Allocation: Allocation rules within the Distribution 3 phase were exactly the same as within the Distribution 2 phase.

- End of life:

Co-Product Allocation: Some incinerators have energy recovery devices that generate electricity by incinerating wastes. Electricity generated in this way at the End-of-Life phase was treated as a co-product, and the environmental burdens that would otherwise occur with normal electricity production were subtracted from the total environmental burden of the End-of-Life phase.

Recycling Allocation: There are several materials that were converted from solid wastes to reusable materials. In this study, it was assumed that some of the PET, PLA and aluminum is sent for recycling. For sake of simplicity, the rest of the materials were assumed to have a zero percent recycle rate.

Calculation procedures

This section explains the procedures by which inputs and outputs generated from each unit process were calculated. All the calculations of inputs and outputs from each unit process were based on the mass of product output from each unit process. For this reason, it was important to determine how much product output each unit process generates. The following section describes (1) how the product output from each unit process was determined and (2) how the inputs and outputs (burdens) from each unit process associated with the product output was calculated.

Functional Unit and Losses

The functional unit of this system is 1,000 L of drink delivered to wholesale retailers. Figure 14 illustrates how the product output from each process unit was calculated considering various losses that occurred in the downstream unit processes. To facilitate calculations, loss in this study is expressed as the fraction of desired process unit output that is lost in the process.

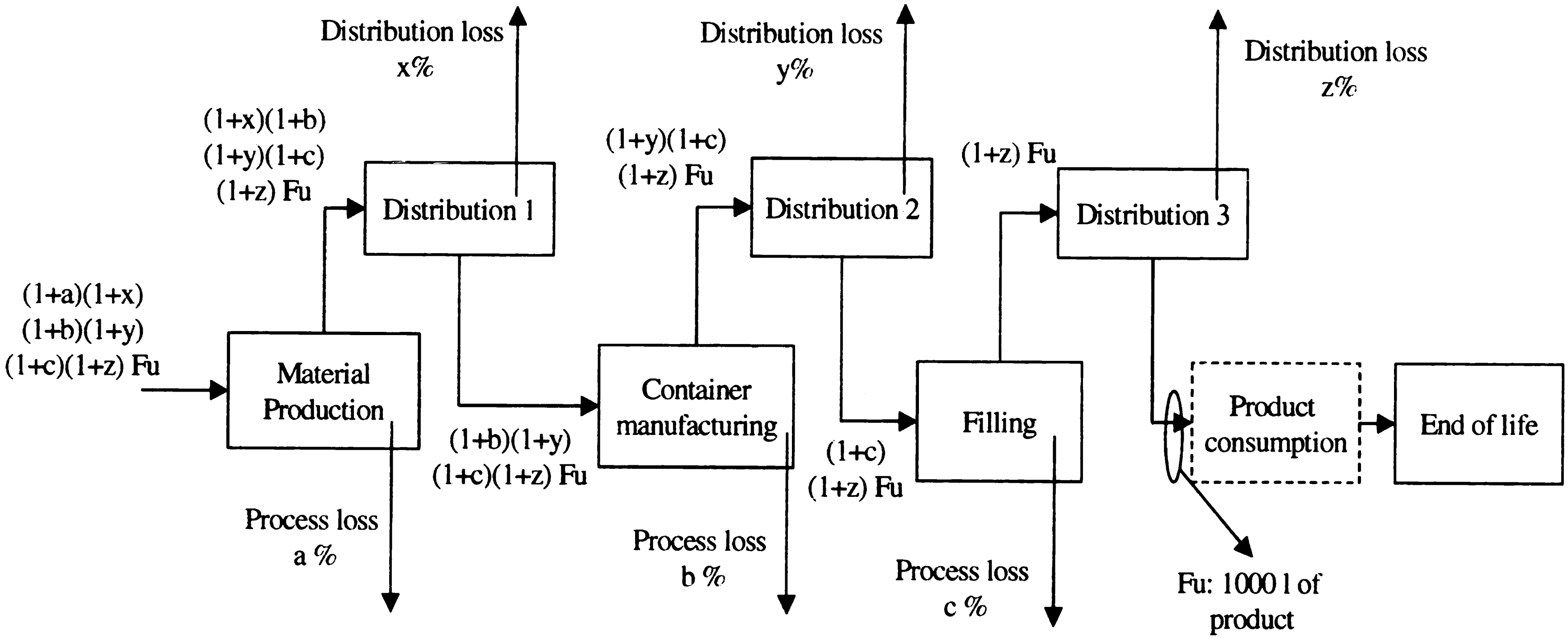


Figure 19. Losses in the product delivery system

For example, if a distribution loss fraction at Distribution 3 was defined as z, the drink Filling unit process has to produce z additional functional units of drink as its product output, because z out of 1,000 L is to be lost before it reaches retailers. The product output from each process unit was calculated by multiplying each of the down stream loss fractions plus unity by the functional unit, as shown in Figure 19.

The Material Production, Manufacturing and Filling phases included losses. The loss fraction "a" for Material Production is embedded in the individual material production DEAM™ modules. The loss fractions "b" and "c" (in Figure 19) were defined as single fractions for the container manufacturing and filling unit processes respectively and details of the values are in the secondary data sources used (SAEFL, LCI for packagings. Vol. II., 1998b). In all cases, "x", "y" and "z" fractions were assumed zero. Once the product output was determined by the formulas described above, total environmental burdens from each unit process were calculated based on the mass of product output. The following sections describe calculation procedures for all phases in order from downstream to upstream.

To obtain the mass of drink containers sent to municipal solid waste (MSW) management site, it was assumed that the entire functional unit of drink products delivered to wholesale retailers was all carried to the MSW facility. In other words, a zero loss fraction after Distribution 3 phase was assumed. The total mass of waste was classified into three categories; MSW sent for incineration, MSW sent for landfilling, and material that is sent to a municipal recycling facility. The total inputs and outputs were calculated by multiplying the mass of each category (i.e. mass for incineration) with mass based environmental burden data, corresponding to each category.

In Distribution 3, three categories were taken into consideration to aggregate the total environmental burdens of this phase. They are: Transportation, Material production of distribution packaging, and Solid waste

Transportation: In order to calculate the burdens of transportation, total km-kg was calculated by aggregating the following pairs of weights and distances.

Weight	Distance
Drink product	Filling facility to retailers/distributors
Drink containers	Filling facility to retailers/distributors
Distribution packaging	Filling facility to retailers/distributors
Distribution packaging	Supplier to Filling facility

The weight of drink product consisted of the weight of a functional unit of drink and the weight of the drink containers. Distribution packaging included corrugated boxes, pallets, and stretch wrap.

Material Production of Distribution Packaging: The burdens of the material production were calculated by multiplying the weight of materials used for the distribution packaging and each material's burden data obtained from secondary data sources. The materials included corrugated board for corrugated box, wood for pallet, and LLDPE film for stretch wrap.

Solid Waste: The masses of solid waste from Distribution 3 were calculated in the two categories described below.

Loss from transportation: A transportation loss fraction was allowed and aggregated by multiplying it with the weight of drink carried by truck.

Distribution packaging: Some of the distribution packaging was assumed to be recycled in an open loop and so counted as a Recycled Material output. The rest non-recycled distribution packaging was landfilled. Total burden was calculated by multiplying the mass of waste by the DEAM™ MSW Landfill module data.

In distribution 2, three different categories again were included. They are: Transportation, Material Production and Solid waste.

Transportation: In order to calculate the burdens of transportation, total kg-km was calculated by aggregating the products of the following weights and distances.

Weight	Distance
Container components	Container component manufacturers to filling facility
Distribution packaging	Container component manufacturers to filling facility
Distribution packaging	Supplier to Container manufacturers

The distribution packaging consisted of corrugated boxes, sleeves, separator cardboard, pallets and the stretch wrap.

Material Production of Distribution Packaging: Calculation procedure was the same as for Distribution 3

Solid Waste: Calculation procedure was the same as for Distribution 3

The mass of all materials required for the functional unit was calculated based on the weight of product output and composition data of each container component. Total environmental burdens were calculated by multiplying the mass of each material and corresponding data modules.

Loss from transportation: Calculation was the same as for Distribution 3.

Assumptions and limitations of this study

General

- Burdens associated with distribution phases were based on the assumptions and calculations of the DEAM™ Road Transport module (Truck, 40 ton capacity, Diesel, kg/km, 100% efficiency).
- Wooden pallets were assumed to be reused until worn or damaged, at which time they were typically remanufactured. The re-manufacturing process entailed replacing only the worn or damaged components of the pallets. Therefore, burdens associated with the production of pallets were calculated using only the quantity of wood consumed per functional unit.

Material Production

- The environmental burdens associated with material production of PP, LLDPE, PET, and PE were based on European production data. Material production burdens for Europe were assumed to be similar to those in North America.
- No additives or fillers were accounted for in the production phase.
- The environmental burdens associated with PLA polymerization were based on estimates by the manufacturer (Vink et al., 2003).

Distribution 1

- All materials were assumed to be shipped in bulk trucks and therefore did not use distribution packaging.
- 40-ton trucks were assumed to be, on average, full on the delivery leg of the trip and empty on the return leg of the trip.

Manufacturing

- The PLA injection stretch blow molding requirements were assumed to be the same as for PET per kg basis, but the weight of the PLA bottle was assumed to be 6% lighter than that of PET.
- The loss fractions for blown film extrusion, injection molding and injection stretch blow molding were based on estimates from SAEFL (LCI for packagings. Vol. II., 1998b).
- The loss fraction for the can making process was assumed to be 0.01.

Distribution 2

- All distribution packaging was assumed to be shipped by truck from the supplier to the manufacturer.
- Trucks were assumed to be full on the delivery leg of the trip and empty on the return leg of the trip.
- It was assumed that damaged pallets were replaced with new pallets in each delivery cycle, and this is what finally was counted as pallet material (wood) input to the distribution phase under study.
- Distribution packaging formats for PLA containers was assumed to be identical to the distribution packaging currently being used for PET containers.

Filling

- The PLA and PET containers were assumed to be similar in design, and therefore existing filling, sealing and palletizing equipment can be used. Requirements for Injection stretch blow molding, which was included in this phase, were assumed identical for both PLA and PET.

Distribution 3

- Trucks were assumed to be, on average, full on the delivery leg of the trip and empty on the return leg of the trip.
- The same distribution packaging was used for PDS 1 and PDS 3.

End-of-Life

- All wastes produced during the Material Production, Manufacturing, Filling and Distribution phases (all except End-of-Life) was assumed to be sent to landfill.
- One hundred percent (100%) of the containers shipped to the distributor/retailer/consumer unit process in Distribution 3 are assumed to reach the End-of-Life phase. In other words, none were assumed to be disposed of improperly or maintained by the consumer to be reused for other purposes.

Input data

Besides foreground and background data, there are input data. Input data were measured (e.g. weight of a PET bottle, aluminum can), estimated using software and/or literature data for average operations (e.g. distribution packaging formats and weights), or arbitrarily set (for all PDSs equal) with the spirit of representing a relatively average operation involving drink products (e.g. transport distances).

Input data used in this study is included in this section. The details about the variable input data (i.e. data were varied during the uncertainty analysis) is postponed for discussion the next chapter.

- Material production phase inputs

PDS	For component	Material resin	Mass per package, Kg	Recycled content, %
1	Bottle	PET	Variable	0
	Cap	PP	0.0025	0
	Sleeve	LDPE	0.0005	0
	Ring	LDPE	0.002	0
2	Can	Aluminum	Variable	Variable
	Ring	LDPE	0.002	0
3	Bottle	PLA	Variable	0
	Cap	PP	0.0025	0
	Sleeve	LDPE	0.0005	0
	Ring	LDPE	0.002	0

- Distribution 1 phase inputs

PDS	For component	Material resin	Transportation distance, Km	Loss fraction
1	Bottle	PET	400	0
	Cap	PP	400	0
	Sleeve	LDPE	400	0
	Ring	LDPE	400	0
2	Can	Aluminum	400	0
	Ring	LDPE	400	0
3	Bottle	PLA	Variable	0
	Cap	PP	400	0
	Sleeve	LDPE	400	0
	Ring	LDPE	400	0

- Container manufacturing phase inputs

PDS	For component	Material resin	Process	Electricity requirement, MJ/Kg	Loss fraction	Information source
1	Bottle	PET	Injection molding	Variable	0.0049	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Cap	PP	Injection molding	3.51	0.01	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Sleeve	LDPE	Conversion	1.29	0.04	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Ring	LDPE	Conversion	1.29	0.04	(SAEFL, LCI for packagings. Vol. II., 1998b)
2	Can	Aluminum	Can making	12.67	0.01	(Mitsubishi Materials, LCA of aluminum can, 2003)
	Ring	LDPE	Conversion	1.29	0.04	(SAEFL, LCI for packagings. Vol. II., 1998b)
3	Bottle	PLA	Injection molding	Variable	0.0049	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Cap	PP	Injection molding	3.51	0.01	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Sleeve	LDPE	Injection molding	1.29	0.04	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Ring	LDPE	Injection molding	1.29	0.04	(SAEFL, LCI for packagings. Vol. II., 1998b)

- Distribution 2 phase inputs

Inputs for primary packaging transportation

PDS	For component	Material resin	Transportation distance, Km (assumed)	Loss fraction (assumed)
1	Bottle	PET	400	0
	Cap	PP	400	0
	Sleeve	LDPE	400	0
	Ring	LDPE	400	0
2	Can	Aluminum	400	0
	Ring	LDPE	400	0
3	Bottle	PLA	400	0
	Cap	PP	400	0
	Sleeve	LDPE	400	0
	Ring	LDPE	400	0

