Development of life cycle assessment methodology:

a focus on co-product allocation

Allocatie in Milieugerichte Levenscyclusanalyse

Thesis

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Abstract

Life Cycle Assessment (LCA) is a developing tool that can assist decision-makers in evaluating the comparative potential cradle-to-grave, environmental impacts of their actions in order to prevent unintended consequences. The LCA tool is designed to assist in the evaluation along the 'value chain' of a product's life from the perspective that the many processes are interdependent, so that decisions made at one point along the life cycle can have consequences elsewhere. LCA enables the estimation of the cumulative environmental impacts, often including impacts that go beyond the boundaries of traditional analyses. By including the impacts throughout the product life cycle, LCA provides a comprehensive view of the environmental aspects and a more accurate picture of the true environmental trade-offs in product or process alteration or selection.

After a brief look at the general LCA methodology, this thesis presents information about some of the current limitations that prevent LCA from being a more widely-used tool that generates replicable, defensible results. The thesis author then introduces the subject of integrating LCA results into the decision-making process, suggesting that the potentially, holistic view offered by LCA can be valuable as one tool for helping to achieve sustainable societal development. Bio-fuels are presented as the subject of a "wicked problem" for which LCA can be used to support the decision-making process. The thesis then focuses on the co-product allocation issue and presents the results of testing various allocation schemes (weight, volume, market value, energy, and demand-based approach) across a fuel system. The relative ranking of conventional gas to alternative ethanol fuels is consistent in all allocation schemes.

This thesis addresses three topical areas within LCA: the evolution and current status of LCA methodology (Chapter 2), integrating LCA with sustainability decision-making with a focus on bio-based materials (Chapter 3), and co-product allocation methodology (Chapters 4 and 5). The final chapter (Chapter 6) presents overall conclusions about the future of LCA and suggests further, urgently needed, research.

Summary

Introduction

The idea of a comprehensive environmental management scheme that identifies unintended consequences before they occur is very alluring. Over the years, the instances in which one problem was solved but caused another are numerous. Tools are needed that can help us to evaluate the comparative potential cradle-to-grave impacts of our actions in order to help us to prevent such wide-ranging effects. Life Cycle Assessment, or LCA, is one tool that can provide assistance in the decision-making process.

LCA evaluates possible outcomes and consequences of different actions within industrial systems. It considers aspects of resource use and environmental releases associated with a system, as defined by the function provided by a product or good. This holistic approach considers relevant impacts up-stream and down-stream of the producer or consumer. Specifically, LCA helps researchers to evaluate potential environmental interactions that cover a range of activities, from the extraction of raw materials from the Earth and the production and distribution of energy, through the manufacture, use, and reuse, and final disposal of a product. LCA can also be applied to a process, an activity, or a service. By including the impacts throughout the life cycle, LCA provides a comprehensive view of the related environmental aspects, thus, providing a more accurate assessment of potential environmental trade-offs when selecting between products and processes.

After a brief look at general LCA methodology, this thesis author presents the current limitations that prevent LCA from being a widely-used tool that generates replicable, defensible results, including issues related to defining the goal and scope of the study, accessing available data sources, aggregating data across technologies, allocating emissions across input materials and co-products, modelling impacts, assessing data quality, and interpreting the results. This thesis author introduces the subject of integrating LCA results into the decision-making process, suggesting that the potentially, holistic view offered by LCA can be valuable as one tool for helping to achieve sustainable societal development. This thesis author then focuses on the co-product allocation issue and presents the results of testing various allocation schemes across an industrial system. The final chapter presents overall conclusions and recommendations for further, urgently needed, research.

Integrating LCA and Sustainability

The issue of environmental impacts related to bio-based materials is a complicated one. There is no simple answer to the question “are materials from bio-based feedstock environmentally preferable?” Bio-fuels, for example, appear to be effective in reducing some aspects (such as fossil fuel use) while increasing others (such as water quality impacts). It brings into question how we define and measure ‘sustainability.’ Whichever metrics are chosen to measure sustainability, the analysis must be on a life cycle basis.

Bio-fuels are presented as the subject of a "wicked problem" for which LCA can be used to support the decision-making process. Despite the rush to champion bio-fuels as an alternative energy source that *can* increase national security, reduce vehicle emissions, and provide increased revenue for the farming community, some are contesting these claims by taking a broader view, often referred to as "systems thinking," of the potential consequences. A dramatic increase of interest in LCA-based tools has been brought to the forefront by the on-going debates of switching from fossil fuels to bio-fuels.

Allocation Methodology in LCA

When creating the input/output data for a life cycle inventory, the LCA practitioner must decide how to allocate the environmental releases that are emitted by a process across its various co-products. A literature review conducted by this thesis author showed that while many authors have written about the allocation issue, no one had looked at the impact that allocation has on an entire product system. To that end, this thesis author conducted research to evaluate the effect that changing the basis of co-product allocation has on the results when comparing products at a system level. The results of the allocation scheme showed remarkable consistency when comparing the impacts of conventional gasoline to two bio-ethanol alternatives. In each scenario, conventional gasoline resulted in lower impact scores in the global warming, ozone depletion, and human health-noncancer (water) categories, while the bio-ethanol products resulted in a lower impact score for human health cancer. The results of the study showed that, for this particular case study, allocation methodology does not alter the final LCA results when comparing systems.

Conclusion

The paradigm of sustainability, with its three aspects of social responsibility, economic performance, and environmental stewardship, has become accepted as the goal of public policy. However, as in the past, approaches to environmental protection continue to be based on solutions which are focused on single media (e.g., air, water, soil) and within a single stage in the life-cycle stage of a product. These do not always lead to an overall reduction in environmental impacts. Often they create unintended consequences, allowing one environmental problem to be solved while generating another. We are beginning to recognize that the path towards sustainability requires life cycle thinking and the cooperation among the various stakeholders throughout the life cycles of products and services.

While LCA is simple in concept, the details of its practice are complex and still evolving. Researchers are continually coming up with new ways to enhance accuracy and applicability. In order for LCA to continue on the path of increased awareness and broadened applicability to serve as a decision-support tool, several key factors must be met in the near future. Life cycle thinking must be implemented in the earliest stage of the decision-making process and it should be encouraged within all industrial and governmental sectors. As interest in LCA continues to grow, our knowledge and understanding of LCA methodology and application has developed significantly over the past three decades. While further refinement of the methodology is needed, the future offers both significant challenges and opportunities for LCA and LCA-related approaches to play important decision-making support roles.

Samenvatting

De gedachte van een alomvattend schema voor milieumanagement, dat onbedoelde consequenties identificeert nog voordat ze zich voordoen, is zeer aantrekkelijk. Het is al heel lang zo dat voor milieuproblemen, na uitvoerige analyse, een oplossing gevonden wordt die vervolgens nieuwe milieuproblemen veroorzaakt. We hebben instrumenten nodig die ons helpen om zulke effecten te voorkomen, door het vergelijken van de milieu-effecten van onze activiteiten 'van wieg tot graf'. Levenscyclusanalyse, veelal aangeduid als LCA, is zo'n instrument, dat behulpzaam kan zijn in het besluitvormingsproces.

LCA evalueert de mogelijke resultaten en consequenties van verschillende keuzes in industriele systemen. Het beschouwt aspecten van het gebruik van natuurlijke hulpbronnen en de uitstoot van stoffen die uit het systeem voortkomen, zoals bepaald door het eindgebruik van een bepaald product. Deze holistische benadering beschouwt de gevolgen in de keten, bovenstrooms van producent en benedenstrooms van consument. LCA helpt onderzoekers om de mogelijke milieugevolgen van een breed scala aan activiteiten – van de onttrekking van grondstoffen aan de aarde en de productie en distributie van energie, naar de productie, gebruik, hergebruik en het afdanken van een product. LCA kan ook toegepast worden op een proces, activiteit of dienst. Door die gevolgen in de gehele levenscyclus in kaart te brengen verschaft LCA een alomvattend beeld van de milieugevolgen. Zo geeft LCA een nauwkeuriger inzicht in de mogelijke `trade-offs' tussen milieu-effecten bij keuzes tussen producten en processen.

In dit proefschrift bespreekt de auteur allereerst de methode van LCA in het algemeen, om vervolgens de huidige beperkingen ervan zichtbaar te maken. Het zijn deze beperkingen die ertoe leiden dat LCA niet gezien wordt als een veelgebruikt instrument dat repliceerbare, verdedigbare resultaten oplevert. Daartoe behoren onder andere vraagstukken bij het bepalen van doel en reikwijdte van een LCA, toegang tot beschikbare gegevensbronnen, het aggregeren van data die op verschillende technieken betrekking hebben, het alloceren van emissies aan `inputs' en co-producten, het modelleren van milieugevolgen, het toetsen van de kwaliteit van gegevens, en het interpreteren van resultaten. De auteur adresseert de vraag hoe de resultaten van LCA in besluitvormingsprocessen benut kunnen worden, en neemt de positie in dat de holistische benadering van LCS waardevol kan zijn in het realiseren van duurzame maatschappelijke ontwikkeling. De auteur richt zich vervolgens vooral op het allocatievraagstuk, en bespreekt de resultaten van verschillende allocatieregels in een industrieel systeem. Het slothoofdstuk bevat de conclusies en aanbevelingen voor verder onderzoek, dat zeer urgent is.

Integratie van LCA en duurzaamheid

Het vraagstuk van de milieugevolgen van biomassa is ingewikkeld. Er is geen simpel antwoord op de vraag "is biomassa uit milieu-oogpunt te verkiezen boven conventionele grondstoffen?". Biobrandstoffen bijvoorbeeld, lijken beter te zijn bij het terugdringen van sommige milieugevolgen, zoals het gebruik van fossiele brandstoffen, terwijl ze slechter zijn als het gaat om de gevolgen voor de kwaliteit van water. Dat leidt tot de vraag hoe we 'duurzaamheid' definiëren en meten. Welke metriek ook gekozen wordt om duurzaamheid te meten, de analyse moet plaatsvinden over de gehele levenscyclus.

Biobrandstoffen worden besproken als een "taai probleem", waarbij LCA ingezet kan worden om het besluitvormingsproces te ondersteunen. Ondanks de haast waarmee biobrandstoffen worden aangeprezen als een alternatieve energiebron die kan bijdragen tot een verbetering van de voorzieningszekerheid, een verlaging van de uitstoot uit voertuigen en een verhoging van de inkomsten van de agrarische sector, zijn tegenovergestelde beweringen, die voortkomen uit het hanteren van een breder perspectief, dat vaak omschreven wordt als 'systeem denken'. De belangstelling voor instrumenten op basis van LCA is sterk toegenomen door het maatschappelijk debat over de overstap van fossiele brandstoffen naar biobrandstoffen.

Allocatie in LCA

Bij het verzamelen van de input/output gegevens voor de inventarisatiefase in een LCA moet besloten worden hoe de uitstoot van emissies uit een proces toegewezen worden aan de verschillende co-producten die het proces voorbrengt (zoals bij olieraffinage). Uit het literatuuronderzoek voor dit proefschrift blijkt dat veel auteurs geschreven hebben over het vraagstuk van de co-allocatie, maar dat geen van hen de effecten van allocatie op een geheel productsysteem bestudeerd had. Dit proefschrift doet dat wel, door onderzoek naar het effect van het veranderen van de grondslag voor de allocatie, en het nagaan welke gevolgen dit heeft voor het vergelijken van producten op systeemniveau. De resultaten zijn opmerkelijk consistent bij het vergelijken van conventionele benzine met twee bio-ethanol producten. In beide gevallen heeft conventionele benzine een beter resultaat voor de opwarming van de aarde, de uitputting van de ozonlaag and waterkwaliteit (effecten op de volksgezondheid, met uitzondering van carcinogeniteit), terwijl de bio-ethanol producten beter scoren op carcinogeniteit. Deze studie laat zien, dat voor de beschreven productalternatieven, het veranderen van de grondslag voor allocatie geen effect heeft op de LCA-resultaten.

Conclusie

Het duurzaamheidparadigma, met zijn drie aspecten van sociale verantwoordelijkheid, economische prestatie en milieuverantwoordelijkheid, is ondertussen ingeburgerd als doel van overheidsbeleid. Milieubescherming blijft echter gebaseerd op oplossingen die naar afzonderlijke media kijken (zoals lucht, water, bodem) en naar enkelvoudige stappen in de levenscyclus van een product. Daardoor wordt niet altijd een netto vermindering van de milieugevolgen bereikt. Er ontstaan vaak onbedoelde gevolgen, waardoor één milieuprobleem wordt opgelost door een ander te veroorzaken. We beginnen er achter te komen dat de weg naar duurzaamheid vraagt om het denken in levenscycli, en samenwerking tussen actoren in productketens en diensten (integraal ketenbeheer).

LCA is een eenvoudig concept, maar de praktijk is complex, en blijft zich ontwikkelen. Onderzoekers komen steeds weer met nieuwe manieren om de accuratesse en toepasbaarheid te vergroten. Om de bewustwording en toepasbaarheid te vergroten moet LCA zich verder ontwikkelen, en een steeds meer een instrument worden voor het ondersteunen van de besluitvorming. Dat vraagt om een aantal zaken. Allereerst moet LCA al vanaf de eerste fase van het besluitvormingsproces ingezet worden, en de toepassing ervan moet aangemoedigd worden in alle sectoren van overheid en bedrijfsleven. Hoewel de belangstelling voor, de toepassing van, en het repertoire van de LCA methodologie de afgelopen dertig jaar fors is toegenomen blijft het zaak te werken aan verbetering en verfijning. Niettemin liggen er forse uitdagingen in de ondersteuning van de besluitvorming die de LCA-aanpak niet uit de weg mag gaan.

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Acronyms and Abbreviations

BCA	Benefit Cost Analysis
BEES	Buildings for Environmental and Economic Sustainability
CFC	Chlorofluorocarbon
CML	Center for Environmental Research, Leiden University
DALY	Disability Adjusted Life Year
DDT	Dichloro-Diphenyl-Trichloroethane
DQI	Data Quality Indicator
DQO	Data Quality Objective
EC	European Commission
EDF	Environmental Defense Fund (now Environmental Defense, ED)
EIA	Energy Information Administration
EPA	Environmental Protection Agency (US)
EPP	Environmentally Preferable Purchasing
EtOH	Ethyl Hydroxide (Ethanol)
FRED	Framework for Responsible Environmental Decision-making
GREET	Greenhouse gases, Regulated Emissions, and Energy use in Transportation
GWP	Global Warming Potential
IDCE	International Design Center for the Environment
IPP	Integrated Product Policy
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organization
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory

LCIA	Life Cycle Impact Assessment
LCM	Life Cycle Management
LCSEA	Life Cycle Stressor Effects Assessment
MTBE	Methyl Tertiary-Butyl Ether
NGO	Non-Governmental Organisation
NIST	National Institute for Standards and Technology
NPI	National Pollutant Inventory (Australia)
NREL	National Renewable Energy Laboratory (DoE)
NRMRL	National Risk Management Research Laboratory (US EPA)
ODP	Ozone Depletion Potential
OECD	Organisation for Economic Cooperation and Development
ORD	Office of Research and Development (US EPA)
REPA	Resource and Environmental Profile Analysis
RFG	Reformulated Gasoline
SETAC	Society of Environmental Toxicology and Chemistry
TRACI	Tool for the Reduction and Assessment of Chemical and other environmental Impacts
TRI	Toxics Release Inventory (USA)
TEP	Toxic Equivalency Potential
UNEP	United Nations Environment Program
USDA	US Department of Agriculture
USGBC	US Green Building Council
WMO	World Meterological Organization

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1 Introduction

"They see the immediate situation. They think narrowly and they call it 'being focused.' They don't see the surround. They don't see the consequences."

Michael Crichton, *Jurrasic Park*

1.1 Thesis Scope

The idea of a comprehensive environmental management scheme that identifies unintended consequences before they occur is very alluring. Over the years, the instances in which one problem was solved but caused another are numerous. Consider, for example, the unanticipated consequences of the extensive usage of the pesticide DDT which was extremely effective in combating the spread of malaria by mosquitoes, yet was later suspected of having a severe effect on wildlife, particulary birds and the thinning of egg shells. Or, the unanticipated ozone layer-depleting effects of halons (such as Halon 1301, or bromotrifluoromethane) which are synthetic chemicals used in fire extinguishers, because they are lighter and more effective than water.

When first introduced in the 1930s, CFCs and their related bromine halon compounds seemed to be ideal man-made chemicals. They are stable, nontoxic, nonflammable, non-corrosive, and relatively inexpensive to produce, making them uniquely suited for a myriad of consumer and industrial applications. Over the years, they found more and more uses in thousands of products and processes, including pharmaceuticals, cosmetics, spray cans, agriculture, petroleum, microchips, electronics, automotive, defense, aircraft, insulation, plastic foam, aerospace, telecommunications, refrigeration, and air conditioning. CFCs became virtually synonymous with modern standards of living. It was later found in the 1970s that CFCs had highly destructive effects on high-altitude ozone. Furthermore, molecule for molecule, halon compounds were found to be three to ten times as destructive to the ozone layer as chlorofluorocarbon 11 and 12.

Tools are needed that can help us to evaluate the comparative potential cradle-to-grave impacts of our actions in order to help us to prevent such wide-ranging effects. Life Cycle Assessment, or LCA, is one tool that can provide assistance in the decision-making process. The inventor of DDT, Paul Hermann Müller, received the Nobel Prize in 1948 for his discovery. This gives one pause to consider the irony of this. Would the selection committee have made the same choice if life cycle information had been made available to them at the time?

LCA, however, has limited applicability in that it can only help us to evaluate the data that are available at the time. That is, it is not a predictive tool. So conducting an LCA may not have altered the choice to use halons in fire extinguishers since ozone depletion was not recognized until many years after we had used large quantities of them. However, it has become increasingly evident that we must look much more holistically at our actions in order to more effectively protect human health and the environment in the short and long-term and to therefore, contribute to the development of more sustainable societies.

The origins of the concept behind the evolving LCA methodologies can be traced back to 1969 (a more detailed accounting of the history of LCA can be found in Section 2.2.3). The mid-1990's saw an explosion in LCA activity (Hunt and Franklin 1996). This increased activity can be attributed to the convergence of three key factors: 1. There was a "reawakening" of environmental issues in the mid 1980's (this was initially related to improper management of solid and hazardous wastes); 2. The growing availability of computers simplified the complex task of collecting and manipulating large amounts of data; and 3. Support from the Society of Environmental Toxicology and Chemistry (SETAC) provided the LCA community with a regular venue to meet and advance the LCA field. All these factors led to an expanded interest in cradle-to-grave approaches for products, processes, and services which evolved into the constellation of tools, called LCA.

One of the early workshops on LCA, hosted by SETAC in 1990, was held in Smugglers Notch, Vermont (Fava, Denison et al. 1990). Even at that early time, well-known LCA practitioners, such as Bill Franklin, Gustav Sundström, and Ian Boustead, discussed the complexities involved in life cycle inventory data modeling, as well as other aspects of LCA practice. These early discussions were, of necessity, fairly academic and were focused mainly on establishing terminology and setting boundaries for what should be included in an LCA. Even the term "LCA" was debated among the workshop participants who concluded that the word "assessment" was the more appropriate word since an LCA is not an entirely quantitative process, as the word "analysis" implies.

That workshop was followed by a series of meetings that were held in the United States and Europe. Every subsequent workshop contributed to a better understanding and to the development of more detail in LCA methodology. After this initial flurry of activity, development of the LCA methodology stalled as work on the international environmental management standards evolved under the auspices of the International Standards Organization (ISO). The standards mirrored the four phases of an LCA: clearly defining the goal and scope of the study; compiling an inventory of relevant energy and material inputs and environmental releases; evaluating the potential environmental impacts associated with identified inputs and releases; and interpreting the results to help decision-makers make a more informed decision. The initial standards series included General Principles (14040), Life Cycle Inventory (14041), Life Cycle Impact Assessment (14042) and Life Cycle Interpretation (14043). These were later condensed in 2006 into two documents, entitled 'Principles and Framework' (14040) and 'Requirements and Guidelines' (14044).

Completed in the year 2000 after seven years of effort, the original 14000 series of standards was instrumental in further defining and refining LCA tools and practice. However, the standards were written to be "flexible" in that they were descriptive and not prescriptive. That is, while the standards define LCA and identify the key concepts, tools, and approaches, they do not give step-wise instructions on how to conduct an LCA. The advantage to this flexibility is that it encourages flexibility in LCA practice by enabling LCA practitioners and researchers to conduct assessments using 'best engineering/scientific' judgment. However, such descriptive standards leave detailed methodological issues unresolved, which sometimes leads to questions, covering both the practice of inventory data collection and impact assessment modeling. In life cycle impact assessment practice, the global level impacts, such as ozone depletion and global climate change, are closer to consensus while the models of regional and local impacts, such as human and ecological toxicity, vary more widely. Land use and water use remain particularly problematic with consistent models yet to be developed.

More detailed guidance is needed at key decision points in the creation of life cycle inventory, including areas such as: co-product allocation; recycling allocation; rules of exclusion; age-appropriateness of data; surrogate and estimated data; inventory for impact assessment; matching the goal to the method; collecting primary data; report format; iterative procedure for data collection; choosing boundaries; capital equipment/infrastructure exclusions; and time and location meta data. This thesis author delves into the current state of LCA practice and explores how these decision-making points lead to methodological difficulties. This author worked to make valuable contributions to resolving some of these issues.

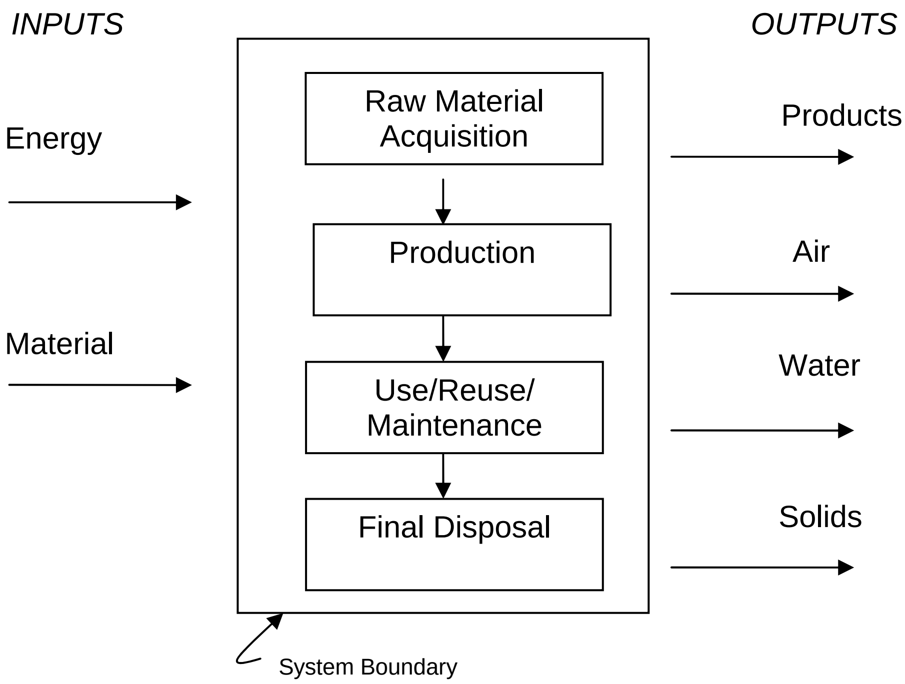
While these various issues have not been formally prioritized in order of importance, the allocation of life cycle inventory data across multiple input materials and co-products from industrial processes clearly stands out as a central question in LCI modeling and is one of the most controversial issues in the development of LCA methodology (Weidema 2001). This thesis author chose to research co-product allocation methodology because it clearly continues to be the most important and least understood aspect of inventory data modeling. The objective of this research was to explore the impact that the choice of allocation methodology has on comparing alternative life cycle systems.

After a brief look at general LCA methodology, this thesis author presents the current limitations that currently prevent LCA from being a widely-used tool that generates replicable, defensible results. This thesis author then introduces the subject of integrating LCA results into the decision-making process, suggesting that the potentially, holistic view offered by LCA can be valuable as one tool for helping to achieve sustainable societal development. Bio-fuels are presented as the subject of a "wicked problem" for which LCA can be used to support the decision-making process. This thesis author then focuses on the co-product allocation issue and presents the results of testing various allocation schemes across an industrial system. The final chapter presents overall conclusions and recommendations for future research.

1.2 Defining Life Cycle Assessment

LCA is a tool for evaluating possible outcomes and consequences of different actions. The specific application under review is a "cradle-to-grave" approach for assessing industrial systems. It considers aspects of resource use and environmental releases associated with a system, as defined by the function provided by a product or good. This holistic approach considers relevant impacts up-stream and down-stream of the producer or consumer. Specifically, LCA helps researchers to evaluate potential environmental interactions that cover a range of activities, from the extraction of raw materials from the Earth and the production and distribution of energy, through the manufacture, use, and reuse, and final disposal of a product (see Figure 1-1). LCA can also be applied to a process, an activity, or a service. By including the impacts throughout the life cycle, LCA provides a comprehensive view of the related environmental aspects, thus, providing a more accurate assessment of potential environmental trade-offs when selecting between products and processes.

Figure 1-1 A Life Cycle Assessment includes attention upon four inter-related stages of a product from raw material acquisition through production, use, and final disposal



LCA evaluates all stages of a product's life from the perspective that they are interdependent, meaning that decisions made at one point along the life cycle can have consequences somewhere else. LCA enables the estimation of the cumulative environmental impacts resulting from all stages in the product life cycle, often including impacts that go beyond the boundaries of traditional analyses. By including the impacts throughout the product's life cycle, LCA provides a comprehensive view of the environmental aspects and a more accurate picture of the true environmental issues in product or process selection. A life cycle approach helps us recognize how our choices influence each point of the life cycle so that we can balance potential trade-offs and avoid shifting problems from one medium to another (e.g., controlling air emissions which creates wastewater effluents or soil contamination) and/or from one life cycle stage to another (e.g., the raw material acquisition stage which impacts upon the consequences of reuse of materials for subsequent life cycles of products), thereby potentially helping managers to reduce the negative environmental and human health impacts. LCA is a way of thinking about the choices we make, when purchasing products, selecting materials, or identifying process alternatives by putting our decisions into the context of the entire life cycle system.

This ability to track and document shifts in environmental impacts can help decision-makers and managers more fully characterize the environmental trade-offs associated with product or process alternatives. By performing LCAs analysts can, for example:

- Develop a systematic evaluation of the environmental and human health consequences associated with a given product.