Inputs for distribution packaging transportation

PDS	For component	Distribution packaging	Component units per dist. packaging unit	Weight of unit, Kg	Transportation distance, Km (assumed)	Loss fraction (assumed)	Information source
1, 3	Bottle preform	Corrugated box	12000	3	100	0	(Giles, 1999)
		Stretch wrap	12000	0.32	100	0	(White, 2005)
		Wood pallet	12000	23.7	100	0	(Giles, 1999)
	Cap	Corrugated box	20000	0.8	100	0	(Giles, 1999)
		Stretch wrap	20000	0.32	100	0	(White, 2005)
		Wood pallet	20000	23.7	100	0	(Giles, 1999)
	Sleeve	Corrugated box	38500	0.8	100	0	(Giles, 1999)
		Stretch wrap	385000	0.32	100	0	(White, 2005)
		Wood pallet	385000	23.7	100	0	(Giles, 1999)
	Ring	Corrugated box	38500	0.8	100	0	(Giles, 1999)
		Stretch wrap	385000	0.32	100	0	(White, 2005)
		Wood pallet	385000	23.7	100	0	(Giles, 1999)
2	Aluminum can body	Pads	361	0.5	100	0	(Giles, 1999)
		Stretch wrap	7942	0.32	100	0	(White, 2005)
		Wood pallet	7942	23.7	100	0	(Giles, 1999)
	Aluminum can lid	Sleeves	35	0.25	100	0	(Giles, 1999)
		Stretch wrap	3500	0.32	100	0	(White, 2005)
		Wood pallet	3500	23.7	100	0	(Giles, 1999)
	Ring	Corrugated box	38500	0.8	100	0	(Giles, 1999)
		Stretch wrap	385000	0.32	100	0	(White, 2005)
		Wood pallet	385000	23.7	100	0	(Giles, 1999)

- Filling inputs

PDS	For component	Material resin	Process	Electricity requirement, MJ/Kg	Loss fraction	Information source
1	Bottle	PET	Injection stretch blow molding	8.40	0.001	(SAEFL, LCI for packagings. Vol. II., 1998b)
	Cap	PP	N/A	N/A	N/A	
	Sleeve	LDPE	N/A	N/A	N/A	
	Ring	LDPE	N/A	N/A	N/A	
2	Can	Aluminu m	N/A	N/A	0.001	Assumed
	Ring	LDPE	N/A	N/A	0	
3	Bottle	PLA	Injection stretch blow molding	8.40	0.001	Assumed
	Cap	PP	N/A	N/A	N/A	
	Sleeve	LDPE	N/A	N/A	N/A	
	Ring	LDPE	N/A	N/A	N/A	

- Distribution 3 phase inputs

PDS	Distribution packaging	Product units per dist. packaging unit	Weight of unit, Kg	Transp. distance, Km (assumed)	Loss fraction (assumed)	Information source
1, 3	Corrugated board tray	24	0.26	100	0	(CAPE, CAPE Systems: palletization and package design software, 2000)
	Stretch wrap	1296	0.32	100	0	(White, 2005)
	Wood pallet	1296	23.7	100	0	(CAPE, CAPE Systems: palletization and package design software, 2000)
2	Corrugated board tray	24	0.26	100	0	(CAPE, CAPE Systems: palletization and package design software, 2000)
	Stretch wrap	1296	0.32	100	0	(White, 2005)
	Wood pallet	1296	23.7	100	0	(CAPE, CAPE Systems: palletization and package design software, 2000)

- End of life inputs

PDS	Component	EOL stream	Percentage	Transportation distance, Km (assumed)	Loss fraction (assumed)	Information source
1, 3	Bottle	Recovery	Variable	100	0	
		Incineration	11.5%	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	Variable	N/A	0	
	Cap	Recovery	0	N/A	0	(U.S EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Incineration	24	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	76	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
	Sleeve	Recovery	0	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Incineration	24	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	76	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
	Ring	Recovery	0	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Incineration	24	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	76	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)

PDS	Component	EOL stream	Percentage	Transportation distance, Km (assumed)	Loss fraction (assumed)	Information source
2	Aluminum can body	Recovery	Variable	100	0	
		Incineration	1%	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	Variable	N/A	0	
	Ring	Recovery	0	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Incineration	24	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)
		Landfilling	76	N/A	0	(U.S. EPAMunicipal solid waste in the U.S.: facts and figures, 2001)

III. Preliminary conclusions

This chapter has presented detailed information regarding a comparative life cycle assessment (LCA) based on a hypothetical drink product for which we evaluate three main container material alternatives: (1) a polyethylene terephthalate (PET) bottle; (2) an aluminum can; and (3) a polylactide (PLA) bottle.

It has followed ISO's LCA steps, and details and information sources used for this comparative LCA were presented. Representative realistic implications of the use of the three main material/container alternatives and their actual packaging operations, production and use of distribution packaging (i.e. corrugated trays, stretch wrap, and pallets), as well as transportation steps through several parts of the packaging life cycle were included. Relevant end-of-life scenarios such as landfilling, incineration and recycling were also taken into account in this analysis. Each of the main material alternatives (i.e. PET, PLA, aluminum) along with its whole set of operations was defined as a product delivery system (PDS) and environmental burdens associated with each of the three PDSs were included in the study.

As shown, the procedure for performing an actual LCA according to ISO's recommendations is both time consuming, and greatly data intensive and heavily relies on gathering of quality data. More significant is the fact that, in spite of the effort made in this study to identify quality sources of data and represent production systems in as standard a form as possible, this comparative LCA is still a streamlined version and thus, its results and conclusions need to be handled acknowledging such limitation.

The actual results and discussions of the LCA presented here are in the next chapter along with the actual uncertainty estimation.

CHAPTER 4 UNCERTAINTY ANALYSIS OF THE COMPARATIVE LIFE CYCLE ASSESSMENT OF DRINK DELIVERY SYSTEMS

I. Introduction

Uncertainty analysis is something still new in LCA, as discussed in Chapter Two. Just in the last years a number of methodologies for uncertainty estimation have been proposed. Currently, the most favored approach for uncertainty and variability analysis in LCA is incorporating it into the life cycle inventory (LCI) stage. In fact, the study of uncertainty in the inventory data is a natural inclination since LCI is the actual quantitative analysis of an LCA and relies heavily on the quality of the information regarding inputs of raw materials and fuels into a system and the outputs of solid, liquid and gaseous wastes from it. At this stage, data associated with the flows is collected using literature studies, interviews, measurements, theoretical calculations, data banks and qualified guesses. In theory, the application of allocation principles and procedures should also be explained, and information required in recycle or reuse situations should also be presented. However, this is not the reality in most cases, and some LCA experts even argue that since there are no standards for co-product or recycling allocations of environmental emissions, it is impossible, for instance, to look at an off-the-shelf computer LCA program and understand how the numbers were generated. This is a serious problem for the credibility of the LCA method since, with the idea of user convenience and to speed the process of performing an LCA, it is not uncommon to find private groups developing computer programs using prepared databases which are, in turn, compilations of other external data sources. Moreover, leaving the uncertainty due to variation in allocation procedures aside (which can be up to 50%), Finnveden and Lindfors (1998) found that apparent mistakes in the reporting of inventory data can be real and more detrimental. In fact, mistakes in LCI data are not uncommon and have been found to explain some very large variations in emission data (sometimes orders of magnitude different). On the other hand, data variations (large or small) can be real since they may simply reflect natural variations in emission data or differences in types of technologies.

But, as explained earlier as well, uncertainties in LCA are not restricted to LCI. Very commonly and with great consequences in packaging, uncertainties due to the analysis of different scenarios may be the reason for conflicts in LCA results. For instance, differences in percent recycled content in a packaging material, packaging end-of-life practices (e.g. landfilling, incineration, recovering), and product distribution distances all can greatly change the outcome of the LCA. Even within the same country, the same packaging end-of-life alternative can be viewed both as environmentally friendly and environmentally unsound. In fact, analyzing several comparative packaging based LCA results, it is not uncommon to find directly opposite results that are often the subject of heated debates among industry groups and even countries (IFEU, World largest PET LCA, 2004; Hockerts et al., 1999; Hocking, 1991; Saphire, 1994; Wells et al., 1991).

In this dissertation we argue that conflicts in the results of comparative packaging based LCAs may arise mainly due to uncertainties in both the LCA method and the actual packaging operations under consideration. Thus, in order to understand the nature behind the conflicts in LCA results, this chapter presents an analysis of how uncertainties actually affect the outcome of the comparative packaging-based LCA developed in Chapter Three.

Specifically, the uncertainty analysis in this study comprises two aspects. First, we perform a scenario analysis by identifying different domains that may have the potential to change the outcome of the comparative packaging based LCA. Under each domain, two alternatives are proposed. Then, a technique is used to select an alternative from each domain and randomly create different scenarios.

The second aspect of the uncertainty analysis in this study comprises inventory parameter uncertainty. The study on inventory parameter uncertainty is achieved by the use of appropriate published uncertainty estimations along with stochastic simulation (i.e. Monte Carlo simulation) in order to represent inventory data uncertainty as probability distributions. By this analysis, relevant inventory parameters are identified and further analyzed, and final results of the comparative LCA along with the combined uncertainties from inventory data and scenarios are obtained.

The next section discusses with more detail the methodology for the study, including details of the areas selected for the scenario analysis and the procedure followed. Later in that section, information regarding the parameter uncertainty analysis of the inventory data itself is presented, including specifics about the uncertainty distributions based on engineering estimates and computer simulation procedures. Lastly, we present the results of the uncertainty evaluation based on the comparative LCA results along with an extensive discussion of the reasons for the differences and their consequences.

Chapter Five summarizes the overall conclusions of the dissertation along with several recommendations for future work.

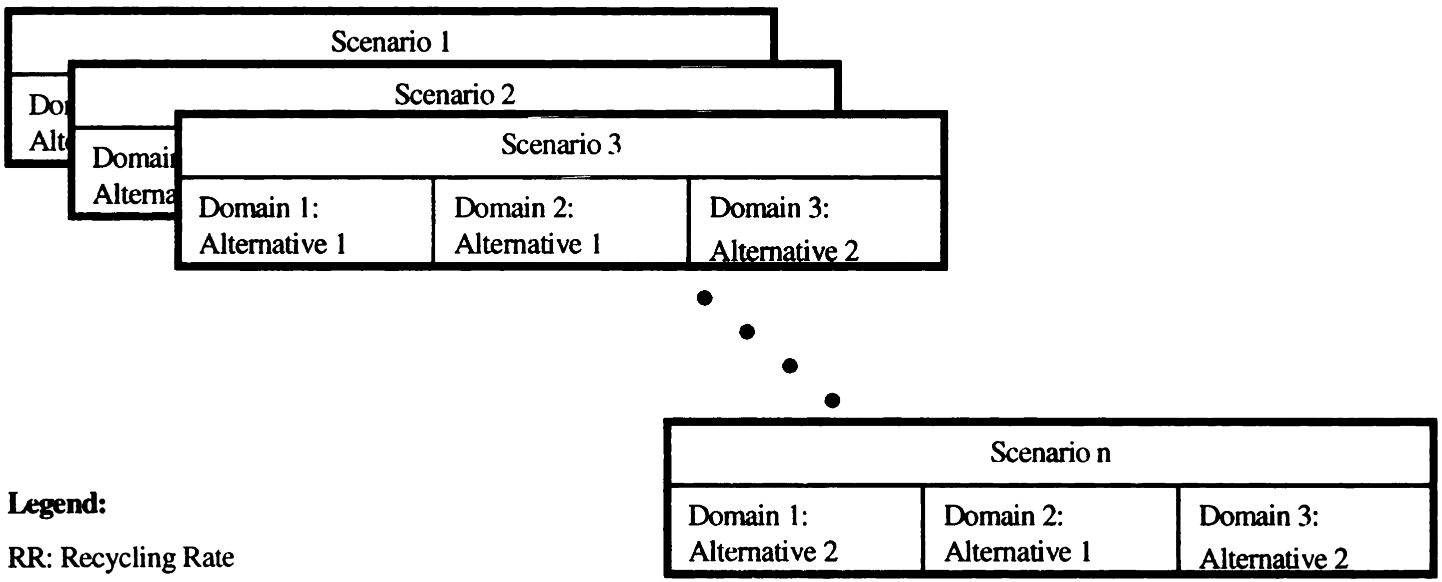
II. Methodology

a) Scenario uncertainty analysis

Figure 20 outlines the procedure followed for the scenario analysis in this study. First, for the PDSs explained in Chapter Three, domains associated with packaging situations with perceived effect on the LCA outcome were identified. In this study, we used three domains: material recycled content percent, breakdown of end-of-life streams, and Distribution 1 distance in the case of PLA. Material recycled content percent was perceived as relevant since, for instance, in the case of aluminum, it has been reported to greatly decrease the energy consumption of the material production operations. Differences in the end-of-life streams were also assumed relevant, since for instance in the U.S., deposit refund systems, which may or may not be in place or not, have been found to greatly affect the percentage of material sent for recycling. The last domain, Distribution 1, which specifically targets the distance traveled to supply PLA resins to a component manufacturer, was assumed relevant, since so far in the whole U.S. there is only one facility commercially producing this plastic, and thus the effect of transportation for large distances may be significant.

Domain 1: Recycled content		Domain 2: End of life		Domain 3: Distribution 1	
Alternative 1: PET: 0% Aluminum: 30% PLA: 0%	Alternative 2: PET: 0% Aluminum: 60% PLA: 0%	Alternative 1 : PET RR: 10% Aluminum RR: 19% PLA RR: 0%	Alternative 2: PET RR: 39% Aluminum RR: 75% PLA RR: 10%	Alternative 1: PLA Distribution 1: 200 Km.	Alternative 2: PLA Distribution 1: 2500 Km

Random selection of one alternative per domain every iteration



Legend:

RR: Recycling Rate

Figure 20. Procedure followed for scenario analysis

The scenario analysis procedure continues with the identification of two alternatives for each domain. In this study, though alternatives for each domain were arbitrarily selected, they in many cases can still represent actual situations. After the identification of the alternatives, an iterative procedure was put in place in order to, by random selection of one alternative per domain at a time, create a scenario and calculate its LCA outcome. In this study, this calculation was performed 10,000 times, and statistics (i.e. median, confidence intervals) were calculated afterwards. The resulting output distribution reflects the uncertainty of the comparative LCA outcome, taking into account the alternatives in the domains under consideration. The Visual Basic code developed to perform this scenario analysis is in the Appendix.