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

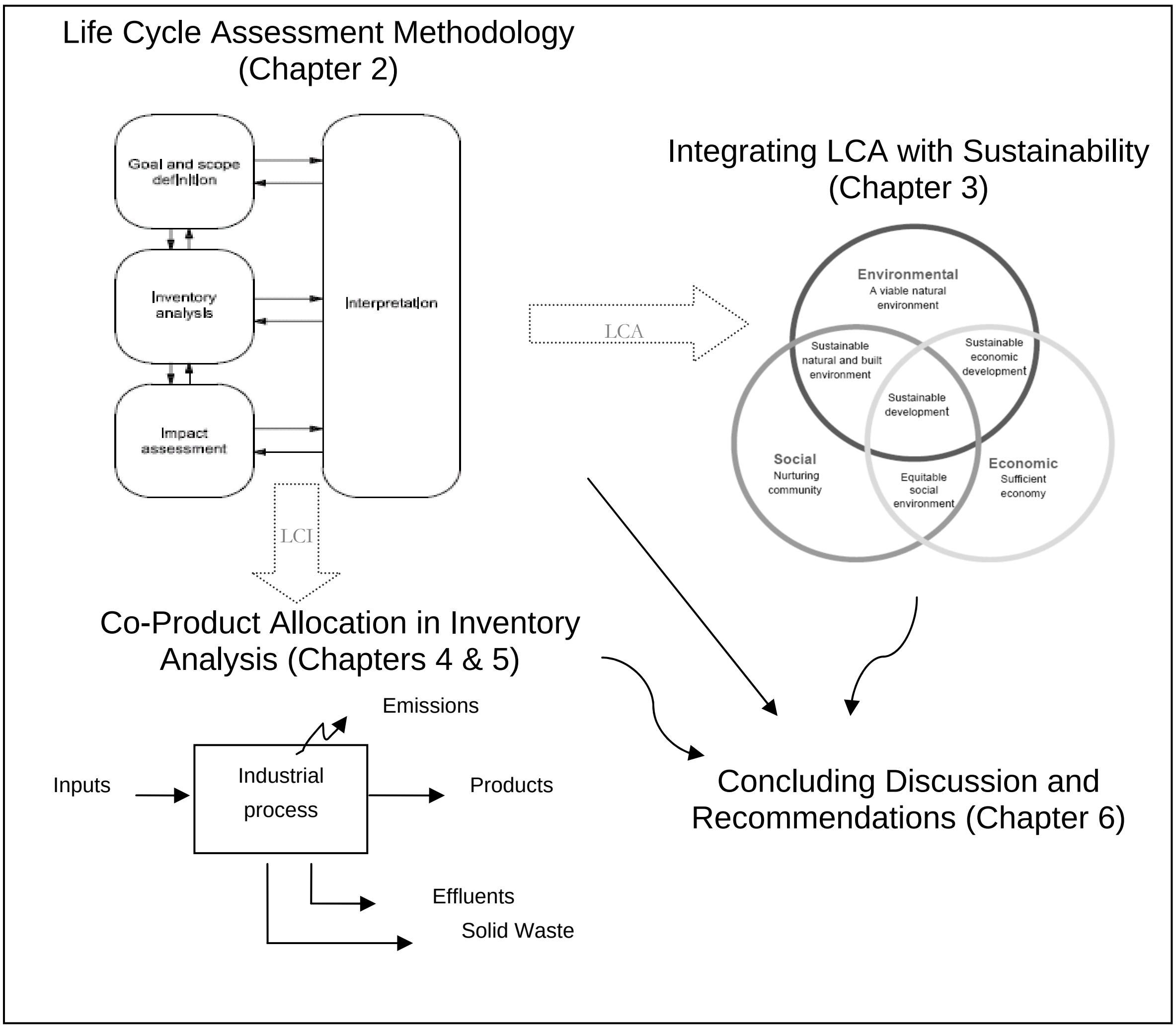
(U.S. Environmental Protection Agency 2006)

Broadening a study's boundaries helps decision-makers select the products and processes that result in the least impact to the environment. The fact that LCAs deal only with environmental effects does not, of course, imply that other aspects, such as economic or social aspects, are not as relevant or as important. Used with other tools, information and data generated by an LCA can be used to support decision-making for achieving sustainability goals. This integration of concepts and techniques to address environmental, economic, technological, and social aspects of products, services, and organizations on a life cycle basis is referred to as Life Cycle Management (LCM). LCM, as any other management pattern, is applied on a voluntary basis and can be adapted to the specific needs and characteristics of individual organizations (Hunkeler, Rebitzer et al. 2004).

1.3 Thesis Overview

This thesis addresses three topical areas within LCA. It begins with a general description of the state-of-the-practice of LCA, then moves to the use of LCA information in decision-making, and continues with a focus on the methodological differences in how life cycle inventory data are created. The flow of these topics within this thesis is shown in Figure 1-2. The evolution and current status of LCA methodology are covered in Chapter 2; integrating LCA into sustainability decision-making with a focus on bio-based materials is discussed in Chapter 3; and co-product allocation methodology is further explored in Chapters 4 and 5. In Chapter 6, this thesis author discusses the implications of the findings presented in the previous chapters. This final chapter presents the overall conclusions along with recommendations and suggestions for further, urgently needed, research.

Figure 1-2 Roadmap showing the three topics within Life Cycle Assessment covered in this thesis and their connections among the chapters



The research reported within this thesis spans a period of four years (October 2003 - July 2007). During this time this thesis author published seven peer-reviewed papers which have been incorporated into the thesis as follows:

Chapter 2 – LCA Methodology

I. Curran, MA (in press) "Human Ecology: Life Cycle Assessment," to be published in the new *Encyclopaedia of Ecology*, Elsevier, expected publication July 2008.

II. Curran, MA, Mann, M and Norris G (2005) "The International Workshop on Electricity Data for Life Cycle Inventories," *Journal of Cleaner Production* 13(8), pp853-862.
(J Cle Pro 2006 Impact Factor 0.762)

III. Curran, MA (2004) "The Status of Life Cycle Assessment as an Environmental Management Tool," *Environmental Progress* 23(4), pp277-283.
(Env Prog 2006 Impact Factor 0.653)

Chapter 3 –Integrating LCA and Sustainability

IV. Curran, MA (2005) "Do Bio-Based Products Move Us Toward Sustainability? A Look at Three USEPA Case Studies," *Environmental Progress* 22(4), pp277-295.

V. Von Blottnitz, H and Curran, M (2006) "A Review of Assessments Conducted on Bio-Ethanol as a Transportation Fuel from a Net Energy, Greenhouse Gas, and Environmental Life Cycle Perspective," *Journal of Cleaner Production*, Vol 15(7), pp607-619.

Chapter 4 – Getting to Allocation

VI. Curran, MA (2006) "Co-Product and Input Allocation Approaches for Creating Life Cycle Inventory Data: A Literature Review," *International Journal of Life Cycle Assessment* 12(1) pp65-78. (I JLCA 2006 Impact Factor 1.42)

Chapter 5 – Co-Product Allocation

VII. Curran, MA (2007) "Studying the Effect on System Preference by Varying Co-Product Allocation in Creating Life Cycle Inventory," *Environmental Science & Technology* Vol 41, No 20, pp7145-7151.
(Env Sci Tech 2006 Impact Factor 4.04)

Chapter 6 contains an overall discussion of the results and their implications. It also includes recommendations for urgently needed follow-up research. The intended audience for this thesis includes LCA researchers, practitioners, energy authorities, policy makers and stakeholders within related industries.

Figure 1-3 Environmental choices should be placed in context with the entire life cycle system

BLANK

2 Life Cycle Assessment Methodology

2.1 Chapter Overview

This chapter presents a general discussion on LCA practice and methodology, that is, the compilation and evaluation of the inputs and outputs and potential impacts of a product system through its life cycle. Other publications are available that provide detailed instruction on how to conduct an LCA, see, for example, (Guinee 2001), (Baumann and Tillman 2004), (U.S. Environmental Protection Agency 2006). In order for readers to understand the topic at hand, this section provides a general overview of LCA. To do this, an entry for the new Encyclopaedia of Ecology, entitled "Human Ecology: Life Cycle Assessment," is included in Section 2.2 in its entirety (Curran in press). The encyclopaedia is expected to be published by Elsevier in July 2008 in a 5-volume set.

To further assist the reader in better understanding LCA methodology, a brief case study is included in Section 2.3. The case involves replacing methyl tertiary-butyl ether (MTBE) in gasoline with ethanol made from corn for use in a typical American passenger car. The study is described along the lines of the four phases of an LCA (goal and scope definition, inventory analysis, impact assessment, and interpretation) and amplifies upon how each stage was approached.

Moving from the overview of LCA, the thesis author then focuses upon issues that are involved in modeling and collecting inventory data and in conducting impact assessment. Section 2.4 includes the results of an international workshop held in October 2001 in Cincinnati, Ohio, to discuss the various issues related to life cycle inventory of electricity production ("The International Workshop on Electricity Data for Life Cycle Inventories." *Journal of Cleaner Production* 13(8), 2005, pp853-862). Section 2.5 includes a summary article which covers two key areas of LCA methodology: availability of inventory data and impact assessment modeling ("The Status of Life-Cycle Assessment as an Environmental Management Tool," *Environmental Progress*, 23(4), 2004, pp277-283). These two papers are included to present the 'state-of-the-art' practice as well as to introduce the topic of limitations in life cycle assessment practice, which is further elaborated upon in Section 2.6.

2.2 Paper I - Human Ecology: Life Cycle Assessment

Curran, MA (in press) "Human Ecology: Life Cycle Assessment." to be published in the new *Encyclopaedia of Ecology*, Elsevier, expected publication July 2008.

2.2.1 Abstract

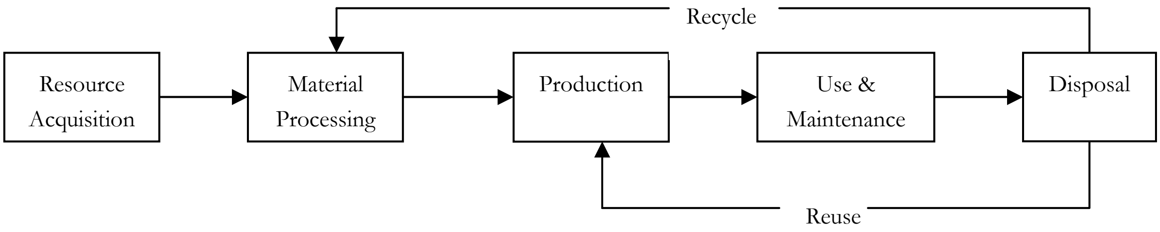
Life Cycle Assessment, or LCA, is an environmental accounting and management approach that considers all the aspects of resource use and environmental releases associated with an industrial system from cradle-to-grave. Specifically, it is a holistic view of environmental interactions that cover a range of activities, from the extraction of raw materials from the Earth and the production and distribution of energy, through the use, and reuse, and final disposal of a product. LCA is a relative tool intended for comparison and not absolute evaluation, thereby helping decision-makers compare all major environmental impacts when choosing between alternative courses of action. This article presents a brief history of the development of LCA methodology and describes the basic components of conducting an LCA, i.e. selecting a functional unit; defining the goal and scope of the study; compiling an inventory of relevant energy and material inputs and environmental releases; evaluating the potential environmental impacts; and interpreting the results to help decision-makers make a more informed decision. Key issues associated with data collection, impact assessment modeling, and interpretation of the results is also outlined. The article concludes with the caution that while LCA is an environmental management tool that informs decision-makers, other decision criteria, such as cost and performance, should also be considered in order to make a well-balanced decision.

2.2.2 Introduction

Life Cycle Assessment, or LCA, is an environmental accounting and management approach for assessing industrial systems. It considers all the aspects of resource use and environmental releases associated with a system, as defined by the function provided by a product, process, or activity. This cradle-to-grave approach considers all relevant impacts up-stream and down-stream of the consumer or producer. Specifically, LCA is a holistic view of environmental interactions that cover a range of activities, from the extraction of raw materials from the Earth and the production and distribution of energy, through the use, and reuse, and final disposal of a product[1]. LCA is a relative tool intended for comparison and not absolute evaluation, thereby helping decision-makers compare all major environmental impacts when choosing between alternative courses of action.

[1] For simplicity, the word 'product' is used although the life cycle concept applies equally well to processes and activities.

Figure 2-1 Generic life cycle stages (arrows represent transportation) from cradle to grave



LCA evaluates all stages of a product's life from the perspective that they are interdependent, meaning that decisions made at one point along the life cycle can have consequences somewhere else. LCA enables the estimation of the cumulative environmental impacts resulting from all stages in the product life cycle, often including impacts that go beyond the boundaries of traditional analyses. By including the impacts throughout the product life cycle, LCA provides a comprehensive view of the environmental aspects and a more accurate picture of the true environmental trade-offs in product or process selection. A life cycle approach helps us recognize how our choices influence each point of the life cycle so that we can balance potential trade-offs and avoid shifting problems from one area to another, thereby positively impacting the overall environment. LCA is a way of thinking about the choices we make, when purchasing products, selecting materials, or identifying process alternatives by putting our decisions in context with facts related to all parts of the life cycle system.

In connecting the different parts of the system, many LCAs lead to unexpected and non-intuitive results. For example, bio-based products, such as paper bags, paper cups, and cloth diapers, are not obviously superior in terms of using less energy and materials. Paper requires the harvesting and transportation of trees to pulp mills, activities which require energy. Paper making releases air pollutants and water discharges of chlorine and biological waste. After use, bags end up in landfills where they decay and release methane in the process. The amount of hot water needed to wash and dry cloth diapers is not inconsequential, especially for those who live where water is scarce or sewage is not treated. These kinds of analyses highlight how the environmental impacts of alternative products may lead to unanticipated consequences.

LCA identifies the potential transfer of environmental impacts from one medium to another (e.g., eliminating air emissions by creating a wastewater effluent instead) and/or from one life cycle stage to another (e.g., from use and reuse of the product to the raw material acquisition stage). If an LCA were not performed, the transfer might not be recognized and properly included in the analysis because it is outside of the typical scope or focus of product selection processes. By broadening the study boundaries, LCA can help decision-makers select the product or process that causes the least impact to the environment. This information can be used with other factors, such as cost and performance data, in the selection process.

2.2.3 The History of Life-Cycle Assessment

LCA had its beginnings in the 1960's. Concerns over the limitations of raw materials and energy resources sparked interest in finding ways to cumulatively account for energy use and to project future resource supplies and use. In 1969, researchers initiated an internal study for The Coca-Cola Company that laid the foundation for the current methods of life cycle inventory analysis in the United States. In a comparison of different beverage containers to determine which container had the lowest releases to the environment and least affected the supply of natural resources, this study quantified the raw materials and fuels used and the environmental loadings from the manufacturing processes for each container. Other companies in both the United States and Europe performed similar comparative life cycle inventory analyses in the early 1970's.

The process of quantifying the resource use and environmental releases of products became known in the United States as a Resource and Environmental Profile Analysis (REPA) while in Europe it was called an Ecobalance. With the formation of public interest groups encouraging industry to ensure the accuracy of information in the public domain, and spurred on by the oil shortages in the early 1970's, a protocol or standard research methodology for conducting these studies was developed and further evolved.