In this analysis, to determine the potential impact of the identified alternatives on the comparative study outcome, all alternatives per domain were given equal probability. To illustrate the consequences of ignoring or reducing scenario uncertainty, the results of this simulation will be compared with the outcomes that would have resulted if one specific alternative had been chosen.

b) Inventory data uncertainty analysis

Inventory uncertainty analysis was performed using Monte Carlo (MC) simulation. MC, in general, is a numerical technique to randomly generate or sample data to obtain approximate solutions to complex mathematical and statistical problems. In fact, MC is a technique to randomly generate a number under a defined probability distribution and can be used to represent uncertainty and variability of inventory parameters of an LCA (Heijungs and Suh, 2002; Huijbregts, 1998a; 1998b; 2001; Huijbregts et al., 2003). To actually perform MC simulation in LCA, first each uncertain input parameter is specified as an uncertainty distribution. Next, the method randomly selects one number from the specified uncertainty distribution for each parameter. All selected numbers are then used for the calculation of the desired output. Iterative calculations end up producing a distribution of the predicted output values, reflecting the combined parameter uncertainties.

Since this LCA study involves an enormous amount of parameters, it was not feasible to specify the uncertainty distributions for all these parameters in detail. This problem was overcome with a stratified procedure developed by Huijbrets et al (2003). A diagram of the method is presented in Figure 21.

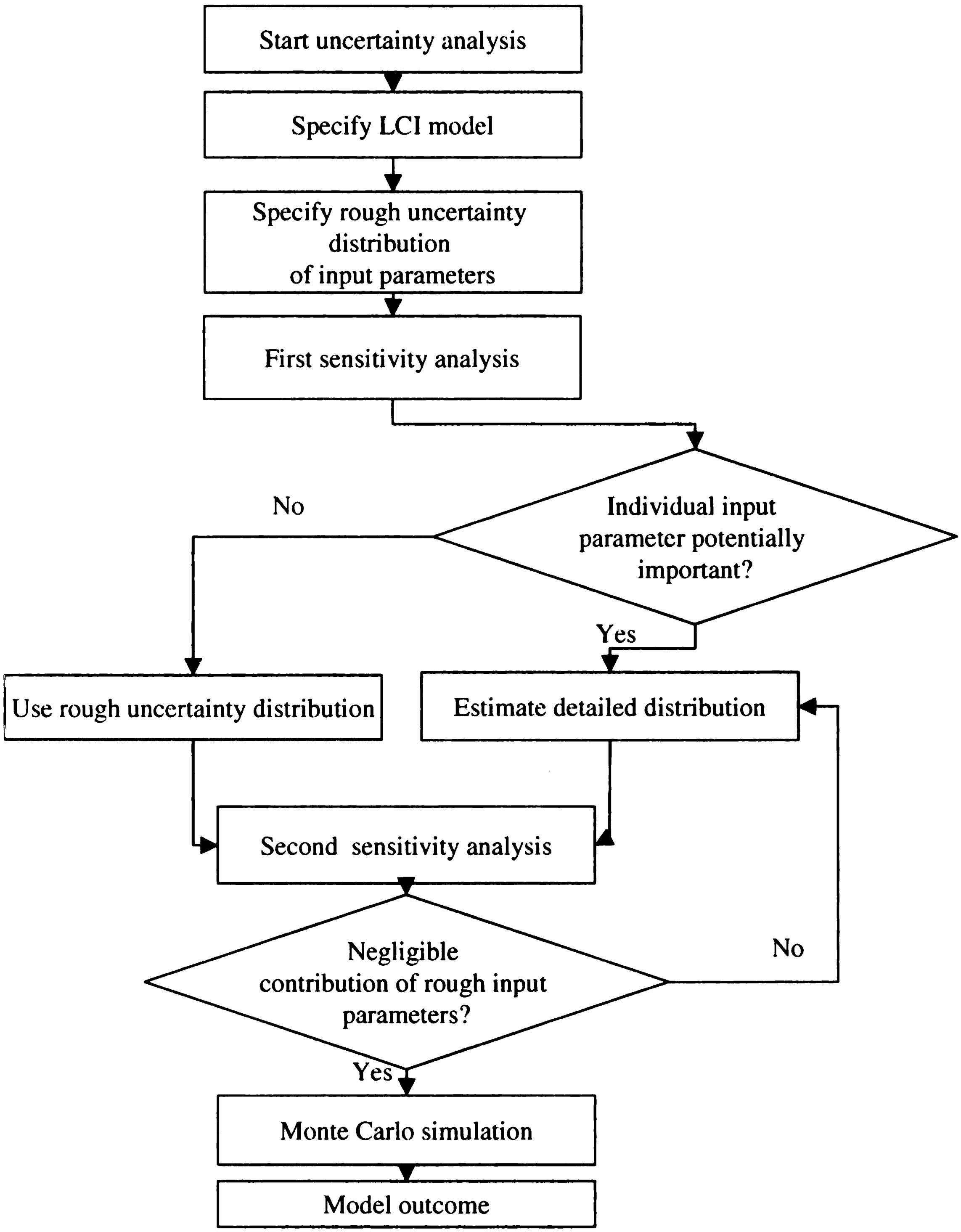


Figure 21. Inventory data uncertainty analysis procedure (Huijbregts et al., 2003)

First, each uncertain input parameter was assigned a distribution in order to reflect a conservative estimation of its uncertainty based on experience or engineering estimations. In this study, all inventory data was assigned the same log-normal distribution defined using upper and lower ends covering 95% probability in the distribution. The upper and lower ends were estimated by the approach used by Slob (1994) and expressed by the following formula:

$$P\left(\frac{M(X)}{k} < X < k \times M(X)\right) = 0.95 \qquad (1)$$

where M(X) is the median value of each parameter, and k is the uncertainty factor assigned to that parameter. It was assumed that all the inventory values reported in all DEAM™ modules were medians. Their initial uncertainty factors (k) were obtained using expert recommendations published in the literature (Huijbregts et al., 2003) and are listed in Table 1 of the Appendix.

Next, a first sensitivity or screening analysis was performed to identify the parameters that had a strong correlation (i.e. were "important") with the LCA results. This analysis was performed by running a first MC simulation followed by a Spearman's rank correlation procedure. This rank correlation method was done by first ranking each of the simulated values of each LCI parameter and their correspondent LCA result in ascending order. Next, for each simulated LCI parameter-LCA result pair a correlation coefficient (i.e. Spearman's coefficient) was calculated between them using their ranking values instead of their actual values (since a correlation coefficient measures the strength of the linear relationship between two variables, it is unlikely to find any linear relationship between variables that have different distributions as in our case; thus, using their rank instead of their actual values, that limitation was avoided) (Decisioneering Inc., Crystal ball - User manual, 2004). Afterwards, for each LCI parameter-LCA result its average correlation coefficient was calculated based on the total number of simulated values. Finally, the contribution to the overall variance of each LCI parameter was estimated by squaring each averaged rank correlation coefficient and normalizing them to 100% (i.e. adding all squared coefficients, dividing each squared coefficient by this sum and multiplying it by 100%) (Technical note from Decisioneering, How does Crystal Ball calculate sensitivity?, 1998). Parameters that contributed at least 1% to the overall variance of each LCA result were considered "important". More details about this calculation and Spearman's formula are included in the Appendix.

When more complete uncertainty information was available, those important parameters were specified in more detail, and that information is presented in Table 2 of the Appendix. The importance of those parameters was confirmed in a second sensitivity analysis. This second analysis was necessary because defining parameters in more detail may have affected their uncertainty importance. When important inventory parameters were specified in detail, and keeping the "unimportant" parameters with their rough uncertainty distributions, a final MC simulation was performed to quantify the output uncertainty.

Some notes of caution about the actual accuracy of this uncertainty analysis are required at this point. First, as stated earlier, the uncertainty analysis is something new in LCA, and the use of MC simulation as well as the appearance of other approaches and methodologies for uncertainty estimation in the last few years only reinforces the idea that the quantification of uncertainty is itself uncertain. Moreover, within the same uncertainty estimation approach, uncertainties in the uncertainty analysis itself may not be avoided. For instance, a case can be made for the use of different distributions (i.e. triangular, uniform, gamma, etc) to represent parameter uncertainty. In this study, for the sake of simplicity, a log-normal distribution was chosen to represent uncertainty (i.e. lack of knowledge) of most of the parameters. The few exceptions are parameters such as packaging material weight and product weight for which a normal distribution was used, representing variability (i.e. natural variation) rather than uncertainty (see Table 2 of the Appendix). Worth noting is the fact that the use of a log-normal distribution appears to be a standard practice by risk analysis experts to represent uncertainty since it avoids negative values, it captures a large value range, and the uncertainty in many processes and parameters follows a skewed distribution (Slob, 1994).

A final word of caution is regarding the application of this analysis to prepared data such as those contained in the DEAM™ database. Since this database is a compilation of data from a number of other sources, and Ecobilan itself does not guarantee that the original studies have delivered high quality results (Ecobilan Group Inc., DEAM module databases and manuals, 1999), the basis for the application of this uncertainty analysis is to represent the uncertainty contained in those sources (in fact Ecobilan in its LCA specific software called TEAM™, which in turn works with its DEAM™ database, allows for uncertainty analysis using MC simulation). Moreover, since many details about the procedures followed by Ecobilan to adapt those databases (e.g. to convert them to the single format that it uses) are not easily known or validated and outside the scope of this analysis, the study also represents that uncertainty. Thus, results of this uncertainty analysis should be interpreted to be a conservative estimation of that of the overall results of the comparative LCA under consideration.

Presentation of results

All results of this uncertainty analysis refer to and use the same identification notation of the three product delivery systems (PDS) explained in Chapter Three. As explained there, each PDS was defined as the system employed for the distribution of a hypothetical drink to customers and included the primary and distribution packaging, as well as manufacturing, and transportation associated with the function of the system. The alternative product delivery systems considered are:

- PDS 1: utilizing PET bottles and PP caps as one choice for drink delivery.
- PDS 2: utilizing aluminum cans as the other choice for drink delivery.
- PDS 3: utilizing PLA bottles and PP caps to investigate the use of renewable material feedstocks.

To facilitate the comparison of the LCA results (i.e. the characterized and not characterized environmental impact indicators as explained in Chapter Three) for each of the three PDSs, they were related using the following relationship (Huijbregts et al., 2003):

$$CR_u = \frac{r_{u,n}}{r_{u,m}} \qquad (2)$$

where CR_u is the comparison ratio for impact indicator u (dimensionless) and $r_{u,n}$ and $r_{u,m}$ are the environmental impacts related to PDS n and m, respectively (e.g. expressed in MJ for energy use). The environmental impact indicators of the product systems compared can be considered to be significantly different, for instance, if 95% of the iterations gave results above 1.

Simulation details, hardware and software

In this study, simulations for both inventory data uncertainty analysis and scenario uncertainty analysis were simultaneously carried out. The number of iterations used for all simulations was 10,000, as recommended in the literature to achieve a representative distribution of cumulative results (Huijbregts et al., 2001). MC simulation and scenario analysis were performed using codes and tools available in Crystal Ball 2000.v5 (Denver, CO) (Crystal ball - Forecasting and risk analysis for spreadsheet users, 2004), and Visual Basic programming and spreadsheets of Microsoft Excel (Microsoft Corp., Microsoft Excel, 2000).

III. Results

This section will be divided into two main parts. First, overall comparative LCA results will be presented with the inclusion of both scenario and LCI data uncertainty. Next, outcomes that would have resulted if one specific scenario had been chosen are presented.

a) Overall comparative LCA results

Figure 22 shows the overall uncertainty in the comparison ratios of impact indicators due to the combination of scenario and LCI data uncertainty. For a given pair of PDSs, if the ratio of the indicator is higher than 1, the PDS in the numerator has a higher environmental impact indicator than that of the PDS in the denominator, and vice versa for a comparison ratio value lower than 1. In Figure 23 to Figure 25, each comparison ratio is represented as an interval in which the top end represents the 95th percentile, the mark is the median and the bottom end is the 5th percentile. When the comparison ratio value is significantly different than 1 (with one sided, 95% confidence level), an asterisk (*) is placed by the interval. Details of the data on which Figure 23 to Figure 25 are based are presented in Tables 3 to 5 in the Appendix.

A first look at Figure 22 shows that when comparing PDS 1 with PDS 2 (left side), it appears that the system using aluminum cans has significantly higher ozone depletion potential (ODP) value than that of the system using PET bottles. When we compare PDS 1 with PDS 3 (middle part), the global warming potential (GWP) value of the system using PLA appears to be significantly lower than that using PET. Lastly, when we compare PDS 2 with PDS 3 (right side), the system using aluminum cans appears to have significantly higher ODP and GWP values than the system using PLA bottles. All other comparison ratios do not differ significantly. Figure 22 also shows that the widest uncertainty ranges correspond to water use and ODP ratios.

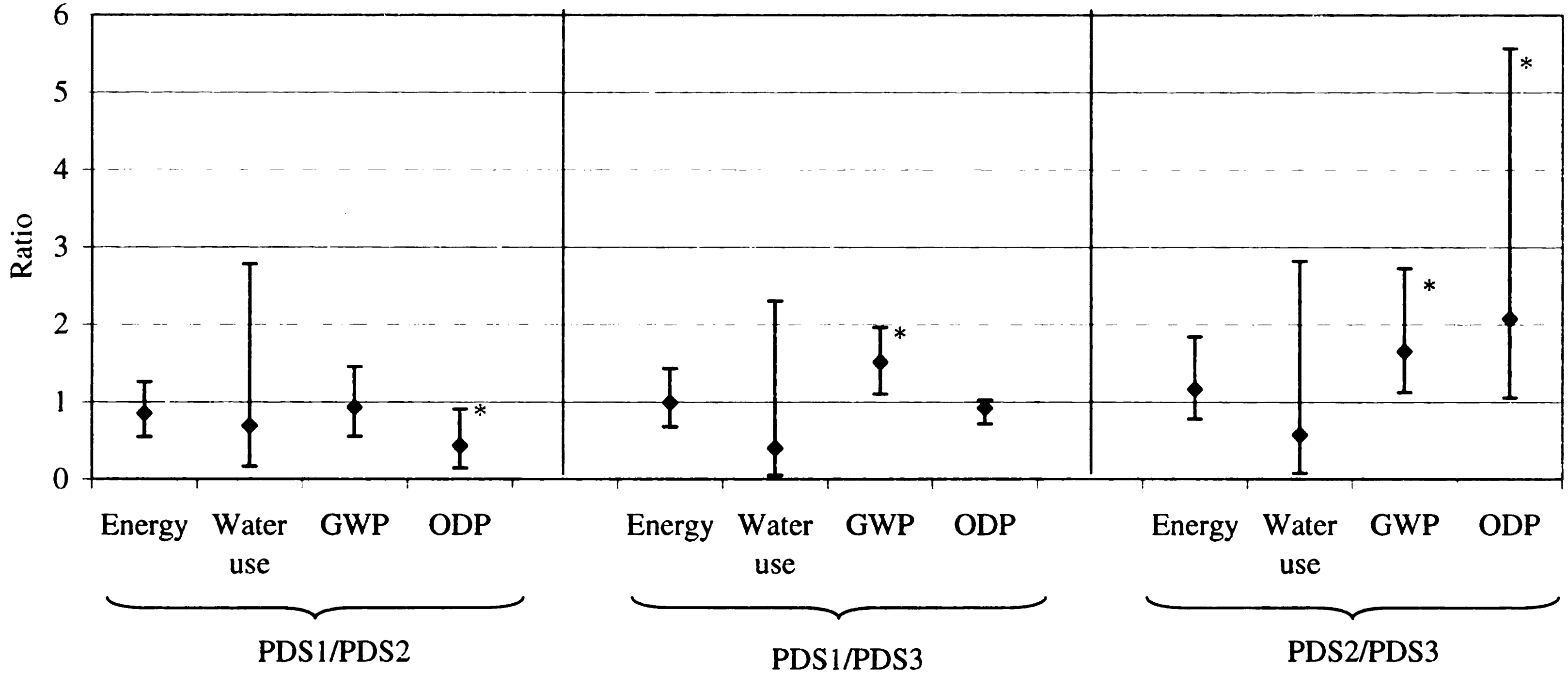


Figure 22. Overall uncertainty in the comparison ratios due to the combination of scenario and LCI data uncertainties (asterisk denotes significant difference from unity at 95% confidence level)

b) Scenario-specific comparative LCA results

In order to evaluate whether any comparison ratio differs significantly when scenario uncertainty is not included, or vice versa, the results of the specific scenarios are needed. All 8 scenarios were evaluated separately; Table 26 indicates their characteristics along with the notation code used in Figure 23 to Figure 25. In every evaluation of each scenario, LCI data uncertainty was included.

Table 26. Different scenarios evaluated in this study

Identification code of the scenario	Domain: Recycled content %	Domain: End of life: Recycling Rate (RR)	Domain: Distribution 1
A	PET: 0% Aluminum: 30% PLA: 0%	PET RR: 10% Aluminum RR: 19% PLA RR: 0%	PLA Distribution 1: 200 Km.
B	PET: 0% Aluminum: 30% PLA: 0%	PET RR: 10% Aluminum RR: 19% PLA RR: 0%	PLA Distribution 1: 2500 Km.
C	PET: 0% Aluminum: 30% PLA: 0%	PET RR: 39% Aluminum RR: 75% PLA RR: 10%	PLA Distribution 1: 200 Km.
D	PET: 0% Aluminum: 30% PLA: 0%	PET RR: 39% Aluminum RR: 75% PLA RR: 10%	PLA Distribution 1: 2500 Km.
E	PET: 0% Aluminum: 60% PLA: 0%	PET RR: 10% Aluminum RR: 19% PLA RR: 0%	PLA Distribution 1: 200 Km.
F	PET: 0% Aluminum: 60% PLA: 0%	PET RR: 10% Aluminum RR: 19% PLA RR: 0%	PLA Distribution 1: 2500 Km.
G	PET: 0% Aluminum: 60% PLA: 0%	PET RR: 39% Aluminum RR: 75% PLA RR: 10%	PLA Distribution 1: 200 Km.
H	PET: 0% Aluminum: 60% PLA: 0%	PET RR: 39% Aluminum RR: 75% PLA RR: 10%	PLA Distribution 1: 2500 Km.

Figure 23 to Figure 25 show the intervals of the comparison indicators that would have resulted if each specific scenario had been chosen. For reference purposes, the intervals for the combination of scenario and data uncertainty are shown as "All" in each Figure as well.

Inspection of Figure 23 to Figure 25 shows that the only cases in which an indicator appears not to agree with the overall uncertainty results (at the same confidence level at least) are the ODP ratios between the system using aluminum cans and that using PLA bottles (Figure 6d, scenarios G and H) in which aluminum recycled content is 60% and the end of life involves 75% and 10% recovery rates for Aluminum and PLA respectively. Nevertheless, the confidence level for the agreement in the ODP values between these systems in these scenario-specific results is still very high (94.5% and 94.7% for scenarios G and H respectively).

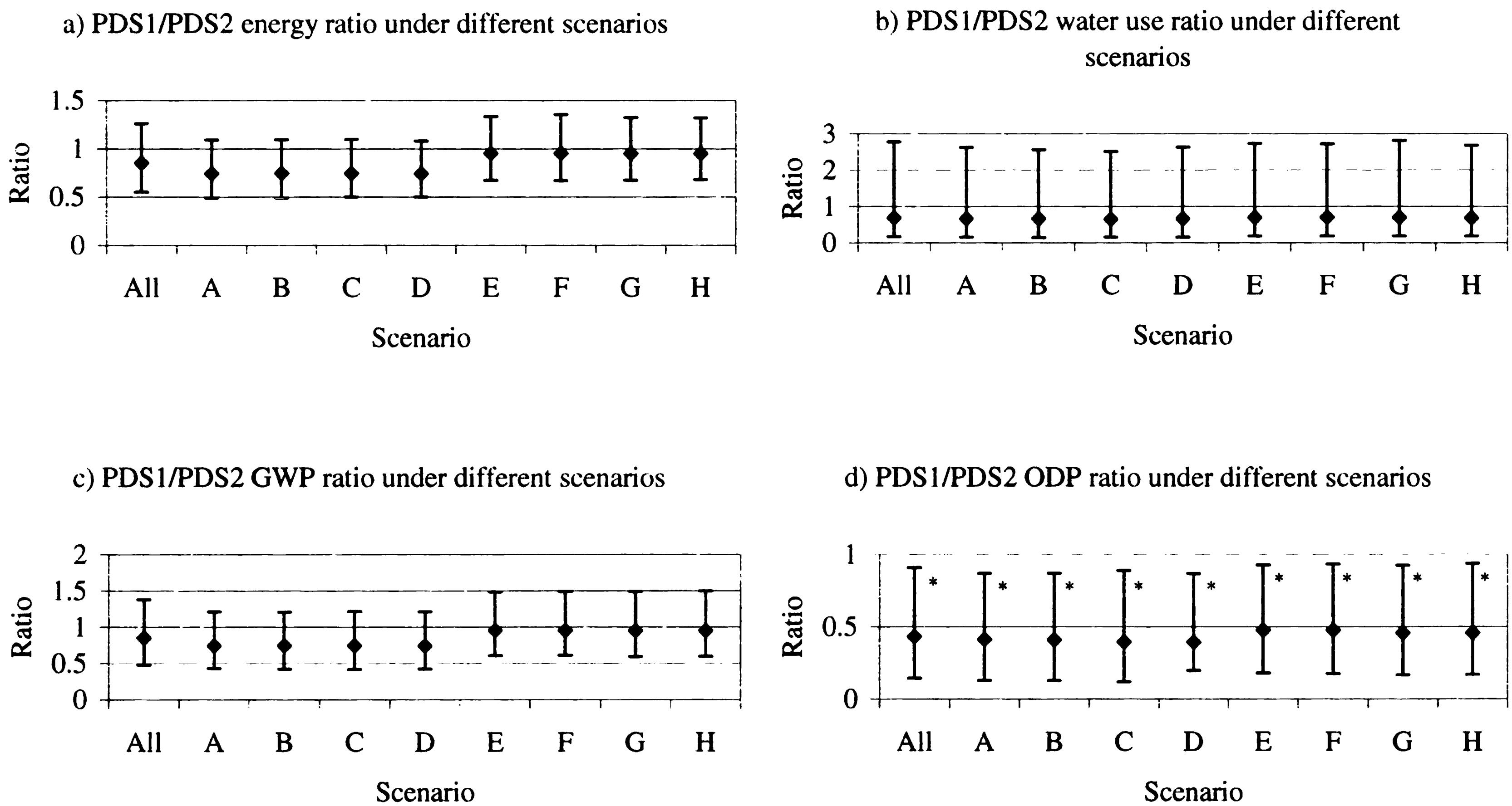


Figure 23. Scenario-specific results for the relationship PDS1/PDS2 (asterisk denotes significant difference from unity at 95% confidence level)

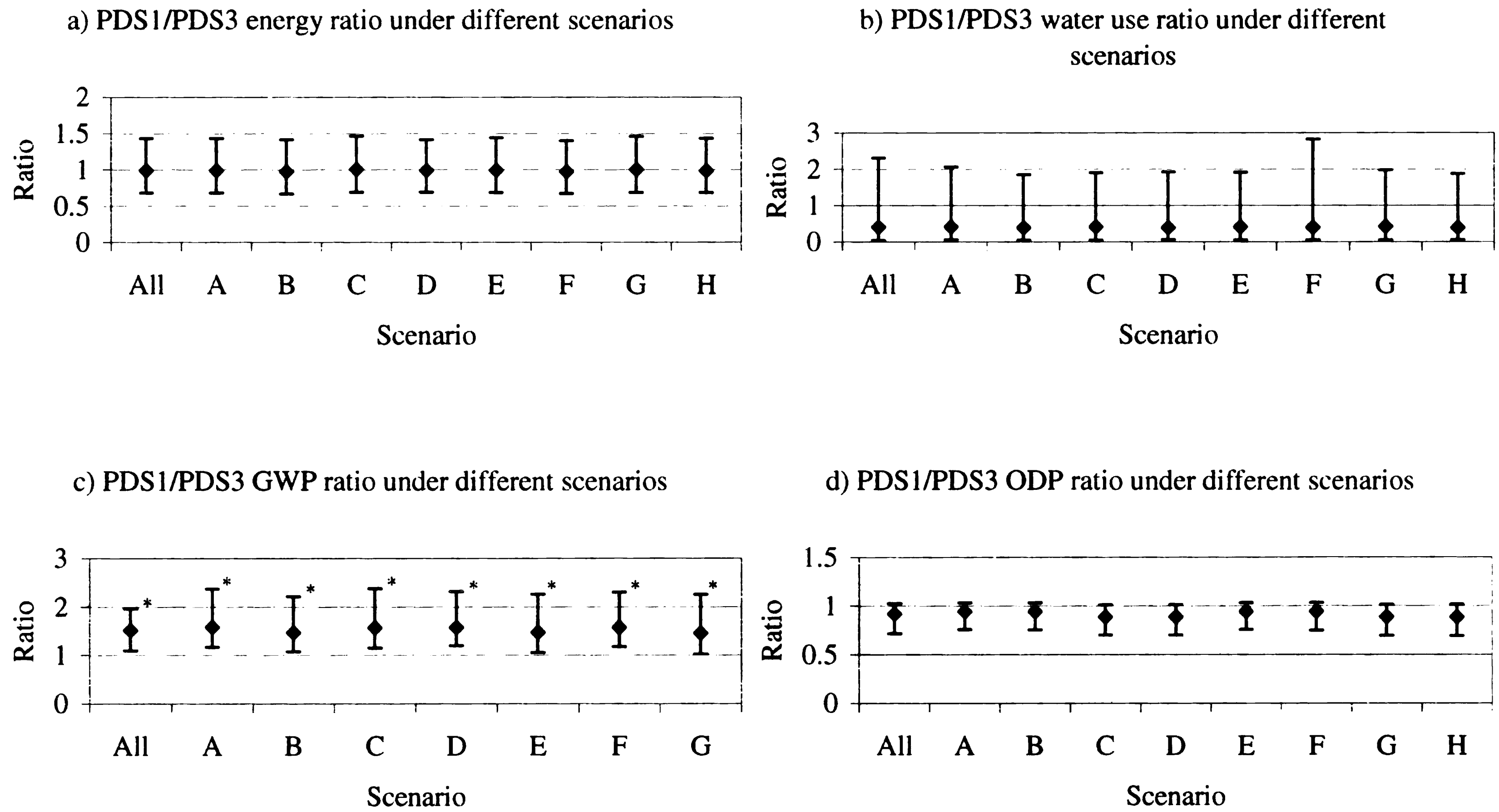


Figure 24. Scenario-specific results for the relationship PDS1/PDS3 (asterisk denotes significant difference from unity at 95% confidence level)