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

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

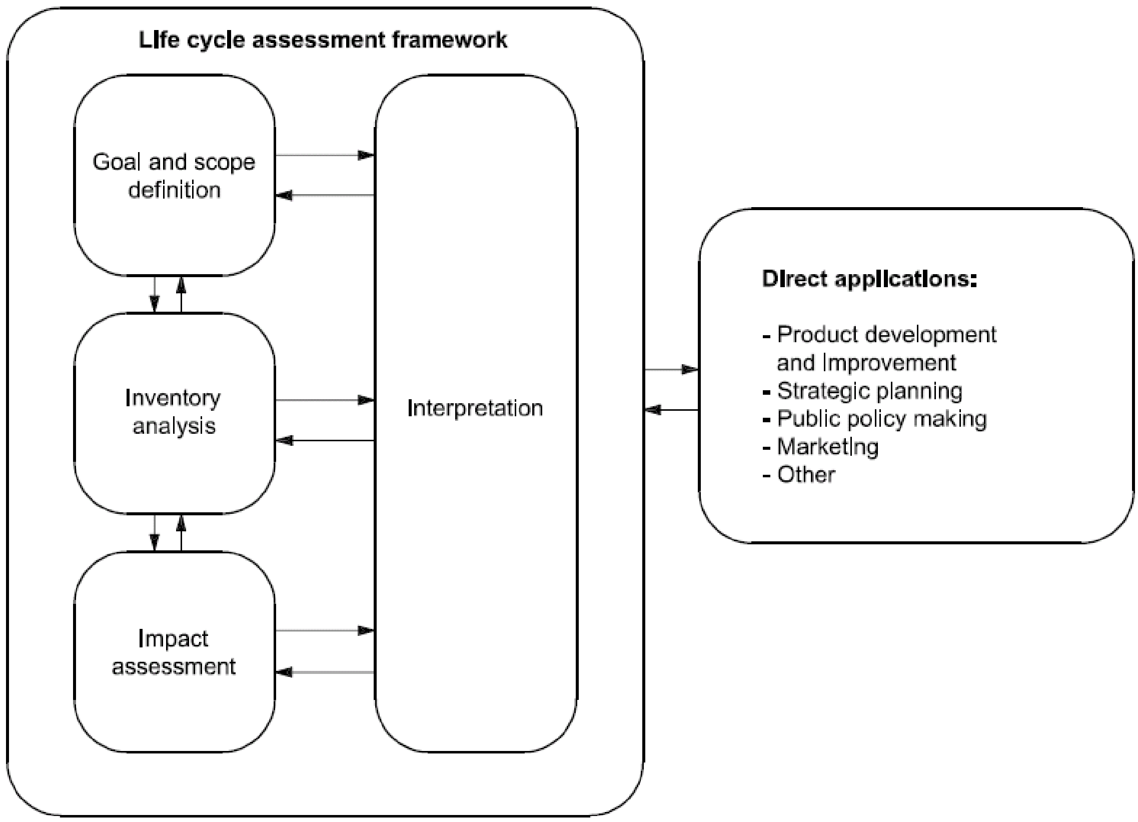
Beginning in 1991, concerns over the inappropriate use of LCAs in making broad marketing claims by product manufacturers, along with pressure from other environmental organizations to standardize LCA methodology, led to the development of the LCA standards in the International Standards Organization (ISO) 14000 series (1997 through 2002). In 2002, the United Nations Environment Programme (UNEP) joined forces with the Society of Environmental Toxicology and Chemistry (SETAC) to launch the Life Cycle Initiative, an international partnership. The three programs of the Initiative aim to put life cycle thinking into practice and to improve the supporting tools through better data and indicators. The Life Cycle Management (LCM) program creates awareness and improves skills of decision-makers by producing information materials, establishing forums for sharing best practice, and carrying out training program in all parts of the world. The Life Cycle Inventory program improves global access to transparent, high quality life cycle data by hosting and facilitating expert groups whose work results in web-based information systems. The Life Cycle Impact Assessment program increases the quality and global reach of life cycle indicators by promoting the exchange of views among experts whose work results in a set of widely accepted recommendations.

2.2.4 Conducting an LCA

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

- Appropriately selecting a functional unit.
- Clearly defining the goal and scope of the study.
- Compiling an inventory of relevant energy and material inputs and environmental releases.
- Evaluating the potential environmental impacts associated with identified inputs and releases.
- Interpreting the results to help decision-makers make a more informed decision.

Figure 2-2 Phases of a Life Cycle Assessment



(ISO 2006)

This ability to track and document shifts in environmental impacts can help decision-makers and managers fully characterize the environmental trade-offs associated with product or process alternatives. By performing an LCA, analysts can, for example:

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

2.2.5 Comparing Apples with Apples

When an LCA is used to compare two or more products, the basis of comparison should be equivalent use, i.e., each system should be defined so that an equal amount of product or equivalent service is delivered to the consumer. For example, if bar soap is compared with liquid soap, the logical basis for comparison would be an equal number of hand washings. Another example of equivalent use would be in comparing cloth diapers to disposable diapers. Cloth diapers need to be changed more frequently, or they are doubled, whereas disposables are not. Thus, more cloth diapers will be used. In this case, a logical basis for comparison between the systems would be the number of diapers used over a set period of time.

Equivalent use for comparative studies can often be based on volume or weight, particularly when the study compares packaging for delivery of a specific product. A beverage container study might consider 1,000 liters of beverage as an equivalent use basis for comparison, because the product may be delivered to the consumer in a variety of container sizes having different life-cycle characteristics.

2.2.6 Life Cycle Inventory

A life cycle inventory is a process of quantifying energy and raw material requirements, atmospheric emissions, waterborne emissions, solid wastes, and other releases for the entire life cycle of a product, process, or activity. An inventory analysis produces a list containing the quantities of pollutants released to the environment (after treatment or control) and the amount of energy and material consumed. The results can be segregated by life cycle stage, media (air, water, and land), specific processes, or any combination thereof.

In the life cycle inventory phase of an LCA, all relevant data are collected and organized. Without a life cycle inventory, no basis exists to evaluate comparative environmental impacts or potential improvements. The level of accuracy and detail of the data collected is reflected throughout the remainder of the LCA process.[2]

Resource constraints for data collection may be a consideration in defining the system, although, in no case should the scientific basis of the study be compromised. The level of detail required to perform a thorough inventory depends on the size of the system and the purpose of the study. In a large system encompassing several industries, certain details may not be significant contributors given the defined intent of the study. These details may be omitted without affecting the accuracy or application of the results. However, if the study has a very specific focus, such as a manufacturer comparing alternative processes or materials for inks used in packaging, it would be important to include chemicals used in very small amounts.

Life cycle inventory analyses can be used in various ways. The data can assist an organization in comparing products or processes and considering environmental factors in material selection. In addition, inventory analyses can be used in policy-making, by helping the government develop regulations regarding resource use and environmental emissions.

Although much can be learned about a process by considering the life cycle inventory data, an impact assessment provides a more meaningful basis to make comparisons. For example, although we know that 9,000 tons of carbon dioxide (CO_2) and 5,000 tons of methane released into the atmosphere are both potentially harmful greenhouse gases, a life cycle impact assessment can determine which could have a greater impact. Which is worse? What are their potential impacts on global warming? On smog formation? Using science-based characterization factors, a life cycle impact assessment can calculate the impacts that each environmental release may have on problems such as smog or global warming.

[2] No pre-defined list of data quality goals exists for all LCA projects. The number and nature of data quality goals necessary depends on the level of accuracy required to inform the decision-makers involved in the process.

2.2.7 Life Cycle Impact Assessment

The life cycle impact assessment phase of an LCA is the evaluation of potential human health and environmental impacts of the environmental resources and releases identified during the inventory. Impact assessment should address ecological and human health effects; it can also address resource depletion. A life cycle impact assessment can be used to help us better understand the linkage between the product or process and its potential environmental impacts.

The following example demonstrates how characterization factors can be calculated to estimate the potential contribution towards an impact category. In this example, global warming potential (GWP) is presented in terms of equivalent emissions of carbon dioxide (CO_2) using units of teragrams of carbon dioxide equivalents (Tg CO_2 eq):

Quantity of CO_2 released = 9,000 tons = .009 Tg (CO₂ GWP Factor Value* = 1)

Quantity of Methane released = 5,000 tons = .005 Tg (Methane GWP Factor Value* = 21)

Therefore;

CO_2 GWP = 0.009 Tg x 1 = 0.009 Tg CO_2 eq.

Methane GWP = 0.005 Tg x 23 = 0.115 Tg CO_2 eq.

*Values are from the Intergovernmental Panel on Climate Change (IPCC) Model, 100 year time horizon, Third Assessment Report, 2001.

The key to impact characterization is using the appropriate characterization factor. For some impact categories, such as global warming and ozone depletion, there is a consensus on acceptable characterization factors. For other impact categories, such as resource depletion, consensus is still being developed.

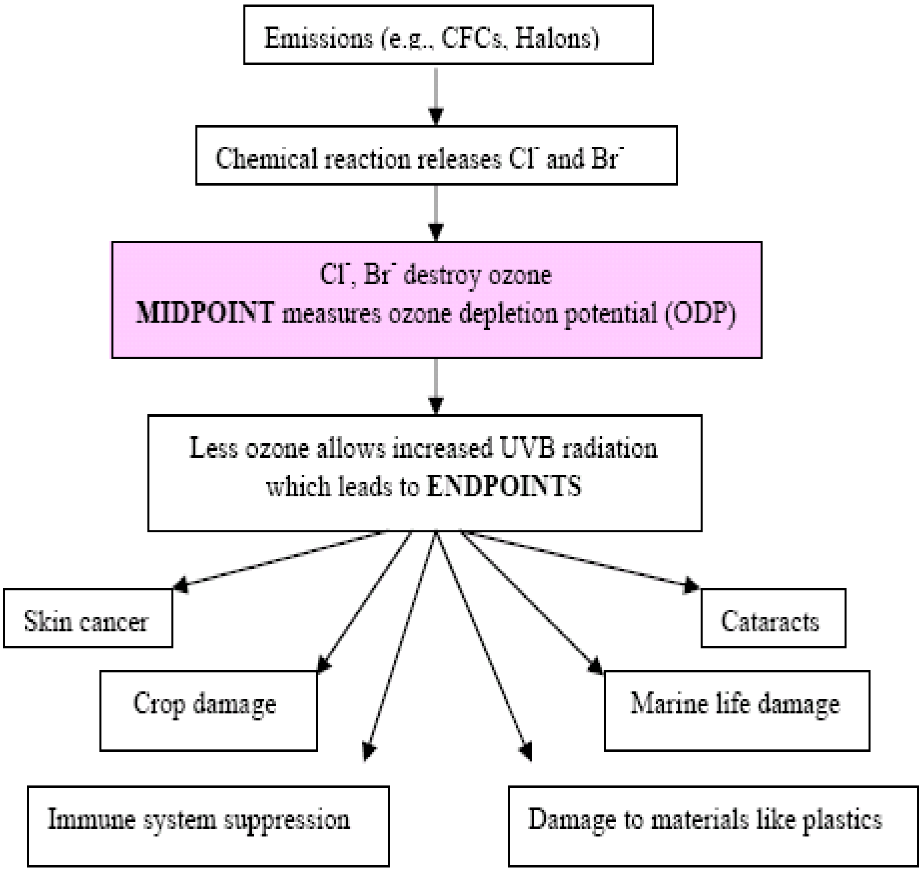
An important distinction exists between life cycle impact assessment and other types of impact analysis. The life cycle impact assessment is not necessarily designed to quantify any specific actual impacts associated with a product, process, or activity. Instead, it is designed to understand the linkage between a system and potential impacts. The models used within impact assessment are often derived and simplified versions of more sophisticated models within each of the impact categories. These simplified models are suitable for relative comparisons of the potential to cause human or environmental damage, but are not indicators of absolute risk or actual damage to human health or the environment. For example, risk assessments are often very narrowly focused on a single chemical at a very specific location. In the case of a traditional risk assessment, it is possible to conduct very detailed modeling of the predicted impacts of the chemical on the population exposed and even to predict the probability of the population being impacted by the emission. In the case of LCA, hundreds of chemical emissions (and resource stressors) which are occurring at various locations are evaluated for their potential impacts in multiple impact categories. The sheer number of stressors being evaluated, the variety of locations, and the diversity of impact categories makes it impossible to conduct the assessment at the same level of rigor as a traditional risk assessment. Instead, models are based on the accepted models within each of the impact categories using assumptions and default values as necessary. The resulting impact models are suitable for relative comparisons, though insufficient, for absolute predictions of risk.

The key concept in this component is that of stressors. A stressor is a set of conditions that may lead to an impact. For example, if a product or process is emitting greenhouse gases, the increase of greenhouse gases in the atmosphere *may* contribute to global warming. Processes that result in the discharge of excess nutrients into bodies of water *may* lead to eutrophication. A life cycle impact assessment provides a systematic procedure for classifying and characterizing these types of environmental effects.

2.2.7.1 Midpoint versus Endpoint Impact Modeling

In order to simplify the process and make LCIA more broadly applicable, modeling is typically conducted at the mid-point level instead of modeling effects to specific end-points (see the Ozone Depletion Potential example that follows). Midpoint impact assessment models reflect the relative potency of the stressors at a common midpoint within the cause-effect chain. Analysis at a midpoint minimizes the amount of forecasting and effect modeling incorporated into the LCIA, thereby reducing the complexity of the modeling and often simplifying communication. Midpoint modeling can minimize assumptions and value choices, reflect a higher level of societal consensus, and can be more comprehensive than model coverage based on endpoint estimation.

Figure 2-3 Stressor-impact chains model an emission from release, to midpoint, and on to multiple endpoints



(Udo de Haes, Finnveden et al. 2002)

Table 2-1 Life Cycle Impact Assessment models inventory data to common midpoints or endpoints

Life Cycle Impact	Relevant Inventory Data	Midpoint Modeling	Example Endpoints
Global Impacts			
Global Warming	Carbon Dioxide (CO_2) Nitrogen Dioxide (NO_2) Methane (CH_4) Chlorofluorocarbons (CFCs) Hydrochlorofluorocarbons (HCFCs) Methyl Bromide (CH_3Br)	Converts release data to carbon dioxide (CO_2) equivalents Note: global warming potentials can be 50, 100, or 500 year potentials.	Polar melt Soil moisture loss Longer seasons Forest loss/change Change in wind and ocean patterns
Stratospheric Ozone Depletion	Chlorofluorocarbons (CFCs) Hydrochlorofluorocarbons (HCFCs) Halons Methyl Bromide (CH_3Br)	Converts release data to trichlorofluoromethane (CFC-11) equivalents.	Increased UV radiation Skin cancer Cataracts Crop damage Marine life damage Immune system depression
Resource Depletion	Quantity of minerals used Quantity of fossil fuels used	Converts release data to a ratio of quantity of resource used versus quantity of resource left in reserve.	Decreased resources for future generations
Regional Impacts			
Photochemical Smog	Non-methane hydrocarbon (NMHC)	Converts release data to ethane (C_2H_6) equivalents.	Decreased visibility Eye irritation Respiratory distress Vegetation damage
Acidification	Sulfur Oxides (SOx) Nitrogen Oxides (NOx) Hydrochloric Acid (HCl) Hydroflouric Acid (HF) Ammonia (NH_4)	Converts data to hydrogen (H^+) ion equivalents.	Building corrosion Vegetation damage Soil quality decrease
Eutrophication	Phosphate (PO_4) Nitrogen Oxide (NO) Nitrogen Dioxide (NO_2) Nitrates Ammonia (NH_4)	Converts data to phosphate (PO_4) equivalents.	Excessive plant growth Oxygen depletion
Local Impacts			
Terrestrial Toxicity	Toxic chemicals with a reported lethal concentration to rodents	Converts LC_{50} data to equivalents.	Decreased production Biodiversity loss
Aquatic Toxicity	Toxic chemicals with a reported lethal concentration to fish	Converts LC_{50} data to equivalents.	Decreased production Biodiversity loss
Human Health	Total releases to air, water, and soil.	Converts LC_{50} data to equivalents.	Increased morbidity and mortality
Land Use	Quantity of land used in agriculture and for waste disposal.	Converts release data to a quantity of soil lost due to agricultural erosion Converts mass of solid waste into volume using an estimated density.	Decreased production Decreased land availability Habitat loss Biodiversity loss
Water Use	Water used or consumed.	Converts release data to a ratio of quantity of water used versus quantity of resource left in reserve.	Decreased production Habitat loss Biodiversity loss

(U.S. Environmental Protection Agency 2006)

2.2.8 Comparing Alternatives Using Life Cycle Interpretation

Life cycle interpretation, the remaining phase of the LCA process, is a systematic technique to identify, quantify, check, and evaluate information from the results of the LCI and the LCIA, and communicate them effectively. However, interpreting the results of an LCA is not as simple as two is better than three, therefore, Alternative A is the better choice. While conducting the life cycle inventory and impact assessment, it is necessary to make assumptions, engineering estimates, and decisions based on your values and the values of other involved stakeholders. Each of these decisions must be included and communicated within the final results to clearly and comprehensively explain conclusions drawn from the data. In some cases, it may not be possible to state that one alternative is better than another because of the uncertainty in the final results. This does not imply that the efforts have been wasted. The LCA process will still provide decision-makers with a better understanding of the environmental and health impacts associated with each alternative, where they occur (locally, regionally, or globally), and the relative magnitude of each type of impact in comparison to each of the proposed alternatives included in the study. This information more fully reveals the pros and cons of each alternative.