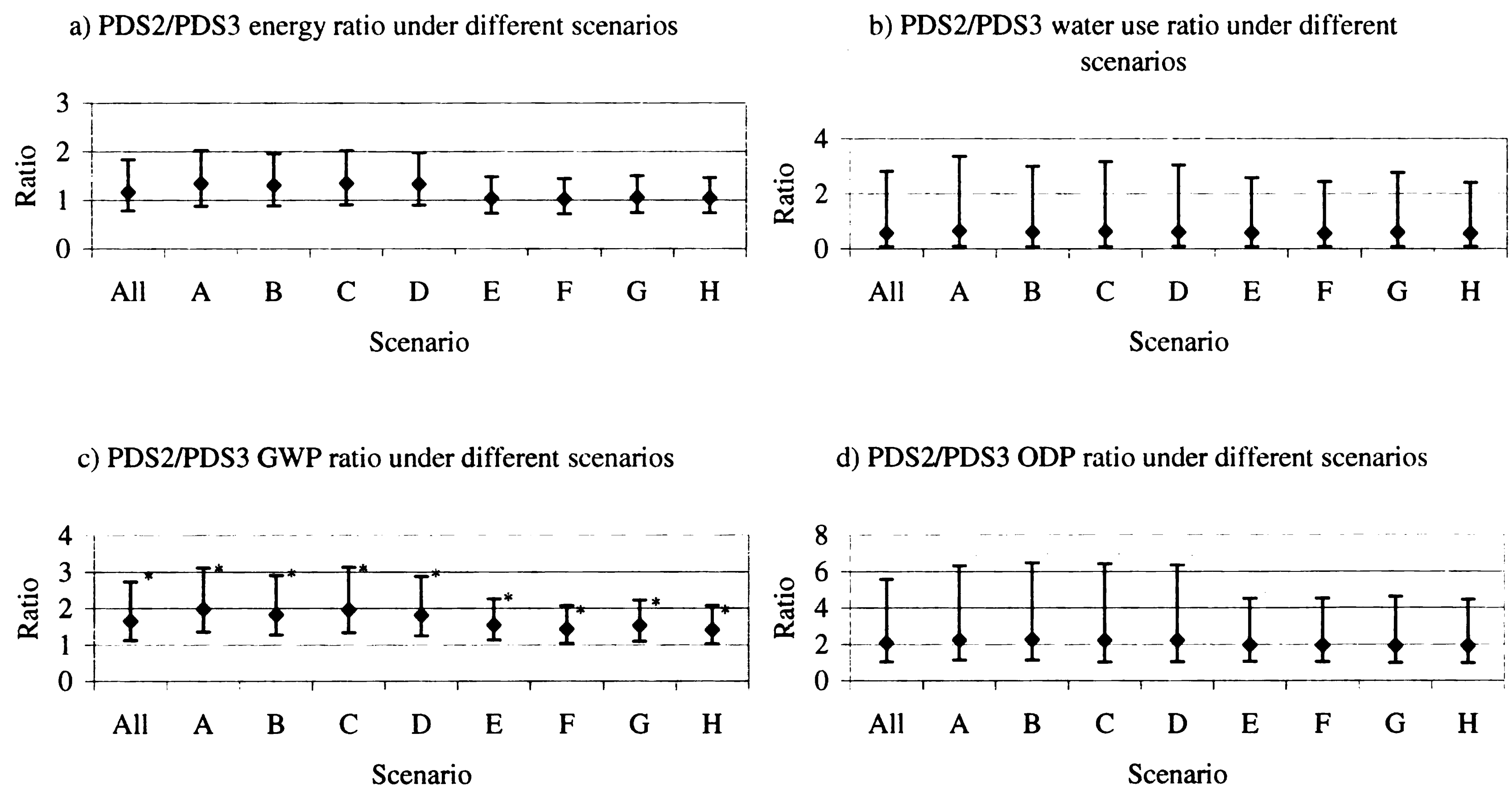


Figure 25. Scenario-specific results for the relationship PDS2/PDS3 (asterisk denotes significant difference from unity at 95% confidence level)

IV. Discussion

For organization purposes details of the data in which the figures included in this section are based are located in Tables 6 to 9 in the Appendix.

The present discussion is based on the results presented in the previous section and on the systems considered and assumptions made in Chapter Three of this dissertation. However, the aim of this section is not only to present a discussion based on the quantitative uncertainty assessment, but also to highlight other potential factors not included in the quantitative estimation of uncertainties in the comparative LCA.

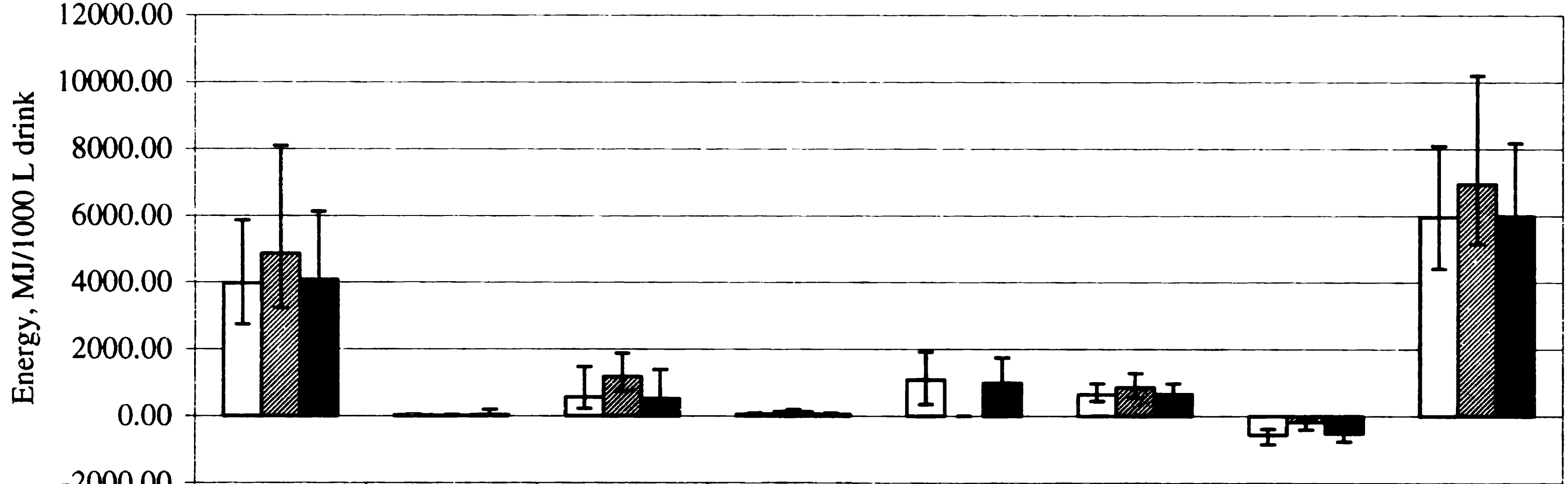
Looking at the results presented in the previous section, we can identify two important considerations. First is that LCI parameter uncertainty seems to have a dominant effect in the intervals of the LCA outcomes. In fact, most of the times the uncertainty intervals of scenario-specific results appear to be of similar magnitude to those in which scenario uncertainty is included. Nevertheless, in many cases scenario-specific results do seem to affect the spread and location of the median of the comparison ratio value, and this is further discussed in later paragraphs.

The second consideration is that most of the times the uncertainty intervals contain one, which in fact indicates that the systems under study are similar to one another, at least with regard to the environmental indicators included here. In fact, looking at the life cycle energy profile of the three systems (with all scenarios considered) shown in Figure 26, their similarities in median values and 95% confidence intervals are more noticeable. For instance, in the material production phase, the system using PLA bottles consumes per functional unit a median of 4080 MJ (68% of the total energy use), the system using PET bottles consumes per functional unit a median of 3978

MJ (66.8% of the total energy use), and the system using aluminum consumes per functional unit a median of 4858 MJ (69.8% of the total energy use) with a somewhat greater interval since scenarios containing 30% and 60% recycled content affected this phase of the system based on aluminum cans.

This last consideration is important since it ultimately means that the perceived preferability of one system over the other, at least in this case, is strongly dependent on the quality of the data (which defines the interval).

Further considerations can also be made by analyzing the scenario-specific results of Figure 23 to Figure 25, which reveal that several uncertainty ranges show systematic differences between the scenarios chosen. For instance, when comparing systems using aluminum cans, the effect of the use of recycled content is always noticeable since it generally lowers the energy use, and the emissions of global warming and ozone depleting substances due to material production and thus it helps move the median of the ratio closer to one (Figure 23a, c, d to Figure 25a, c, d,).

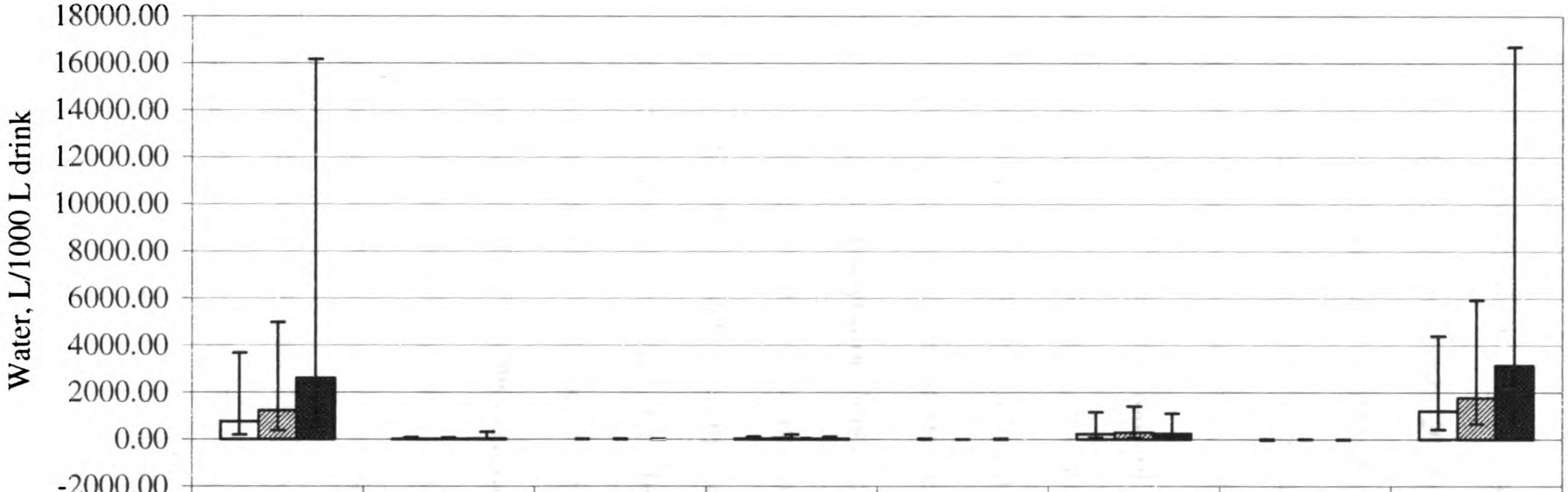


	Material Production	Distribution 1	Container comp. manufacture	Distribution 2	Filling	Distribution 3	End of life	Total
PDS1	3978.79	22.83	556.34	60.06	1084.04	651.37	-564.02	5955.71
PDS2	4858.18	14.94	1180.53	135.64	-0.29	849.78	-172.97	6954.1
PDS3	4080.39	29.73	524.28	58.63	987.26	655.15	-515.19	5997.49

Figure 26. Life cycle energy profiles of the three systems (values shown are medians)

Water use, in contrast, (Figure 23b to Figure 25b), does not seem to show this systematic variation. Instead, these graphs show large intervals in spite of the consistent appearance of a median value below one. This situation highlights the effect of the actual uncertainty analysis method (and the assumptions) on the LCA result. In fact, the large intervals for water use seem to occur because the uncertainty factor used for this parameter in this study was a conservative value of 10, at least 5 times bigger than that used for energy values and values for global warming and ozone depleting substances. More details of the effect of the magnitude of the uncertainty factor can be noticeable when we look at the life cycle water use profile shown in Figure 27.

As clearly shown, the largest uncertainty intervals belong to the material production phase, in which the system using PLA bottles has the greatest. This peculiar situation seems to occur by the concurrence of two factors: the large uncertainty factor used for this impact and the larger number of inventory "cradle to gate" modules used to model the corn growing and corn wet milling operations included in the material production phase of this system. That is, the more "cradle to gate" modules a system has, the larger the resultant uncertainty spread. Obviously this situation does not help in drawing a conclusion, in spite that, since it has a higher median, it may be perceived that the system using PLA uses more water.