The purpose of conducting an LCA is to better inform decision-makers, who could be governmental officials, multi-national corporation leaders, non-governmental leaders (NGOs) or, ideally, multi-stakeholder panels, by providing a particular type of information (often otherwise not considered) with a life cycle perspective of environmental and human health impacts associated with each product or process. Since LCA does not take into account technical performance, cost, or political and social acceptance, it is recommended that the results be used in conjunction with these other parameters. Furthermore, LCA is viewed as providing some insights to the environmental pillar of the three pillars of sustainability, with the other two being economic and social. The development of the economic pillar is advancing through the efforts of various groups while the societal assessment is still in its infancy. Ultimately, all three complimentary pillars should be based on the same system boundaries and functional unit.

2.2.9 Limitations of Conducting an LCA

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

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

There are a number of ways to conduct life cycle impact assessments. While the methods are typically scientifically-based, the complexity of environmental systems has led to the development of alternative impact models.

The role of impact assessments is to categorize and quantify potential environmental effects. Once this is done, deciding whether one impact is worse than another is necessarily a subjective process in which the perceptions of the decision-maker are applied.

While LCAs can help identify potential environmental tradeoffs, converting the impact results to a single score requires the use of value judgments, which must be applied by the commissioner of the study or by the modeller. This can be done in different ways such as through the use of an expert panel, but it cannot be done based solely on natural science.

All assumptions or decisions made throughout the entire project must be reported alongside the final results of the LCA analysis. If assumptions are omitted, the final results may be taken out of context or be misinterpreted. As the LCA process advances from phase to phase, additional assumptions and limitations to the scope may be necessary to accomplish the project with the available resources.

2.2.10 Summary

Adding LCA to the decision-making process provides a better understanding of the human health and environmental impacts that traditionally is not considered when selecting a product or process. This valuable information provides a way to account for the full impacts of decisions, especially those that occur outside of the site that are directly influenced by the selection of a product or process. LCA is an environmental management tool that helps to inform decision-makers and should be included with other decision-support criteria, such as cost and performance in order to make a more balanced decision. While there is not always a straightforward or easy choice, it is important to understand the potential impacts related to each choice.

2.2.11 Further Reading

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2.3 Case Study: Replacing Methyl Tertiary-Butyl Ether (MTBE) with Ethanol as a Gasoline Oxygenate in a Typical U.S. Passenger Car

To help with the further understanding of LCA practice, the following case study is presented as an example. The information in this case study is based on research conducted by the U.S. EPA, in which methyl tertiary-butyl ether was compared to ethanol as an additive in gasoline used in passenger cars (U.S. Environmental Protection Agency 2007). This thesis author participated as a member of the research team. The case study is presented in parallel with the phases of an LCA as described in the previous section, i.e. goal and scope definition, inventory, impact assessment, and interpretation.

2.3.1 Background

Gasoline and gasoline additives are ubiquitous chemicals in our society. One gasoline additive, methyl tert-butyl ether (MTBE), has been used in U.S. reformulated gasoline (RFG) to create a cleaner burning fuel and to decrease emissions that lead to smog formation. After the adoption of MTBE as an oxygenate additive in gasoline, it was found to have become a contaminant in drinking water supplies, initially in California, then later in other parts of the United States. This unexpected environmental consequence prompted EPA to consider using a more holistic approach to consider a broad spectrum of multimedia environmental impacts over the full life cycle of gasoline and gasoline additives. In this case study, reformulated gasoline with MTBE as an oxygenate (RFG-MTBE) is compared to reformulated gasoline with ethanol as an oxygenate (RFG-EtOH).

2.3.2 Defining the Goal and Scope

Until recently reformulated gasoline was required to have 2.0% oxygen content by weight. To satisfy this requirement various oxygenates, such as MTBE and ethanol, had been added to gasoline. Although many oxygenate alternatives are available, the study was not intended to be a comprehensive assessment of all oxygenates. Instead, the goal was to identify the potential environmental tradeoffs in moving from MTBE to an alternative additive. Ethanol is the apparent preferred choice so it was selected for the study. Mixtures of 11.1% MTBE and 8.7% ethanol by volume (11.2% and 9.3% by weight) were considered.

It was anticipated that the results of this work could be used in support of the development of various EPA policies and rulemakings concerning oxygenates and oxygenated fuels, such as under TSCA Section 6 to ban or limit the use of MTBE. It was anticipated that the analysis could ultimately be of use to others outside the EPA both in the public and private sectors in the United States and abroad.

Since the releases and impacts of the ultimate use of the product, i.e. combustion in a passenger car, is an important aspect behind current air quality regulations, the project team decided to base the study on oxygenated gasoline, as opposed to studying individual oxygenates only. The project team also considered the basis of the study and whether the project should model the "US Fleet." The final decision was to base the assessment on a single car so that one can extend the results for a single vehicle by scaling up to millions of vehicles or by changing other scenario parameters. Decision-makers could then apply the results to either specific regions or the country as a whole.

Due to the number and complexity of processes involved in the three alternatives, a screening level assessment was deemed to be appropriate. Although, no specific guidelines exist on how to conduct such a streamlined approach, the basic idea was to use a mix of qualitative and quantitative generic data. The intent of screening is to provide 'directional' information regarding the environmental trade-offs between alternatives and highlight where the more significant impact areas occur. Screening can identify knowledge/data gaps and identify where to focus future data collection and impact assessment efforts, e.g. risk assessment of a chemical. In short, a screening LCA is comprehensive in coverage, but is generally conducted with less detail. Also, a screening LCA can be done in a shorter period of time and with fewer resources than a detailed LCA. A longer, more detailed discussion on streamlined LCA can be found in Section 2.8 of this thesis.

For RFG-MTBE, the total life cycle includes crude oil extraction, refinery operations, MTBE production, distribution/storage, and product use. For RFG-EtOH, the total life cycle also includes corn production (growing and harvesting), ethanol manufacture, distribution/storage, and product use. Transportation effects and impacts from energy use are considered at each life cycle stage as well. Typically, final disposal is included in an LCA, but in a study of gasoline, this stage is not included because the product is consumed in the use phase.

The function of the three product systems under review is to provide fuel for a generic 3200-pound passenger automobile in the United States driving 12,000 miles (19,312 kilometers) over a one-year period. The functional unit (i.e. number of gallons of gasoline and additive) was calculated for each system based on the typical fuel economy for this type of vehicle using the two fuel types (22 and 20 miles per gallon). Therefore, 545 gallons (2,063 liters) of the fuel with MTBE and 600 gallons (2,271 liters) of the fuel with EtOH would be needed.

Reformulated gasoline with MTBE (RFG-MTBE) is produced at petroleum refineries by blending blendstock gasoline with MTBE prior to transport to bulk terminal storage for regional distribution. It has been blended to form a fuel suitable for spark ignition engines and possesses different properties than other oxygenated fuels. These properties cause the fuel to behave differently during summer and winter months. Consequently, two blends of RFG-MTBE have been developed to accommodate changing weather conditions. An average summer and winter blend, assuming 50% summer and 50% winter, was used to develop the LCI for RFG-MTBE.

Table 2-1 Properties of reformulated gasoline with MTBE oxygenate (RFG-MTBE) used in this study to compare it with ethanol-oxygenated gasoline

Properties of RFG-MTBE	Summer	Winter	Average
Oxygen wt%	2.11	1.95	2.03
Reid Vapor Pressure, psi	6.78	10.6	8.69
Sulphur ppm	30	30	30.00
E200 F	47.9	55.7	51.80
E300 F	84.1	85.6	84.85
Aromatics vol%	19.5	19.1	19.30
Olefins vol%	11.9	13.2	12.55
Benzene vol%	0.55	0.61	0.58
Energy Content, Btu/gal	--	--	114,043
Fuel Density, g/gal	--	--	2,793
Fuel Economy, mi/gal	--	--	22

The LCI for RFG-MTBE is based on the following set of assumptions and data limitations.

- Lower heating value (LHV) of fuels was used in this analysis for all modes of transport and non-stationary engines. Higher heating value (HHV) was used for all utility and industrial boilers when calculating fuel consumption and emissions.
- MTBE production was modeled to have the same process inputs and outputs as 1-pound of blendstock gasoline produced from a petroleum refinery. MTBE production is identified as a significant data gap in this study.
- 95% of MTBE is produced at off-site facilities and shipped to the petroleum refinery while the remaining 5% is produced at captive (on-site) facilities within the petroleum refining boundary.
- Average U.S. petroleum refining process efficiency is 99.5% on a mass basis.
- Material Production and Delivery (P&D) contribution from drilling fluids used in crude oil extraction are negligible.
- The LCI for conventional gasoline consumed as a fuel for crude oil extraction and natural gas extraction is assumed to be similar to diesel fuel.
- Transport losses of petroleum products on ocean tankers, barges, rail, pipeline, and truck are less than 0.005% on average.
- Transport losses of liquid products were assumed to be released 50% to air and 50% to water as volatile organic compound (VOC) related emissions.
- Leaks (non-evaporative) from above-ground and under-ground storage tanks are negligible.
- RFG-MTBE gasoline was produced and distributed in a manner similar to conventional gasoline (Year 2000) consumed in the U.S.

Reformulated gasoline with ethanol (RFG-EtOH) is an oxygenated gasoline with 3.25% oxygen content. Ethanol is blended with blendstock gasoline in tanker trucks prior to departure for the refueling station. This blending process is different from RFG-MTBE in that it occurs at the bulk terminal storage as opposed to the petroleum refinery. An average between summer and winter blends, assuming 50% summer and 50% winter, was also used to develop the LCI for the ethanol-blended gasoline.

Table 2-2 Properties of reformulated gasoline with ethanol oxygenate (RFG-EtOH) used in this study to compare it with MTBE-oxygenated gasoline

Properties of RFG-EtOH	Summer	Winter	Average
Oxygen wt%	3.3	3.2	3.25
Reid Vapor Pressure, psi	6.76	11.6	9.18
Sulphur ppm	30	30	30
E200 F	46.1	58.3	52.2
E300 F	86.2	86	86.1
Aromatics vol%	18	18.3	18.15
Olefins vol%	3.9	5.9	4.9
Benzene vol%	0.83	0.82	0.825
Energy Content, Btu/gal	--	--	113,424
Fuel Density, g/gal	--	--	2,801
Fuel Economy, mi/gal	--	--	20

The LCI for RFG-EtOH is based on the following set of assumptions and data limitations.

- 100% of the ethanol is produced using corn as the feedstock.
- 60% of ethanol is produced via wet mill processing and 40% is produced via dry mill processing.
- The LCI for RFG-EtOH does not include inventory data for the production and delivery of pesticides, herbicides, and fertilizers, which were assumed to be small inputs (less than 1%).
- Lower heating value (LHV) of fuels was used in this analysis for all modes of transport and non-stationary engines. Higher heating value (HHV) was used for all utility and industrial boilers when calculating fuel consumption and emissions.
- Average U.S. petroleum refining process efficiency is 99.5% on a mass basis.
- Material Production and Delivery (P&D) contribution from drilling fluids used in crude oil extraction are negligible.
- The LCI for conventional gasoline consumed as a fuel for crude oil extraction and natural gas extraction is assumed to be similar to diesel fuel.
- Transport losses of petroleum products on ocean tankers, barges, rail, pipeline, and truck are less than 0.005% on average.

- Transport losses of liquid products were assumed to be released 50% to air and 50% to water as volatile organic compound (VOC) related emissions.

- Leaks (non-evaporative) from above-ground and under-ground storage tanks are negligible.

2.3.3 Inventory

Completing the life cycle inventory included the following steps: 1. Construction of the flow chart(s) to define the system boundaries and identify all the relevant operations, 2. Collection of data, and documentation, for all the processes in the system, including transportation, and 3. Calculation of environmental loads (resource use and pollutant emissions) for each process in relation to the functional unit.

Figure 2-4 Scope of the RFG-MTBE system

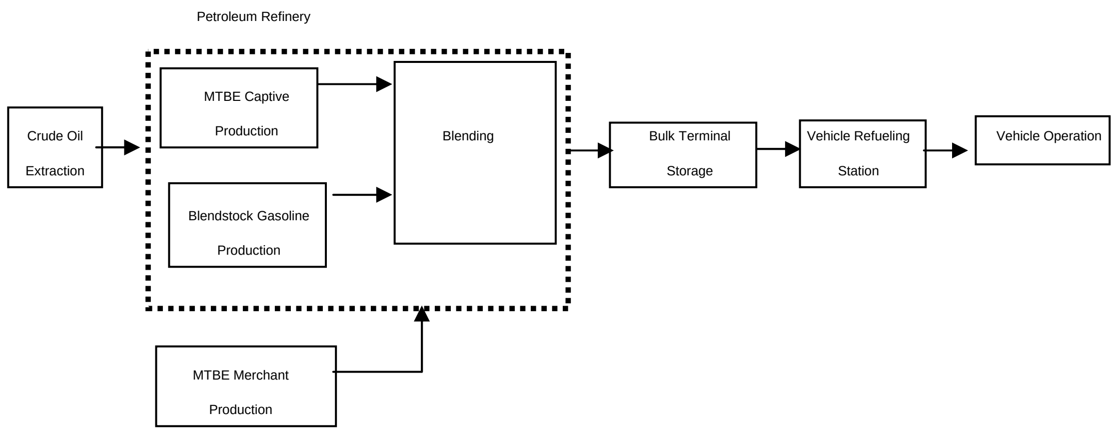
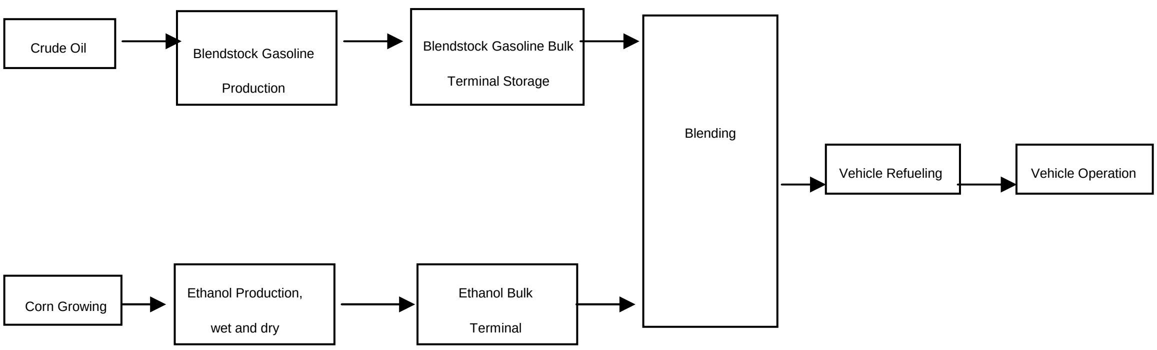


Figure 2-5 Scope of the RFG-EtOH system



Secondary data sources (i.e. data that are gathered and used for other purposes other than those for which they were originally collected) were obtained from various sources, including literature (government documents, reports, journals, books, etc.), computerized databases and information systems, and models of environmental processes. Using publicly-accessible data reduced the time and effort needed to complete the LCA as well as ensured the reproducibility of the results.

To maintain transparency in the inventory, data were collected and presented in Excel spreadsheets. The screening level approach is consistent with a "cradle-to-grave" analysis without the expectation of obtaining highly detailed or comprehensive data. Data quality was addressed through a contribution analysis and visual scanning of initial, intermediate, and final results for values that appeared systematically or randomly irregular. (A contribution analysis can determine if the most significant stressors are consistent with expectations. This relative comparison of inventory values requires engineering judgment to recognize whether some values appear too big or too small in various contexts.) Thus, while effort was expended to achieve accurate results, only the precision of some of the values could be verified.

2.3.4 Impact Assessment

The U.S. EPA's Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) was used to assess environmental impacts (Bare, Norris et al. 2003). TRACI contains models for the following ten categories:

- Acidification,
- Ecotoxicity,
- Eutrophication,
- Global Warming,
- Human Health Cancer,
- Human Health Criteria (pollutants),
- Human Health Noncancer,
- Ozone Depletion,
- Photochemical Smog Formation, and
- Fossil Fuel Depletion.

TRACI was applied to inventory data on resource depletion, air emissions and water discharges. Land use, water use, and solid waste effects were not modeled by TRACI but were maintained in the inventory. The impact assessment results are indicators of potential relative effects among the fuel alternatives in each category. The summary results of the TRACI models are given, by life cycle stage, in the following tables.

Table 2-3 Impact indicator results for reformulated gasoline with MTBE as an oxygenate (RFG-MTBE) and reformulated gasoline with ethanol as an oxygenate (RFG-EtOH)

Life Cycle Stage	RFG-MTBE	RFG-EtOH
Acidification (moles H+ equivalents)		
Raw Material Acquisition	160	312
Production	210	219
Use	522	523
Total	892	1050
Ecotoxicity (kg 2,4-D-equivalents)		
Raw Material Acquisition	1.02	14.8
Production	0.923	0.761
Use	2.75	2.75
Total	4.70	18.3
Eutrophication (kg N equivalents)		
Raw Material Acquisition	0.131	1.49
Production	0.116	0.108
Use	0.587	0.588
Total	0.83	2.19
Global Warming (kg CO_2 equivalents)		
Raw Material Acquisition	340	-259
Production	675	707
Use	5440	5490
Total	6450	5940

Human Health Cancer (kg Benzene equivalents)		
Raw Material Acquisition	0.028	0.031
Production	0.149	0.149
Use	0.442	0.448
Total	0.615	0.627
Human Health Criteria (Disability-Adjusted Life Years, DALYs)		
Raw Material Acquisition	82	108
Production	90	81
Use	191	191
Total	363	381
Human Health Noncancer (kg toluene equivalents)		
Raw Material Acquisition	2,160	2,270
Production	1,040	729
Use	302	304
Total	3500	3310
Ozone Depletion (kg CFC-11 equivalents)		
Raw Material Acquisition	1.63E-06	1.85E-06
Production	1.81E-05	1.65E-05
Use	1.44E-08	1.45E-08
Total	1.97E-05	1.83E-05
Photochemical Smog Formation (g NOx equivalent/m)		
Raw Material Acquisition	3.86	6.53
Production	5.02	4.67
Use	28.0	28.7
Total	36.9	39.9

Fossil Fuel Depletion (megajoules, MJ, surplus energy)		
Raw Material Acquisition	8.81E+03	9.21E+03
Production	2.38E+03	1.19E+03
Use	2.90E-01	2.92E-01
Total	1.12E+04	1.04E+04

2.3.5 Interpretation

Given the screening nature of this analysis, the findings of this work should not be considered conclusive in themselves, but should be considered simply as an insight to some areas that may deserve additional research and analysis. In line with the screening level analysis, assessments of uncertainty were not attempted, so one cannot mathematically define the significance of the differences found in the results for the alternatives.

Some areas which may warrant further investigation include the following:

Based on the limitations of the data and modeling conducted here, including the lack of impact assessment modeling site-specificity and not including the population density data, RFG-EtOH showed increased effects in land and water use, acidification, ecotoxocity, eutrophication, human health cancer, human health criteria, and photochemical smog formation, while it showed decreased global warming potential, human health noncancer, ozone depletion, and fossil fuel depletion. These results were calculated assuming additional corn production would be required within the U.S., however, determining whether additional corn growing will need to be produced, or whether the corn requirements will be met with existing corn growing was beyond the scope of this study.

Impacts from vehicle operation make it the dominant life cycle stage for each of the alternatives. Vehicle operation causes relatively large impacts in acidification, global warming, human health cancer, human health criteria, and photochemical smog formation. A concern in evaluating vehicle operation is the calculation of only the criteria pollutants (e.g., grouped as volatile organic compounds, VOCs) and hazardous air pollutants as emissions. Ethanol and other substance species could have significant effects on various impact categories, but the emissions have not been determined.

A relatively small difference can be seen between the alternatives in a higher photochemical smog formation potential for RFG-EtOH, mainly caused by greater VOC releases. The larger VOC emission rate associated with RFG-EtOH is mainly due to higher hot soak and running losses from vehicles.

The scenario modeled here simulated ethanol being produced and then transported separately to the refuelling stations. Although some have raised a concern that the associated releases from transportation would be unacceptable, when put into the perspective of the functional unit, a car driven 12,000 miles, these effects do not appear substantial in this study. Furthermore, when ethanol is burned as part of RFG-EtOH acetaldehyde emissions increase, however, the potential environmental impacts from acetaldehyde appear to be a small fraction of the total impacts for RFG-EtOH.

Although, the occurrence of MTBE in drinking water supplies has become a national concern, the findings of this screening level LCA indicate that the RFG-MTBE alternative does not show relatively higher impacts due to local releases of MTBE at refuelling stations. This is consistent with expectations, since LCA is best at addressing issues at a larger perspective, whereas MTBE leaking from underground storage tanks is a localized issue (and thus would be best modeled with more detailed groundwater models with localized input parameters). Precisely because MTBE's effects made an impact at a local level (and with a strong taste and odor) it was much easier to identify the chemical of concern and its source, whereas longer range and more persistent pollutants such as mercury may actually cause comparable or even greater damages in the long run, the actual damages are often difficult to pinpoint and trace back to the original source of emission. Also, LCAs do not typically include the issues of taste and odor.

An analysis was conducted to determine the alternative(s) which displayed the best or worst potential impact scores in each of the categories considered. At a minimum 5% difference, the impact assessment indicates that RFG-EtOH is the best alternative in some categories (global warming, human health noncancer, ozone depletion, and fossil fuel use) and the worst in others (ecotoxicity, eutrophication, acidification, photochemical smog, and human health criteria). These findings, and those described above, should be considered within the context of the constraints of the study.

LCAs should be used in conjunction with other environmental assessment tools because together they provide important perspectives on environmental issues, and all of this analytical information should be considered within the context of a very complex decision. In addition to localized environmental, health, and safety concerns, economic forces can significantly impact the decisions made at the time. For this reason, LCA is simply one more supporting piece of information that may be considered within the decision-making framework. In this case, a very site-specific localized risk assessment (including taste and odor) could be used in conjunction with a more comprehensive LCA to provide additional information which could be valuable for the decision-maker.

2.4 Paper II – The international workshop on electricity data for life cycle inventories

Curran, MA, Mann, M and Norris G (2005) "The International Workshop on Electricity Data for Life Cycle Inventories," *Journal of Cleaner Production* 13(8), pp853-862. DOI: 10.1016/j.jclepro.2002.03.001.

Sections 2.2 and 2.3 covered the basics of LCA practice, including a case study demonstration of its use. Now, this thesis turns to look at the practice from the inside perspective of the LCA practitioner, or someone who is attempting to conduct an LCA. Through this perspective, the specific challenges of creating life cycle inventory data become clearer. In 2001, recognized international experts gathered at a 3-day workshop in Cincinnati, Ohio, USA, to discuss inventory data specifically for the electrical power industry. The following paper summarizes the discussion and findings from the workshop. This meeting was a milestone in that it allowed for the first public discussion on consequential versus attributional LCAs. This distinction in LCA practice had not been delineated before. In addition, the paper summarizes the discussions on key issue areas, such as allocation, boundary definition, environmental flows, and new technologies. Although the workshop attendees did not resolve these issues, the discussion went a long way in helping to identify areas in need of further research.

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Note from the field

The international workshop on electricity data for life cycle inventories

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Abstract

A three-day workshop was held in October 2001 to discuss life cycle inventory data for electricity production. Electricity was selected as the topic for discussion since it features very prominently in the LCA results for most product life cycles, yet there is no consistency in how these data are calculated and presented. Approximately 40 people attended all or part of the meeting to discuss issues of data modeling and collection. Attendees included recognized experts in the electricity generation and life cycle assessment fields.

Five main topics of discussion were identified before the meeting began: (1) modeling the response of the energy supply system to demand (i.e. marginal versus average data); (2) defining the breadth and width of system boundaries to adequately capture environmental flows and data that are needed for impact modeling; (3) allocating environmental burdens across co-products that come from the same process; (4) modeling new and non-traditional technologies in which the data are highly uncertain; and (5) including transmission and distribution in modeling of electricity generation. Breakout groups addressed the first four topic areas in individual discussion groups and reported the results in a plenary session on the last day of the workshop (it was decided during the meeting to include "transmission and distribution" in other discussions).

A key success of the workshop was the creation of the larger network of LCA and electricity production experts which will provide a good foundation for continued discussions.
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Keywords: Life cycle assessment; Inventory; Electricity; Energy

1. Introduction

Data collection for life cycle inventories (LCIs) remains a critical factor in the successful completion of a life cycle assessment (LCA). Access to reliable data continues to be a significant barrier to the advancement and use of LCAs in environmental management.

Over the years, LCA practitioners have been left to their own means to collect and model inventory data as they have conducted studies for clients. However, these data are the property of the practitioner and not typically made available to the public, or they must be purchased. Furthermore, since different modeling assumptions can be made, there is no consistency in how these data are calculated and presented in different LCAs.

While most LCI data are specific to a particular study and its goal, there are data that are common in all LCIs, namely electricity, transportation and waste management. Electricity use, especially, features very prominently in the total LCA results for a majority of product life cycles. Therefore, the benefits of public LCI data on electricity generation would be high for those who undertake LCAs and for those who draw conclusions based on LCAs.

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2. Background

Electricity is a major consideration in any LCA. It is important to accurately calculate and model resource use and pollutant releases for activities related to the generation and distribution of electricity, such as how and where electricity is produced, with what input requirements and with what pollution and waste consequences. As LCAs are being conducted more frequently as part of overall environmental management approaches within both the public and private sectors, it is becoming increasingly important that LCI data become more readily available. Also it is vital that data be used consistently between LCAs in order to lead to more fairly comparable results and reliable conclusions.

Modeling of the environmental burdens of electricity production is far from a simple or straightforward task. Indeed, the electricity supply system is among the most complex of all the industries addressed in an LCA. This complexity arises from a number of factors, including:

- the broad geographic scope of power grids and electricity markets with power wheeling,
- the dynamics of supply despatch in response to demand changes, overlaid on daily and seasonal dynamics,
- the wide variation among generation stations in emissions and inputs per unit generation across and even within fuel types,
- the rapid ongoing evolution and regional variety of the electricity system and the regulatory environment in which it operates,
- the rapid and ongoing evolution of electricity generation technologies and uncertainties about future market penetration of new technologies, and
- the potentially long time frames and importance of electricity consumption for the life cycles of durable products.

Existing LCI data sets generally fail to capture the effects of these complexities. Of course, all models must be simplifications of reality to be useful, but the potential effects of these complexities upon the usefulness of LCI results from current databases warrant examination. Another priority issue for resolution is the lack of consistency in scope (both of environmental flows and technospheric flows) among existing databases for different regions, and even among alternative databases covering the same region.