	Material Production	Distribution 1	Container comp. manufacture	Distribution 2	Filling	Distribution 3	End of life	Total
PDS1	751.99	12.99	1.94	26.81	3.44	229.77	-1.42	1200.38
PDS2	1217.77	8.45	3.82	49.18	0	288.42	1.99	1777.48
PDS3	2596.71	20.95	1.75	25.72	2.94	232.45	-1.69	3132.8

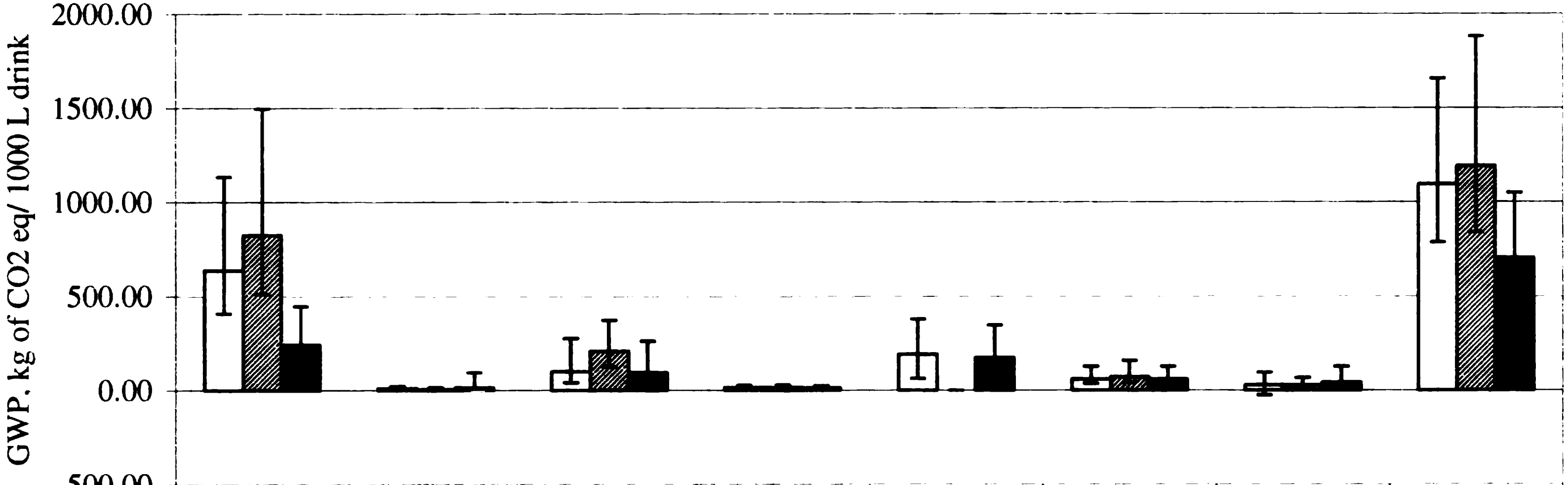
Figure 27. Life cycle water use profile of the three systems

On the other hand, the effect of end of life recycling rate in the scenarios considered is highlighted in Figure 24d. In fact, following estimations from the manufacturer of PLA (Vink et al., 2003), PLA incineration was modeled in this study using the same environmental burdens as for paper incineration (with energy recovery). Since the incineration energy is harnessed, the corresponding DEAM™ module credits (i.e. subtracting to account for the avoided burdens from generating electricity by burning fossil fuels) several of the emissions of the inventory data of the paper incineration. DEAM™ also applies this same reasoning to the module for plastic incineration with energy recovery used in this study to model the incineration of PET. However, the credit given to methyl bromide in the plastic incineration is almost three times higher than the credit given to paper incineration. Since methyl bromide is an ozone depleting substance, the effect of the previous difference can be noticeable when we analyze the ODP comparison indicators. In fact, by using the high recycling rate alternative (i.e. Aluminum RR: 75%, PET RR: 39% and PLA RR: 10%), there is less mass of PLA and PET sent for incineration. This is further affected by the fact that, since PLA is less dense than PET (Leaversuch, 2003) and following performance estimations from the PLA manufacturer (Vink et al., 2003) this study assumed that the weight of a PLA bottle is 6% less than that of a PET bottle. Less mass of any of these materials means less methyl bromide credits, but this effect is more important for PLA since quantity of this inventory value is three times smaller than that of PET. The overall effect would be then a comparison ratio moving away from one, which is exactly what happens in scenarios C, D, G and H in Figure 24d.

Though in this case a significant difference in the ODP ratio of PDS 1 and PDS 3 was not found, the appearance of substances with a high ODP contribution such as methyl bromide and halon 1301 in the incineration inventory data of a material such as PET needs to be further discussed, since this situation actually highlights the uncertainty in the allocation method used for emission data. In fact, it may be argued that there is no "chemical correlation" between PET incineration and halon 1301 emission for example, and thus its inclusion in the incineration inventory is not required, at least from a chemical standpoint. However, if for this incineration process, a mass allocation applied to the incineration facility is in fact used, the situation may well change with potentially great consequences for the calculated ODP of PET.

The significant differences regarding the GWP value between PDS 1 and PDS 3, and PDS 2 and PDS 3 indicated in Figure 22 also require a closer look. Inspecting Figure 28 which shows the life cycle GWP profile for the systems, it appears that the main reason for the higher GWP of the systems using the aluminum cans and PET bottles is because in their production phases the emissions of global warming substances are much higher than in the system using PLA.

In fact, in the material production phase, the comparative study shows that the system using PLA bottles emits, per functional unit, a median of 243 kg CO_2 eq (34.5% of its total), the system using PET bottles emits per functional unit a median of about 639 kg CO_2 eq (58.3% of its total) and the system using aluminum emits per functional unit a median of 824 kg CO_2 eq (69.3% of its total), with again a somewhat greater interval since scenarios containing 30% and 60% recycled content affected this phase of the system based on aluminum cans.



	Material Production	Distribution 1	Container comp. manufacture	Distribution 2	Filling	Distribution 3	End of life	Total
PDS1	638.86	10.78	100.12	14.03	191.88	59.91	30.11	1094.52
PDS2	824.99	6.97	207.86	15.49	0.03	70.17	25.32	1190.32
PDS3	243.25	16.14	94.7	13.22	172.16	59.42	43.49	704.03

Figure 28. Life cycle GWP profile of the three systems

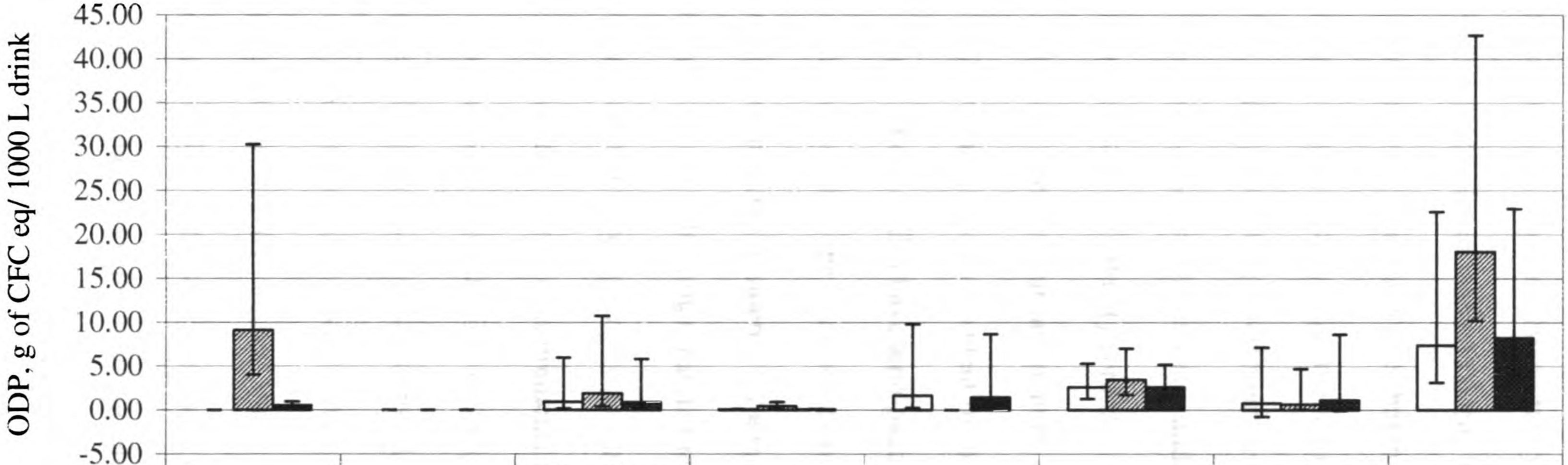
Further analysis of the material production phase of these systems shows that the main GWP contributors in this phase are carbon dioxide and nitrogen oxides that are emitted in the extraction, processing of intermediates and any related transportation of the raw materials as they were modeled in the DEAM™ databases. It is important to emphasize now that the differences found in Figure 3 with regards to GWP correspond to the eight scenarios previously considered, and an alternative scenario may actually give different results. Thus, for instance, if an aluminum can with 100% recycled content is used in the comparison, the GWP of the material production phase of this system drops dramatically to 244 kg CO_2 eq, almost the same value as that of the system using PLA bottles.

The importance of the use (and its inclusion in the comparative LCA study) of packaging components is also highlighted in this analysis as well. For instance, the effect of not including the LDPE sleeve in the system using PET bottles roughly reduces its energy use by 89 MJ, while not including the PLA sleeve in the system using PLA bottles reduces its energy use by about 90 MJ. This difference becomes critical when systems are similar, and quickly shows us why, even though aluminum is generally more energy intensive, the fact that it includes less components may allow it to be comparable to systems which, though less energy intensive when considered individually, require more components overall.

The significant differences regarding the ODP value between PDS 2 and PDS 3 indicated in Figure 22 require a closer look as well. The contributing substances to the ozone depletion potential identified in this study are: carbon tetrachloride, halon 1301, methyl bromide and methyl chloride. Inspecting Figure 29 which shows the life cycle

ODP profile of the systems, it appears that the main reason for the higher ODP of the system using the aluminum can is because in the production phase of this system the emissions of ozone depleting substances are much higher than in other systems.

Further analysis of the material production phase of the system using aluminum indicates that the main ODP contributors in this phase are halon 1301 and methyl bromide, which actually account for 51% (at 60% recycled content) to 56% (at 30% recycled content) of the total ODP of this system. However, it should be noted that halon 1301 and methyl bromide are no longer produced in the U.S. and their use is heavily restricted (U.S. EPA, EPA guidance on Halon emission, 2001; U.S. EPA, Ozone depletion rules & regulations, 2005). It would be expected, then, that updates in data would reflect significant reductions in emissions of ozone depleting substances, and this in turn suggests that the ODP values of these three systems may actually appear similar to one another. Moreover, this situation underlies another aspect of the LCA result, which is its dependence on the location of the operation. For instance, as stated earlier in Chapter 3, alumina production for aluminum manufacture is mostly done outside the U.S. in countries such as Australia, Jamaica or Brazil, in some of which restrictions in the use of these ozone depleting substances may not yet be in place. Then again, updates in data with regard to origin of the operations may change back the standing of the system using aluminum cans.



	Material Production	Distribution 1	Container comp. manufacture	Distribution 2	Filling	Distribution 3	End of life	Total
PDS1	0.00	0.00	0.93	0.08	1.64	2.62	0.77	7.35
PDS2	9.12	0	1.9	0.45	0	3.49	0.65	17.99
PDS3	0.52	0	0.87	0.08	1.46	2.64	1.14	8.19

Scenario

Figure 29. Life cycle ODP profile of the three systems

In Figure 29 the consequences of the uncertainty analysis itself requires a discussion as well. The large uncertainty spread in the material production phase for the system using aluminum seems to occur because of the effect of the uncertainty factors of the "contributor" parameters (i.e. halon 1301, methyl chloride, etc) of the ODP value. Ultimately, this is clearly a problem for characterized impact indicators since they are usually compilations (e.g. weighted sums) of several contributor items, which in turn have their own inherent uncertainty, and so the more inventory parameters compiled, the larger the resultant uncertainty spread.

In fact, we can further ponder on the implications of the uncertainty analysis results shown here. As was evident from the previous quantitative assessment of the LCA uncertainty, uncertainties have a propagating effect and do affect the LCA outcome. This situation also implies the somewhat counterintuitive idea that the more the number of inventory parameters considered for a system in a study, the more (not less) uncertain the result may be. On the other hand, the other extreme is no better. For instance, for practical reasons the compiling of just major inventory parameters (i.e. based on emitted weight, feasibility of the measurement) is not unusual. However, if traces of substances that have a large contribution on a characterized impact (e.g. ODP, or GWP) are neglected, the conclusions can potentially change. This, in fact, may be a problem when "newer" technologies, for which only limited and proprietary information exist, are compared with heavily studied ones. For instance, with regard to the information gathered for PLA in the comparative LCA presented in Chapter Three, polymerization data was obtained from the manufacturer of this resin, which in turn used engineering estimations to account for just selected inventory parameters while other substances were neglected. Therefore, the results for ODP or GWP or any other characterized indicator should be interpreted acknowledging this situation.

Lastly, it needs to be clarified at this point that values of the energy analysis presented here are total energy values, this is, the summation of both renewable as well as non-renewable energy. This note is important since the use of annually renewable biomass (e.g. corn), as opposed to petrochemicals (i.e. oil or natural gas) as the feedstock for the production of polymers is one of the core reasons for the desire to develop PLA or any other biobased polymer. In fact, based on the assumptions of this study, about 26% of the total PLA energy consumption came from renewable resources, while the other systems had smaller renewable energy consumption percentages (about 11% and 0.3% for aluminum and PET, respectively). Nevertheless the percentage of renewable resources consumed in the systems based on aluminum and PET can still greatly change if we use, for instance, location-specific energy production data which may produce very different conclusions than the average electricity generation values used here. Clearly, results may differ when coal-generated electricity data is used for operations that occur in regions in which electricity is produced mainly by hydroelectric power. Coal-generated energy is considered "non-renewable" while hydroelectric power is produced from "renewable" resources and thus appears to be more environmentally friendly.

V. Conclusions

The present study has shown the use of a simultaneous assessment of uncertainty analysis due to LCI data and scenarios relevant to packaging.

The results presented in this study have shown the important consideration that LCI parameter uncertainty seems to have a dominant effect in the outcome of the LCA results considered. Furthermore, in most cases the uncertainty intervals contained unity, which in fact indicated that the systems under study are similar to one another, at least with regard to the environmental indicators included here.

It was also shown that location and age of data and allocation procedures, along with specific packaging scenarios, all affect the impact assessment method and confuse the outcome for results that originally indicated significant differences among the systems compared.

The implications of these considerations are important since it shows that though the systems compared are based on the use of different feedstock materials, uncertainty in the LCI data and the scenarios interfere with determining conclusive differences.

Further improvements in the application of the methodology are needed, including the development of a life-cycle inventory database with uncertainty information, and the further development of location specific and time specific impact assessment models for a more systematic analysis of scenario and model uncertainty.

CHAPTER 5 OVERALL CONCLUSIONS, RECOMMENDATIONS AND FUTURE WORK

In this dissertation, a thorough understanding of the complexities of comparative packaging based life cycle assessment (LCA) outcomes was sought. To achieve this, the research was divided into two major parts. The first part involved a critical review of the literature with regards to LCA and its applications to packaging operations, which was presented in Chapter Two. The main bodies of literature reviewed in this chapter involved actual LCA studies, recent journal publications regarding advances in LCA methodology and case studies, ISO 14000 series of standards, reports from governmental agencies and private sector literature.