In order to comprehensively address the issues involved in modeling data for electricity generation, it was decided to hold a three-day workshop with recognized experts in the electricity generation and life cycle assessment fields to work together to lead to agreement that could be used in developing a uniform/consistent electricity database for life cycle inventories.

3. Workshop attendees

Approximately 40 people attended all or part of the meeting. The list of attendees is given in the next section. The breakdown of representation is approximately:

Industry experts*	10%
Government experts*	15%
LCA practitioners	20%
LCA researchers	20%
Academia	17.5%
Other US EPA'ers	17.5%

*Experts in traditional and non-traditional electricity generation.

4. Identifying the issues

The following topical areas, referred to as the "issues," were identified by the workshop planners and used to organize the presentations and discussions:

- *Marginal versus average:* "Should LCA model the response of the energy supply system to demand?" (and consequences for co-product allocation).
- *Boundaries:* "How wide and broad should the boundaries be to capture environmental flows and data that are needed for impact?"
- *New and non-traditional:* "How should LCA model new technologies, in which the data are highly uncertain, and how should increased demand for new technologies be accounted for?"
- *Co-product allocation:* "How should environmental burdens be allocated across co-products that come from the same process?"
- *Transmission/distribution:* "How should T&D impacts be included in modeling of electricity generation?"

Prior to the workshop, a short summary document, entitled "*An International Workshop on Electricity Data for Life Cycle Inventories: Introduction and Overview (August 2001)*," was prepared to describe these issues and what they mean. To help initiate the thought process and stimulate responses from the invitees to the meeting, a series of 25 questions were also posed in the summary document.

Input on these issues was solicited from everyone who planned to attend the workshop as well as those who were interested in the effort but were unable to attend. Around 10 thoughtpieces and other background material, such as journal articles were submitted (see Suggested Reading below). A summary document was prepared from these submittals and distributed to everyone before the workshop ("*International Workshop on Electricity Data for Life Cycle Inventories: Summary of Feedback on Issues*," 18 October 2001) as well as posted on the website that was created expressly for the workshop (http://www.sylvatica.com/ElectricityWorkshop.htm).

The first day of the workshop began with an initial plenary session in which presentations were made on data sources and a summary of each issue area. This led to breakout working groups that were tasked on the second day of the workshop with discussing the issues and identifying where there was either consensus or disagreement.

- Address the issue of clarifying how the consequential approach might be applied in practice, with what models and data.

A fifth objective had initially been identified for the group, but was never engaged by the group as being particularly interesting, important, or clear:

Workgroup Members

Marginal versus Average Data:
John Abraham, US EPA
John Burckle, US EPA (retired), USA
Tomas Ekvall, Chalmers, Sweden
Bill Franklin, Franklin Associates, USA
Patrick Hofstetter, ORISE Post Doc, Switzerland
Greg Keoleian, University of Michigan, USA
Benoit Maurice, EDF, France
Greg Norris, Sylvatica, USA
Philippa Notten, University of Capetown, S. Africa
Scott Properzi, Energi D2, Denmark
John Sheehan, NREL, USA
Tom Tramm, Consultant, USA
Bo Weidema, 2.0 LCA Consultants, Denmark

New and Non-Traditional (NNT) Technologies:
Merwin Brown, NREL, USA
Joyce Cooper, University of Washington, USA
Rolf Frischknecht, ESU Services, Switzerland
Douglas Gyorke, NETL, USA
Marty Heller, University of Michigan, USA
Wolfram Krewitt, ITT, Germany
Ivars Licis, US EPA
Lynn Manfredo, SAIC, USA
Maggie Mann, NREL, USA
Jonathan Overly, University of Tennessee, USA

Boundaries & Co-Product Allocation:
Jane Bare, US EPA
Bill Barrett, NRC Post Doc, USA
Jamie Meil, Athena Institute, USA
Michael Overcash, North Carolina State University, USA
Bev Sauer, Franklin Associates, USA
Rita Schenck, IERE, USA
Caroline Setterwall, Vattenfall, Sweden
Tim Skone, SAIC, USA
Ray Smith, US EPA

5. Summaries of the discussions on the issue areas

Summaries of the discussions on marginal versus average, boundaries, co-product allocation, and new and non-traditional were presented to the group in plenary. The originally planned discussion on transmission and distribution (T&D) was folded into the discussions under both boundaries and new and non-traditional. The sections below describe the workgroup sessions.

5.1. Deliberations and conclusions from the breakout group on marginal versus average modeling

The group began by clarifying its objectives. They were identified as follows:

- Clarify terminology, define the meanings of key terms.
- Determine when attributional and consequential LCI are each appropriate.
- Characterize the feasibility of attributional and of consequential LCI as applied to electricity supply, in terms of:
  - Cost and time.
  - Data availability, quality, and uncertainty.
- Determine whether there are different or equivalent answers to the above four issues, depending on whether one is addressing either of the following two application areas:
  - Electricity LCI data for use in other, general LCIs.
  - Using LCI to compare electricity generation options.

5.1.1. Terminology

In defining and clarifying terminology, we built on the contributions of Tomas Ekvall.

Decisions mean initial disturbances or changes to some part of the LCI system. Examples of decisions include whether to locate a new factory in a given region, or whether to install a high-efficiency device rather than a standard-efficiency device, or whether to pass more stringent building codes or appliance efficiency standards.

Decisions lead to *Consequences* through whole series or chains of cause–effect relationships. Other synonyms for consequences include effects and outcomes. Consequences of interest, for example, would include emissions from electricity generation, and investments new kinds of power generating capacity in particular.

Both decisions and consequences can have the properties of timing, duration, and magnitude. It is magnitude which leads to the definition of "marginal."

Marginal disturbances or perturbations are infinitesimal disturbances; e.g., installing one new end-use is a small but not an infinitesimal disturbance. A marginal disturbance is in theory infinitesimal, but in practice it is small enough to be approximated as an infinitesimal disturbance. This requires that the response to the disturbance be proportional to the magnitude of the disturbance.

Marginal consequences are the response of the system to a marginal disturbance. For example, the marginal consequences of a very small increase in electricity demand may include slight increase in air pollutant emissions and fuel consumption.

The workgroup's discussion moved from using the term *"marginal versus average" to "consequential versus attributional."* Prior authors have used terminology to differentiate "marginal versus average" LCI, and they have also labeled the options as "retrospective versus prospective" LCI. The breakout group determined that the central distinction being considered by this breakout group was one best described as differentiating "attributional" versus "consequential" LCI. Attributional and consequential LCIs are modeling methods which respond to different questions:

- attributional LCIs attempt to answer "how are things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window?" while
- consequential LCIs attempt to answer "how will flows change in response to decisions?"

Finally the group noted that *retrospective* LCIs are LCIs about prior situations or changes/decisions which occurred in the past, while *prospective* LCIs are about future situations or changes/decisions. An LCI can therefore be prospective attributional (how will things be flowing in the future?), prospective consequential (how will a future decision change flows?), retrospective attributional (how were things flowing in the past?) and retrospective consequential (how did a prior decision change the flows?).

5.1.2. When are attributional and consequential LCI each appropriate?

The *attributional approach* to LCI serves to allocate or attribute, to each product being produced in the economy at a given point in time, portions of the total pollution (and resource consumption flows) occurring from the economy as it is at a given point of time. Thus, annual electricity production from hydropower in the Pacific Northwest would be assigned or attributed to each of the uses of kWh of electricity occurring in the Pacific Northwest during that same year.

The rules used to define which processes are in or out of the system in attributional modeling are those based on an observation of how materials and energy are flowing in the system at the given point of time. For example, if concrete is made with 1 kg fly ash and 1 kg Portland cement per unit of concrete output, then the LCI model will show these flows into and out of the concrete manufacturing process.

Note that the "given point of time" could be past, present, or future.

The *consequential approach* to LCI attempts to estimate how flows to and from the environment will change as a result of different potential decisions. In general, the system response to changes in output demand (e.g., increased or decreased demand for some product) will vary between the short- and long-term. In the short-term, the response will be changes in output from existing production capacity (e.g., existing power plants, factories, etc.). In the long-term, the response will be changes in the timing, and perhaps the nature, of investments in new production capacity.

The rules used to define which processes are in or out of the system in consequential modeling are those based on an estimation of how material and energy flows will change as a result of the potential decisions or disturbances. In the fly ash example, if the output of fly ash is constrained—namely, if it is fixed based on the demand for electricity—then increases in the demand for high-fly-ash-concrete will not change the output of fly ash in the short run. Instead, it would increase the output of concrete made 100% from Portland cement. The consequential LCI model would attempt to take such output constraints explicitly into account.

5.1.3. Characterizing the response of the electric utility system to demand changes

Some members of the breakout group were familiar with realities of how the electricity system (at least in the US) currently responds to changes in demand. Others were familiar with responses of electricity systems in Europe. From their input, the following general facts were captured:

(1) When the results over a year are aggregated, the short-term output responses to electricity demand changes typically occur at plants that have the highest variable cost among those operating at the time of the demand change.[1]

[1] Note that on an hourly basis there are exceptions. For example, hydropower is often dispatched to meet daily peaks rather than base load. Hydro units respond more reliably than more complex generating options, so they are scheduled to come on to meet the daily peaks or to address local environmental concerns. However, limited water supply means that there are only so many kilowatt-hours available per year from a hydro unit, so by the end of the year, demand changes accruing during the year will not have affected the output from the hydro unit.

(2) In the long term, the type of new capacity added is generally the one which is estimated (by investment decision makers) to satisfy the given load shape at the lowest overall cost.
(3) The future is irreducibly uncertain, while the electricity supply system is dynamic and evolving. Thus, there are important levels of irreducible uncertainty concerning how the electricity supply system will respond to demand changes, even if we used the most sophisticated models available.

In addition, it is noted that in contrast with many other products, electricity has the specific characteristic that it cannot be stocked directly. At any moment, production must be equal to the sum of consumption and transmission losses. Throughout the day, the load shape varies greatly due to increasing and decreasing use, such as lighting at night. To produce electricity, utilities typically have different power plants which are able to adapt their production to the consumption, producing electricity as base load, (e.g., nuclear energy), semi-baseload (e.g., coal, gas, fuel power plant) and peak load (e.g., gas turbine). This element has to be taken into account when one tries to characterize the response of the electric utility system to demand changes. A "base load use" or a "peak load use" will not have the same answer. Rather than using simple assumptions to characterize electricity production, LCA practitioners should model for electricity planning which allows for the integration of such parameters.

5.1.4. *Appropriateness and feasibility of each method*

The participants agreed that, "ideally," LCA results would inform decision makers about the consequences of decision options that they are evaluating. However, there remained a significant level of concern about switching from attributional to consequential LCI modeling. Perhaps this is because the participants had not, with only two exceptions, ever undertaken or read the results of a consequential LCA.

Group participants had the following questions about consequential LCI:

- Does the change from attributional to consequential LCA (A → C) affect the results of the LCI? How much? In what cases, i.e., which product types, in which geographic regions?
- Does A → C alter LCA-based decisions?
- How easy will consequential LCA results be to explain to users of the results?
- How easy will consequential LCA be to perform?

5.1.5. *Recommendations*

Based on its deliberations and concerns, the breakout group concluded with the following recommendations:

(1) LCI databases should be developed in a way that is *technology-based*, so that the data can support either attributional or consequential modeling. Specifically, they should:
   (a) not aggregate over different technology types within a sector and
   (b) not aggregate over markets.
   This will require solving issues around the protection of confidential information, such as is already faced by developers of transparent LCI databases.
(2) LCI databases should contain ample meta-data, so that users can make informed modeling decisions to use the data for either attributional or consequential modeling.
(3) Feasibility studies which apply energy system models are needed in order to generate short-, medium- and long-term LCI results for a modest incremental change in demand for different regions, and for different types of end-use, which are characterized by differences in timing (daily and seasonal) and duration. Such studies would provide answers to all four of the questions posed by group participants about currently unknown aspects of consequential modeling of the electricity supply system.

5.2. *Deliberations and conclusions from the breakout group on boundaries and flows*

LCA attempts to approximate the comprehensive treatment of the environmental, health and resource burdens associated with product systems. In theory, this comprehensiveness entails inclusion of "all significant" burdens (e.g., pollution releases, resource consumption flows, or other impacts) from "all" causally connected processes. Thus, the system boundary for a life cycle inventory model requires a series of choices along two dimensions: environment and supply chain. The purpose of the boundary and flows workgroup (WG) was to discuss the following topics related to assembly and handling of electricity LCI data:

1. Which activities and operations along the supply chain should be included? That is, how wide and how broad should the system boundaries be drawn? (e.g., should capital equipment be included? transport of workers to the production sites? service sector inputs such as from designers, lawyers, accountants, advertising, etc.?).
2. Based on prior LCA and non-LCA environmental evaluations of the electricity supply system, is there a set of environmental flows for which reporting in LCI databases should be required? Is it possible to define a recommended set of environmental flows that would be sufficient to include in databases?
3. What is the most commonly accepted system of nomenclature for environmental flows?

The workgroup successfully addressed the first two questions/topics, however, the scope of the third question was determined to be too broad and extensive to be covered within the limited meeting time of the WG.

5.2.1. Boundaries

The participants evaluated which activities and operations along the supply chain/life-cycle should be included for energy supply systems. Particularly, they discussed what should be included (e.g., should capital equipment be included? transportation of workers to the production sites? service sector inputs such as from designers, lawyers, accountants, advertising, etc.?). The consensus was to include infrastructure only for dedicated resources. For example, the material used to construct a boiler used in a coal-fired utility plant should be included, but the materials used to construct the cranes that are used to erect boilers and other plant structures would not be included. Likewise, impacts from workers traveling to and from work should be excluded. This is not a hard-and-fast rule, but more a general rule of thumb to be used in drawing boundaries for energy supply systems. The potential impact from infrastructure operations should always be evaluated, even on a cursory level, to support the exclusion with confidence.

Taking a step back, the workgroup also evaluated the main processes or activities that should be considered when conducting an LCI of any energy supply system. The results of this effort are illustrated in Fig. 1. Process or activities identified in Fig. 1 for energy supply systems should not be excluded without proper process knowledge. If excluded, the corresponding rationale should be documented in a transparent manner and provided with the results of the LCI. The specific nomenclature for each process or activity identified in Fig. 1 may vary from one practitioner to another, but the intent of each box should be evaluated for each LCI.