As shown in Chapter Two, the LCA method covers a great variety of aspects of packaging systems and has been used to provide information for purposes such as strategic planning, marketing, academia, environmental labeling, etc. However, uncertainties play a major role on the LCA method throughout the phases of the study. Uncertainties in the quality (e.g. age, source, completeness) of the data used in the study, uncertainty in the scope and boundaries of the processes included and uncertainty in the scope of the impact assessment stage are some of the most critical ones.

The second part of the research comprised the actual quantitative study of the effect of some types of uncertainty in the outcome of a packaging based LCA. For this, a two-step process was followed. First, a comparative LCA study featuring three different packaging materials was developed for a hypothetical drink product. Following ISO's LCA steps, a comparative LCA study was developed and its extensive details and information sources were presented in Chapter Three. The three material/container alternatives were a PET bottle, a PLA bottle and an aluminum can, which along with their whole set of distribution packaging (i.e. corrugated trays, stretch wrap, and pallets), transportation services and end of life alternatives were evaluated with regards to four environmental burdens: energy use, water use, global warming potential (GWP) and ozone depletion potential (ODP).

The second step was the actual quantitative uncertainty analysis of the comparative LCA presented in Chapter Three. The uncertainty analysis comprised two aspects. First, a scenario analysis was proposed by identifying different domains with effects on packaging situations. Under each domain two alternatives were proposed and a procedure was used to select an alternative from each domain and randomly create different scenarios. The second aspect of the uncertainty analysis comprised the inventory parameter uncertainty. This analysis was achieved by the use of appropriate published uncertainty estimations along with stochastic simulation (i.e. Monte Carlo simulation) in order to represent inventory data uncertainty as probability distributions. Relevant inventory parameters were identified and further analyzed, and the final results of the comparative LCA along with the combined uncertainties from inventory data and scenarios were obtained. A discussion of those results was presented in Chapter Four.

Several conclusions can be drawn from these last two chapters. For instance, the procedure for performing the LCA according to ISO's recommendations was both time consuming, and data intensive and heavily relied on gathering of quality data. However, this comparative LCA is still a streamlined version and thus, its results and conclusions need to be handled acknowledging such limitation. Nevertheless, its average results are in line with a similar comparison study performed by Franklin Associates (The environmental impact of softdrink delivery systems, 1995). This is, the average energy values of PET and Aluminum packaging systems are within the confidence interval of the present work (e.g. PET system uses 6140 MJ/1000 L drink at 36% recycling rate; soft drink; 6698 MJ/1000 L drink at 62% recycling rate). This situation suggests a point in favor towards the accuracy of the LCA result of comparative studies, since the data sources used for this and the aforementioned study can well be considered different.

However, as explained in Chapter One, this dissertation investigated the hypothesis that when some uncertainties inherent to the methodology (i.e. data used) and the nature of industrial processes (i.e. scenarios) are considered, the current life cycle assessment approach does not provide enough evidence to rule out (or favor) one alternative versus another when packaging systems are studied and compared based on their environmental performance.

As shown in Chapter Four, the simultaneous assessment of uncertainty due to LCI data and scenarios relevant to packaging has demonstrated that, in this particular comparison, LCI parameter uncertainty seems to have a dominant effect in the outcome of the LCA results considered. Furthermore, in most cases the uncertainty intervals indicated that, despite the differences in the primary raw materials, the systems under study can be similar to one another, at least with regard to the environmental indicators included here, by including the aforementioned noise, or considering temporal or spatial information or allocation procedures.

Based on these results, it can be seen that the hypothesis of this study holds true especially when the knowledge of the systems under study is limited to average values of the different operations considered. This also confirms that when more information is known about how, where and when the system operates (i.e. location and time of operations, type of technology, raw material quality) more conclusive results and conclusions can be obtained. For instance, it was seen that if the alumina extraction operations were in fact performed in the U.S., and with more updated information regarding emissions that affect ozone depletion (i.e. carbon tetrachloride), the ODP value of aluminum would have been much lower and comparable to those of PLA or PET.

Not having a priority-based approach for performing the impact assessment study could be also detrimental when interpreting the results. In fact, the inclusion of different environmental indicators might as well affect the overall outcome, since there is no agreed basis for the expression "environmental performance" which in fact could encompass any set of environmental indicators. The survey presented in Table 10 of Chapter Two of this dissertation has shown that comparison studies usually limit their scope to listing inventory parameters such as energy use and some air emissions. That approach, naturally followed because that kind of data is often more readily available and thus saves time, indirectly avoids the added complexity of estimating environmental impact indicator scores. Nevertheless, the problem still remains as to what inventory value or impact indicator is more important.

For instance in this dissertation, considering the ODP and GWP values of PET and PLA, we see that their median values presented in Chapter Four are very much the same for ODP values, but when GWP is considered, PET has a higher emission profile. If the median water use profile is included, this study showed that the PLA system would consume more water than PET. In such a case then, a logical systematic approach that can be followed is the use of a priority-based system that can help assign weights to the different impacts. In turn, the development of such priority-based methods is the subject of several publications, and by allowing user decisions has the perceived disadvantage of subjective assessments.

I. Recommendations for future work

A fundamental understanding of the interrelationships governing the results of comparative LCAs was investigated in this research. Critical considerations affecting the LCA results were identified. A methodology was used in order to help structure the uncertainty analysis, aggregating the different types of uncertainty, and identifying the most important sources of uncertainty. However, there are many areas that need to be investigated further.

The following recommendations can be made to improve the LCA method and to further the understanding of situations with regard to their environmental footprint:

1. Improvements in both the reduction of uncertainty in the LCI information, and the development of LCI databases with uncertainty information, are needed. This is especially important for new data and new measurements. Though qualitative in nature, the use of data quality goals (DQG) can help provide a means for such uncertainty reduction. Further development of location specific and time specific databases and impact assessment models for a more systematic analysis of scenario and model uncertainty are needed.
2. Application of similar uncertainty studies to other packaging situations involving different scenarios, products and packaging formats will be required. Moreover, uncertainty considerations in other aspects of the LCA methodology (e.g. functional unit selection) and its effect on conventional as well as future packaging concepts and functions (e.g. degradability, traceability, sensing), need to be investigated.

3. Inventory databases will need to be developed not only for "conventional" packaging materials and components but for new packaging materials and concepts as well. Details and more thorough technical data from new technologies (e.g. PLA polymerization) will be needed to provide the clearest environmental picture possible. Failure to do so will reward the least known technology with a supposed environmental preferability.

APPENDIX

Table A. 1. Uncertainty factors (k) for categories of parameters used in the screening for relevant variables

Item	k factor*	Distribution
Resources used		
Central resources	2	Log-normal
Non-central resources	10	Log-normal
Emissions		
CO_2, SO_2 (calculated from corresponding inflows)	2	Log-normal
Other energy related air emissions	10	Log-normal
Other process specific air emissions	10	Log-normal

* values obtained from Finnveden (Finnveden and Lindfors, 1998) and Huijbregts et al (Huijbregts et al., 2003)

Table A. 2. Uncertainty factors and distributions of parameters with at least 1% contribution to the variance of overall LCA outcomes

Item	Characteristics	Distribution
PET bottle weight, Kg	k factor: 1.01*	Log-normal
Drink volume (bottle), Kg	(16, 16.16)**	Normal
Drink volume (can), Kg	(12, 12.12)**	Normal
Injection molding energy	k factor: 3.02***	Log-normal
Can making energy	k factor: 1.01***	Log-normal
Halon 1301 in extraction and energy generation processes (i.e. Al_2O_3 production, electricity generation)	k factor: 2.3****	Log-normal
NO_X in energy generation processes (i.e. electricity, natural gas extraction)	k factor: 3****	Log-normal
NO_X due to road transport processes (i.e. service by 16 ton and 40 ton truck)	k factor: 2.9****	Log-normal
Methane air emission as leakage (i.e. PET and PLA polymerization, aluminum production)	k factor: 1.1****	Log-normal

* Own estimations based on review of commercial literature

** Own estimation assuming a threshold of 3 standard deviations. Upper and lower 3 sigma values shown.

*** value obtained from SAEFL report (LCI for packagings. Vol. II., 1998b)

**** value obtained from Huijbregts et al (Huijbregts et al., 2003)

I Visual basic code used for scenario analysis procedure

```
Sub Macro1()
'
' Macro1 Macro
' Macro recorded 7/17/2005 by Dario Martino
'
Dim A
Dim B
Dim C
Randomize [Timer]
A = Rnd
If A >= 0.5 Then
    Windows("PDS31.xls").Activate
    Sheets("Distribution 1").Range("C8") = "2500"
    Else
    Windows("PDS31.xls").Activate
    Sheets("Distribution 1").Range("C8") = "200"
    End If
B = Rnd
If B >= 0.5 Then
   Windows("PDS31.xls").Activate
   Sheets("Material production").Range("E6") = "0"
   Windows("PDS21.xls").Activate
   Sheets("Material production").Range("E8") = "30"
   Windows("PDS11.xls").Activate
   Sheets("Material production").Range("E6") = "0"
    Else
   Windows("PDS31.xls").Activate
   Sheets("Material production").Range("E6") = "0"
   Windows("PDS21.xls").Activate
   Sheets("Material production").Range("E8") = "60"
   Windows("PDS11.xls").Activate
   Sheets("Material production").Range("E6") = "0"
    End If
C = Rnd
If C >= 0.5 Then
   Windows("PDS11.xls").Activate
   Sheets("End of life").Range("D3") = "10"
   Windows("PDS21.xls").Activate
   Sheets("End of life").Range("D3") = "19"
   Windows("PDS31.xls").Activate
   Sheets("End of life").Range("D3") = "0"
   Else
   Windows("PDS11.xls").Activate
   Sheets("End of life").Range("D3") = "39"
   Windows("PDS21.xls").Activate
   Sheets("End of life").Range("D3") = "75"
   Windows("PDS31.xls").Activate
   Sheets("End of life").Range("D3") = "10"
   End If
End Sub
```

IIResults of the quantitative uncertainty analysis

Table A. 3. Scenario-specific interval results for the relationship PDS1/PDS2

All

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.85	1.26	0.41	0.55	0.3	no
	Water use	0.69	2.78	2.09	0.17	0.52	no
	GWP	0.9266	1.4532	0.5266	0.5528	0.3738	no
	ODP	0.4319	0.9087	0.4768	0.1447	0.2872	Yes

Scenario A

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.74	1.09	0.35	0.49	0.25	no
	Water use	0.66	2.63	1.97	0.15	0.51	no
	GWP	0.8063	1.2781	0.4718	0.4984	0.3079	no
	ODP	0.4131	0.8697	0.4566	0.1311	0.282	yes

Scenario B

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.75	1.09	0.349777	0.49	0.256567	no
	Water use	0.67	2.56	1.894987	0.15	0.520342	no
	GWP	0.8080	1.2691	0.461101	0.4877	0.320235	no
	ODP	0.4095	0.8705	0.461019	0.1286	0.280883	yes

Scenario C

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.74	1.10	0.351927	0.50	0.241326	no
	Water use	0.65	2.52	1.869611	0.15	0.501137	no
	GWP	0.8074	1.2783	0.470934	0.4853	0.322081	no
	ODP	0.3955	0.8900	0.494449	0.1203	0.275226	yes

Scenario D

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.74	1.08	0.34	0.5	0.24	no
	Water use	0.66	2.64	1.98	0.15	0.51	no
	GWP	0.8073	1.2793	0.472	0.4916	0.3157	no
	ODP	0.3929	0.8681	0.4752	0.1999	0.193	yes

Scenario E

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.95	1.34	0.383011	0.67	0.27801	no
	Water use	0.70	2.73	2.034543	0.19	0.507888	no
	GWP	1.0264	1.5573	0.530945	0.6831	0.343271	no
	ODP	0.4758	0.9274	0.451577	0.1823	0.293524	yes

Scenario F

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.95	1.35	0.40374	0.67	0.280436	no
	Water use	0.70	2.72	2.021116	0.19	0.512914	no
	GWP	1.03139169	1.5733	0.541882	0.695243182	0.336149	no
	ODP	0.47658811	0.9324	0.455764	0.178472687	0.298115	yes

Scenario G

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.95	1.32584727	0.377787	0.674190503	0.27387	no
	Water use	0.70	2.82054096	2.1236	0.187439779	0.509501	no
	GWP	1.0350	1.57705584	0.542031	0.680428214	0.354596	no
	ODP	0.4570	0.92586769	0.46891	0.16769609	0.289261	yes

Scenario H

PDS 1 vs PDS2		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.95	1.32	0.370207	0.68	0.267389	no
	Water use	0.69	2.67	1.986353	0.19	0.501484	no
	GWP	1.03315513	1.58352526	0.55037	0.682816785	0.350338	no
	ODP	0.45703654	0.9345447	0.477508	0.169693439	0.287343	yes

Table A. 4. Scenario-specific interval results for the relationship PDS1/PDS3

Scenario All

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.99	1.43	0.44	0.68	0.31	No
	Water use	0.4	2.3059	1.9059	0.05	0.35	No
	GWP	1.5124	1.96	0.4476	1.0998	0.4126	Yes
	ODP	0.9181	1.0244	0.1063	0.7165	0.2016	No

Scenario A

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.99	1.43	0.44	0.68	0.31	no
	Water use	0.41	2.06	1.65	0.06	0.35	no
	GWP	1.5752	2.3719	0.7967	1.1714	0.4038	yes
	ODP	0.9433	1.0331	0.0898	0.7598	0.1835	no

Scenario B

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.97	1.41	0.44	0.67	0.30	no
	Water use	0.39	1.85	1.461654	0.05	0.332677	no
	GWP	1.4640	2.2147	0.750673	1.0812	0.382823	yes
	ODP	0.9421	1.0313	0.08923	0.7554	0.186735	no

Scenario C

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.00	1.46	0.46	0.69	0.31	no
	Water use	0.41	1.90	1.496007	0.05	0.354632	no
	GWP	1.5625	2.3793	0.816795	1.1496	0.412949	yes
	ODP	0.8848	1.0115	0.12671	0.7006	0.184135	no

Scenario D

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.99	1.41	0.42	0.69	0.3	no
	Water use	0.39	1.92	1.53	0.06	0.33	no
	GWP	1.4453	2.1899	0.7446	1.0678	0.3775	yes
	ODP	0.8849	1.0127	0.1278	0.6995	0.1854	no

Scenario E

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.99	1.44	0.45	0.68	0.31	no
	Water use	0.41	1.91	1.498814	0.05	0.355298	no
	GWP	1.5723	2.3642	0.791854	1.1591	0.413205	yes
	ODP	0.9427	1.0340	0.091318	0.7583	0.18432	no

Scenario F

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.97	1.40	0.427719	0.67	0.301291	no
	Water use	0.39	2.82	2.425947	0.06	0.337271	no
	GWP	1.471083934	2.1989	0.7278	1.078263107	0.392821	yes
	ODP	0.942525948	1.0323	0.08973	0.74777481	0.194751	no

Scenario G

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.00	1.458332093	0.457097	0.689383601	0.311851	no
	Water use	0.42	1.976645174	1.554277	0.056452743	0.365916	no
	GWP	1.5710	2.376320845	0.805328	1.142324443	0.428669	yes
	ODP	0.8855	1.011939197	0.126399	0.695194422	0.190345	

Scenario H

PDS 1/PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	0.99	1.43	0.45	0.69	0.30	no
	Water use	0.39	1.87	1.483024	0.05	0.336426	no
	GWP	1.4517	2.203100239	0.751395	1.065711539	0.385993	yes
	ODP	0.8837	1.016803825	0.133104	0.69078505	0.192915	no

Table A. 5. Scenario-specific interval results for the relationship PDS1/PDS3

All

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.16	1.84	0.68	0.78	0.38	no
	Water use	0.57	2.82	2.25	0.08	0.49	no
	GWP	1.6518	2.724	1.0722	1.1233	0.5285	Yes
	ODP	2.0762	5.5667	3.4905	1.0563	1.0199	Yes

Scenario A

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.34	2.02	0.68	0.88	0.46	no
	Water use	0.65	3.36	2.71	0.09	0.56	no
	GWP	1.9765	3.1086	1.1321	1.3619	0.6146	yes
	ODP	2.2285	6.3081	4.0796	1.1419	1.0866	yes

Scenario B

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.30	1.96	0.652798	0.88	0.419601	no
	Water use	0.61	3.00	2.393281	0.09	0.522199	no
	GWP	1.8249	2.8995	1.074626	1.2725	0.55241	yes
	ODP	2.2565	6.4841	4.22756	1.1348	1.121762	yes

Scenario C

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.35	2.02	0.670103	0.91	0.44056	no
	Water use	0.63	3.17	2.544903	0.09	0.541021	no
	GWP	1.9576	3.1275	1.169853	1.3399	0.617703	yes
	ODP	2.2095	6.4191	4.209585	1.0350	1.174521	yes

Scenario D

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.33	1.98	0.65	0.9	0.43	no
	Water use	0.61	3.05	2.44	0.09	0.52	no
	GWP	1.8101	2.8776	1.0675	1.2454	0.5647	yes
	ODP	2.2196	6.3531	4.1335	1.0589	1.1607	yes

Scenario E

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.04	1.48	0.442204	0.73	0.305302	no
	Water use	0.58	2.59	2.008596	0.08	0.497239	no
	GWP	1.5440	2.2596	0.715639	1.1306	0.413409	yes
	ODP	1.9532	4.5243	2.571071	1.0730	0.880167	yes

Scenario F

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.02	1.44	0.418628	0.72	0.306246	no
	Water use	0.56	2.44	1.881807	0.08	0.474985	no
	GWP	1.43354428	2.069756	0.636212	1.0445216	0.389023	yes
	ODP	1.9487205	4.515195	2.566475	1.0665454 53	0.882175	yes

Scenario G

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.06	1.502824	0.445877	0.7474856 35	0.309461	no
	Water use	0.60	2.774471	2.178857	0.0806949 11	0.514919	no
	GWP	1.5235	2.227837	0.704369	1.0990384 25	0.424429	yes
	ODP	1.9110	4.612746	2.701768	0.9968852 75	0.914093	no

Scenario H

PDS 2 vs PDS3		Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end	Significantly different than 1?
	Energy	1.04	1.46	0.420986	0.74	0.304739	no
	Water use	0.56	2.40	1.837566	0.08	0.478341	no
	GWP	1.40781838	2.073229	0.665411	1.028765753	0.379053	yes
	ODP	1.91125551	4.45282	2.541564	0.983469388	0.927786	no

Table A. 6. Life cycle results for PDS1

Material Production

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	3978.79	5853.64	1874.85	2740.74	1238.05
Water use	751.99	3666.83	2914.84	194.23	557.76
GWP	638.86	1134.71	495.85	407.72	231.14
ODP	0.00	0.00	0.00	0.00	0.00

Distribution 1

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	22.83	43.97	21.14	11.94	10.89
Water use	12.99	94.49	81.50	1.80	11.19
GWP	10.78	22.63	11.85	5.73	5.05
ODP	0.00	0.01	0.01	0.00	0.00

Component manufacturing

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	556.34	1476.91	920.57	225.92	330.42
Water use	1.94	15.57	13.63	0.23	1.71
GWP	100.12	276.65	176.53	40.42	59.70
ODP	0.93	6.00	5.07	0.19	0.74

Distribution 2

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	60.06	83.95	23.89	45.00	15.06
Water use	26.81	119.71	92.90	7.69	19.12
GWP	14.03	26.60	12.57	8.21	5.82
ODP	0.08	0.16	0.08	0.04	0.04

Filling

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	1084.04	1915.77	831.73	357.62	726.42
Water use	3.44	25.72	22.28	0.34	3.10
GWP	191.88	378.10	186.22	62.33	129.55
ODP	1.64	9.79	8.15	0.24	1.40

Distribution 3

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	651.37	976.34	324.97	443.67	207.70
Water use	229.77	1160.30	930.53	61.57	168.20
GWP	59.91	127.88	67.97	35.85	24.06
ODP	2.62	5.30	2.68	1.31	1.31

End of life

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	-564.02	-380.04	183.98	-838.86	274.84
Water use	-1.42	7.15	8.57	-21.94	20.52
GWP	30.11	94.57	64.46	-26.10	56.21
ODP	0.77	7.14	6.37	-0.72	1.49

Total

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	5955.71	8086.07	2130.36	4419.80	1535.91
Water use	1200.38	4387.42	3187.04	417.06	783.32
GWP	1094.52	1655.33	560.81	787.54	306.98
ODP	7.35	22.60	15.25	3.11	4.24

Table A. 7. Life cycle results for PDS2

Material Production

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	4858.18	8088.68	3230.50	3229.52	1628.66
Water use	1217.77	4973.07	3755.30	368.62	849.15
GWP	824.99	1494.67	669.68	511.92	313.07
ODP	9.12	30.26	21.14	4.03	5.09

Distribution 1

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	14.94	28.62	13.68	7.70	7.24
Water use	8.45	62.18	53.73	1.15	7.30
GWP	6.97	14.45	7.48	3.73	3.24
ODP	0.00	0.00	0.00	0.00	0.00

Component manufacturing

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	1180.53	1871.33	690.80	744.69	435.84
Water use	3.82	26.83	23.01	0.54	3.28
GWP	207.86	372.04	164.18	127.33	80.53
ODP	1.90	10.75	8.85	0.41	1.49

Distribution 2

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	135.64	193.18	57.54	97.28	38.36
Water use	49.18	211.17	161.99	13.53	35.65
GWP	15.49	29.93	14.44	9.54	5.95
ODP	0.45	0.91	0.46	0.23	0.22

Filling

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	-0.29	-0.18	0.11	-0.46	0.17
Water use	0.00	0.00	0.00	0.00	0.00
GWP	0.03	0.09	0.06	0.02	0.01
ODP	0.00	0.01	0.01	0.00	0.00

Distribution 3

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	849.78	1283.38	433.60	570.93	278.85
Water use	288.42	1396.53	1108.11	75.04	213.38
GWP	70.17	156.88	86.71	42.47	27.70
ODP	3.49	7.01	3.52	1.73	1.76

End of life

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	-172.97	-68.19	104.78	-393.75	220.78
Water use	1.99	13.90	11.91	0.30	1.69
GWP	25.32	65.05	39.73	13.09	12.23
ODP	0.65	4.71	4.06	0.11	0.54

Total

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	6954.10	10205.44	3251.34	5150.63	1803.47
Water use	1777.48	5924.91	4147.43	663.84	1113.64
GWP	1190.32	1881.50	691.18	840.16	350.16
ODP	17.99	42.69	24.70	10.17	7.82

Table A. 8. Life cycle results for PDS3

Material Production

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	4080.39	6128.49	2048.10	2807.50	1272.89
Water use	2596.71	16160.74	13564.03	489.64	2107.07
GWP	243.25	447.17	203.92	177.99	65.26
ODP	0.52	0.99	0.47	0.28	0.24

Distribution 1

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	29.73	189.50	159.77	7.59	22.14
Water use	20.95	315.87	294.92	1.39	19.56
GWP	16.14	93.43	77.29	3.58	12.56
ODP	0.00	0.02	0.02	0.00	0.00

Component manufacturing

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	524.28	1387.48	863.20	214.19	310.09
Water use	1.75	14.23	12.48	0.22	1.53
GWP	94.70	260.91	166.21	37.83	56.87
ODP	0.87	5.82	4.95	0.17	0.70

Distribution 2

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	58.63	80.66	22.03	44.09	14.54
Water use	25.72	112.11	86.39	7.33	18.39
GWP	13.22	24.98	11.76	7.84	5.38
ODP	0.08	0.16	0.08	0.04	0.04

Filling

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	987.26	1755.48	768.22	313.85	673.41
Water use	2.94	22.70	19.76	0.29	2.65
GWP	172.16	345.68	173.52	53.98	118.18
ODP	1.46	8.66	7.20	0.21	1.25

Distribution 3

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	655.15	975.85	320.70	442.67	212.48
Water use	232.45	1102.40	869.95	60.37	172.08
GWP	59.42	127.88	68.46	36.27	23.15
ODP	2.64	5.19	2.55	1.31	1.33

End of life

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	-515.19	-357.20	157.99	-758.20	243.01
Water use	-1.69	2.19	3.88	-9.54	7.85
GWP	43.49	127.72	84.23	14.80	28.69
ODP	1.14	8.61	7.47	-0.07	1.21

Total

	Median	95th tile	95th tile-Median = Upper end	5th tile	Median-5th tile = Lower end
Energy	5997.49	8172.05	2174.56	4461.40	1536.09
Water use	3132.80	16700.86	13568.06	828.81	2303.99
GWP	704.03	1050.56	346.53	526.84	177.19
ODP	8.19	22.95	14.76	3.97	4.22

IIIDetails of the sensitivity analysis using Spearman's Rank correlation method

Technical note from:

http://www.decisioneering.com/support/simulation/cbl_gen_004A.html

How does Crystal Ball calculate sensitivity?

Sensitivity Analysis is an option in the Run Preferences dialog. When it is on, Crystal Ball calculates how important each assumption is to each forecast while the simulation is running.

Crystal Ball calculates sensitivity this way:

1. During a simulation, Crystal Ball saves all the calculated assumption and forecast values.
2. After a sample size (another Run Preferences option), N , number of trials, Crystal Ball generates a list of ranks. The smallest number is 1, and the largest number is the N.

Table A. 9. Details of sensitivity analysis

Assumption 1 values	Assumption 1 ranks	Assumption 2 values	Assumption 2 ranks	Forecast 1 values	Forecast 1 ranks
12.43	3	-2.44	3	23.01	2
4.82	1	-5.90	2	47.22	3
6.01	2	-12.53	1	7.70	1

3. Crystal Ball pairs up each assumption rank list with each corresponding forecast rank list and then calculates a Pearson correlation coefficient for each pair.

Numerical example

X Y
1 2
2 5
3 6

$\Sigma XY = (1)(2) + (2)(5) + (3)(6) = 30$
$\Sigma X = 1 + 2 + 3 = 6$
$\Sigma X^2 = 1^2 + 2^2 + 3^2 = 14$
$\Sigma Y = 2 + 5 + 6 = 13$
$\Sigma Y^2 = 2^2 + 5^2 + 6^2 = 65$
$N=3$
$\Sigma XY - \Sigma X \Sigma Y/N = 30 - (6)(13)/3 = 4$
$\Sigma X^2 - (\Sigma X)^2/N = 14 - 6^2/3 = 2$
$\Sigma Y^2 - (\Sigma Y)^2/N = 65 - 13^2/3 = 8.667$
$r = 4/\sqrt{(2)(8.6667)} = 4/4.16333$
$= .9608$

4. If the model has more than one forecast, Crystal Ball also pairs up all the corresponding forecast rank lists and then calculates a Pearson correlation coefficient for each pair of forecasts.
5. Crystal Ball stores the running sums of the Pearson correlation coefficients for each pairing in separate matrices.
6. After the simulation, Crystal Ball averages the computed correlation coefficients.

If the sensitivity is measured by Rank Correlation (In the Sensitivity Preferences dialog), these average values appear in the Sensitivity chart.

If the sensitivity is measured by Contribution To Variance (in the Sensitivity Preferences dialog), Crystal Ball normalizes the square of the sensitivity coefficient. Crystal Ball:

1. Squares the sensitivity coefficient.
2. Adds all the squares.
3. Divides the squares by the sum of all the squares to ensure a sum of 1.
4. Converts each normalized value to a percentage.

These normalized percentage values appear in the Sensitivity chart.

For the most accurate analysis, set the Sample Size option to the number of trials for the simulation.

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