5.2.2. Environmental flows

The boundaries and flows workgroup evaluated the feasibility of a “default” or “standard” list of environmental flows for electricity supply systems. The consensus of the workgroup and the workshop attendees was that a “default” list would provide the perception that only those environmental flows were of significant concern and all others could be excluded. This is not true. Our ability (as LCA practitioners) to understand the impacts from energy supply systems is based on previous experience (past LCA work) and to a greater extent, the availability of data to model the energy supply system. Future efforts to model the energy supply system should not be limited to previous experiences or perceived understandings of significant environmental flows; rather, every effort should be made to challenge the validity and accuracy of scientific knowledge upon which the conclusions about energy supply systems are drawn.

Therefore, the workgroup rephrased the question to ask “is there a minimum list of environmental flows for energy supply systems that one should expect to be included in an LCI?” With some apprehension, the

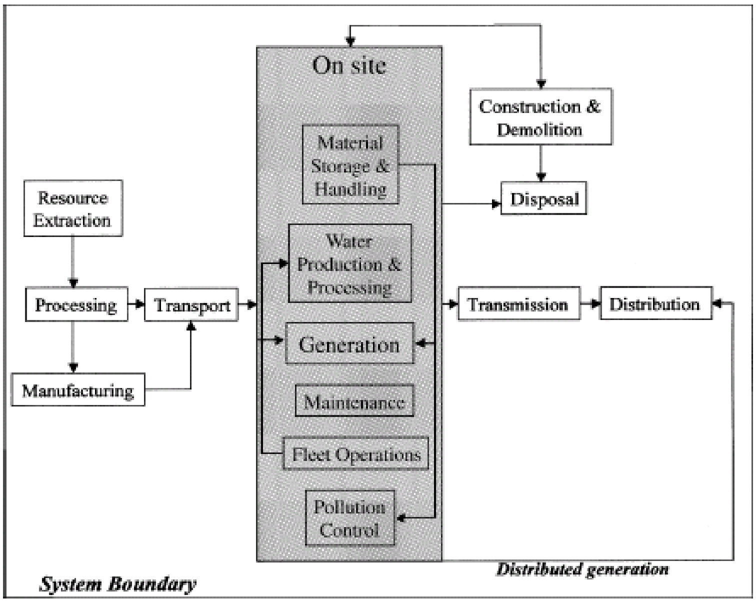

Fig. 1. System boundary for energy supply systems.

Table 1
"Minimum" list of environmental flows for energy supply systems

Resources	Air emission	Water emissions
Water (location and type)	CO_2	Chemical oxygen demand (COD)[a]
Fuel (in ground)	CO	TDS
Minerals (in ground)	PM (10, 2.5)	Total suspended solids (TSS)
Biomass (harvested)	CH_4	Biological oxygen demand (BOD) (5, 7, 10)[a]
Land use	SO_X	Flow
(area and location)	NO_X	Temperature change,[b] or thermal loading in energy units
Wastes	NH_3	NH_3 (as N)
Solid waste	Hg, Pb	Total Kjeldahl nitrogen (TKN) (as N)
Radioactive waste (H, M, L)	VOC (NM)	NO_3, NO_2 (as N)
Hazardous waste	Dioxin	Polycyclic aromatic hydrocarbons (PAH's)
	PAHs	Phosphates (as P)
Other releases	SF6	Cu, Ni, As, Cd, Cr, Pb, Hg
Radionuclides	HFCs	

[a] COD and BOD are indicators of water quality rather than flows.
[b] Limitation on temperature depends on the temperature of the river.

workgroup developed a tentative minimum list of environmental flows to be considered for energy systems (see Table 1). Exclusion of these environmental flows should raise concern towards the comprehensiveness of the LCI data set.

5.2.3. Next steps for boundaries and flows research

The workgroup identified the following next steps to continue the progress made during the electricity workshop.

1. Apply the system boundaries and environmental flows guidance in the development of the following model energy supply systems:
   a. Coal w/anthracite
   b. Coal w/lignite
   c. Natural Gas
   d. Oil
   e. Nuclear
   f. Hydro
   g. Wind
   h. Biomass
   i. Geothermal
   j. Other
2. Research the potential impact from the following, and other, non-traditional environmental flows:
   a. Noise
   b. Radiation
   c. Biological resources

5.3. Deliberations and conclusions from the breakout group on co-product allocation

Co-product allocation arises as an issue whenever a process produces more than one useful product. For example, steam turbine systems may sell both electricity and low pressure steam as useful products. When co-products are present, practitioners must determine how much of the burdens associated with operating and supplying the multi-output process should be allocated to each co-product. Practitioners must also decide how to allocate environmental burdens across co-products when one is a waste stream that can be sold for other uses.

The ISO standards for LCA, particularly ISO 14041 on inventory analysis, provide methodological guidance on this issue. But they call for practitioners to attempt to avoid allocation if possible; and secondly, to attempt modeling approaches which reflect the physical relationships between the process outputs and its inputs. In summary, proper application of the ISO guidelines on allocation requires a physical understanding of the co-product production processes. The consensus of the workgroup was to follow the guidance outlined in ISO 14041 for energy supply systems. The following highlights some key issues related to allocation per ISO 14041.

ISO 14041 requires the following procedure be used for allocation in multifunction processes:

- Allocation should be avoided, wherever possible, either through division of the multifunction process into sub-processes, and collection of separate data for each sub-process, or through expansion of the systems investigated until the same functions are delivered by all systems compared.
- Where allocation cannot be avoided, the allocation should reflect the physical relationships between the environmental burdens and the functions, i.e., how the burdens are changed by quantitative changes in the functions delivered by the system.
- Where such physical causal relationships alone cannot be used as the basis for allocation, the allocation should reflect other relationships between the environmental burdens and the functions.

For allocation in open-loop recycling, ISO 14041 recommends the same procedure but allows a few additional options. If the recycling does not cause a change in the inherent properties of the material, the allocation may be avoided through calculating the environmental burdens as if the material was recycled back into the same product. Otherwise, the allocation can be based on physical properties, economic value, or the number of subsequent uses of the recycled material. The international standard does not include information on the effect of the different methods on the life cycle modeling, for example the feasibility of the methods, the amount of work required, or what type of information that results from the application of the methods.

A major point which came to light during the workshop discussions on allocation was that the choice of allocation method depends considerably upon whether the LCA is being performed from an attributional or a consequential point of view. This point is demonstrated in the dissertation and publications of Ekvall and Tillman [1]. In this paper, they very helpfully differentiate cause-oriented from effects-oriented bases for allocation, and suggest that for LCAs supporting decisions about the future (e.g., for consequential LCAs), effects-oriented basis for allocation is appropriate. System expansion is an effect-oriented approach, while economic allocation is a cause-oriented approach.

This issue of the relationship between consequential/attributional LCA and the choice of allocation method is also discussed together with a detailed presentation of system expansion methods for allocation in a 2001 paper by Weidema [2].

5.4. Deliberations and conclusions from the breakout group on new and non-traditional technologies (NNT)

In working on the question of how to conduct LCAs of NNT technologies for electricity generation, the group felt it was necessary for the purpose of this discussion, to distinguish their role as database developers and not LCA practitioners.

The goal of the database effort was determined to be three-fold: (1) provide good inventory data for each NNT generation technology, (2) provide guidelines and/or models that will help practitioners choose the correct electricity mix, and subsequent environmental stressors, for their product life cycle assessment, and (3) ensure consistency.

5.4.1. Scope of NNT generation

Areas of interest for NNT generation include:

- Future technologies
- Renewables
- Non-baseload generators
- Distributed generation

Future technologies include those that have the potential to someday contribute significantly to the grid mix, but do not currently influence the environmental impact of common electricity usage. For these technologies, there is limited operating data, which are almost never site-specific. While actual operating conditions are difficult to predict with certainty, these technologies are often viewed as being more environmental benign. Future technologies may include the second category, renewables, but will also include generation options such as fuel cells, microturbines, and advanced coal.

Difficulties that arise in conducting LCAs on renewables present challenges for LCI database developers. For example, in LCIs that are based on a functional unit of producing a kWh, operating emissions may be very low or essentially zero. The predominant source of emissions associated with the generating technology may be construction emissions, which are problematic to allocate over the functional unit of kWh.

Additionally, because some impacts can be very different than those from traditional generators (e.g., bird kills), the database must be flexible enough to include different stressors. Finally, an important driver for renewables is the avoidance of conventional generation and impacts. Future discussions on database development will need to agree on how avoided impacts are handled.

Non-baseload generators are those that do not produce power on a continuous or controllable basis. Examples include some renewable generators such as wind and photovoltaics (PV), or power plants that are used for providing peak energy. With regard to database development, care must be taken in data sets when referencing stressors to a functional unit. The functional unit can be defined either from the supply-side as the kWhs that come from the generator itself, or from the demand-side as the kWhs that are consumed by the user of the electricity.

The drivers for distributed generation are the demand for reliable power, the desire to avoid down-time costs, and the mitigation of significant up-front capital expenditures for large generators and transmission and distribution infrastructure. Distributed generators (DGs) are typically small, and may use fossil or renewable fuels. For a large penetration into the grid, LCA practitioners may take account, during impact assessment, of the fact that emission source locations are distributed over a large geographic region as well. Additionally, depending on the reason for a DG installation, the functional unit may not be kWh.

5.4.2. Focus of the discussion

In the course of the discussion on NNT generation, four questions were answered:

- How are data sets constructed for new technologies, for which there are higher degrees of uncertainty in environmental stressors?
- Is there a need to develop a common future energy scenario that considers renewable and distributed energy sources for use in prospective LCAs?
- How should distributed generation be accounted for in national or regional energy grid data?
- What percent of the grid mix does a technology have to supply before we care about it in our product LCAs?

For the entire database, the group felt very strongly that all data should be kept as unaggregated as possible.

That is, each set of data should not represent the cradle-to-gate inventory for the technology it is describing. Rather, construction, mining, transportation, and operation should be provided in data modules such that a user can separate them out.

5.4.3. Results of the discussion

The questions posed above were answered as follows:

1. How are data sets constructed for new technologies, for which there are higher degrees of uncertainty in environmental stressors?
   - Use best available mass and energy and production data.
   - Where there are data gaps, make a conservative expert judgment for missing data points and document assumptions (SETAC working group).
   - Include a calculation routine that allows the users to vary performance/emissions parameters.
   - Document assumptions, sources of data, and year in which data were obtained.
   - Be alert to the situation where you need to input stressors that are not common to current generation technologies (e.g., bird kill, land use).
2. Is there a need to develop a common future energy scenario that considers renewable and distributed energy sources for use in prospective LCAs?
   - No. However, there is a need to provide for the application of various future energy scenarios.
   - Provide a tool or modules that describe different energy mixes/scenarios.
3. How should distributed generation be accounted for in national or regional energy grid data?
   - The same way that traditional generators are accounted for.
   - Different transmission and distribution losses are important.
   - Stressors from non-baseload generation should be discounted to the percent of time that they supply electricity to the consumer.
4. What percent of the grid mix does a technology have to supply before we care about it in our product LCAs?
   - If you can assemble life cycle inventory data for a technology, provide it in the database.
   - Use the module/tool described in list-item 2) to give the user an opportunity to incorporate them into their grid mix, or they can do it manually.

In addition to the issues described above, other concerns should be considered in future related activities. Of key importance is the incorrectness of using current data for future technologies. Conclusions regarding the environmental benefits that could be achieved with future technologies would be misguided when significant technological advancement is possible. Similarly, while the database is to contain inventory data for the various technologies, LCAs conducted for different timeframes will need to take into account predictions of different grid mixes.

6. Conclusions

The workshop successfully met its stated goal to facilitate the exchange of ideas and information. As was identified in the issues paper and follow-up discussions, the information needs to be established are too numerous to be fully explored or resolved at a brief three-day workshop. However, the hard work of the breakout groups led to many of the discussions points being advanced.

- The workgroup on marginal data made an important distinction in terminology by defining "marginal," "attributional," and "consequential." While there was much discussion and many questions remained unresolved, the group did achieve consensus on the following recommendations:
  - LCI databases should be developed in such a way that they support *both* attributional and consequential modeling.
  - There is the strong need for case studies of consequential modeling of the electricity system in order to shed light on many of the current questions surrounding this rather new and unfamiliar approach in LCI.
- The workgroup on boundaries created a first cut at a "minimum list" of environmental emissions that should be included in the inventory.
- The workgroup on new and non-traditional technologies noted that despite difficulties that arise in conducting LCAs on renewable generating technologies, due to uncertain operating data, any database on electricity must be flexible enough to include non-traditional stressors (e.g., bird kills).

A consistent thread throughout all the conversations was the desire for having access to unaggregated data, although the practicalities involved in this, such as confidentiality issues, were not discussed.

A key success of the workshop was the network that was created among experts in the LCA and electricity production fields. The establishment of this larger workgroup will provide a good foundation for continued discussions.

The workshop conveners are exploring next steps, and encourage all workshop participants as well as other interested parties to please use the workshop website (http://www.sylvatica.com/ElectricityWorkshop.htm) as a repository for documents, thoughtpieces, and links which relate directly to the topics discussed at the workshop and summarized in this document.

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2.5 Paper III - Status of LCA as an environmental management tool

Curran, MA (2004) "The Status of Life Cycle Assessment as an Environmental Management Tool" *Environmental Progress* 23(4), pp277-283. DOI: 10.1002/ep.10046.

As demonstrated previously, general agreement has been reached on the overall structure of the LCA framework, i.e. life cycle inventory, life cycle impact assessment and interpretation. The following 2004 paper continues to identify where additional research is needed. The paper shows that there are a growing number of LCA tools and databases that have been developed, and that various countries have efforts underway to create national life cycle data. In addition, there is beginning to be consensus on common impact categories and models. However, as seen here and in the previous paper on the electricity data workshop, there remain critical unresolved issues related to LCA practice and application. The following paper further expands on past work to identify these unresolved issues. The paper identifies three key barriers which are preventing the widespread application of LCA as an effective environmental management tool: 1. Lack of awareness for the need to look holistically at our actions; 2. Lack of reliable publicly-available data, and 3. Lack of consensus on impact assessment categories and models.

The Status of Life-Cycle Assessment as an Environmental Management Tool

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As environmental awareness increases, industries and businesses have responded by providing "greener" products and using "greener" processes. Many companies have found it advantageous to explore ways of moving beyond compliance, using pollution prevention strategies and environmental management systems, to improve their environmental performance. One such way is life-cycle assessment (LCA). This concept considers the entire life cycle of a product, process, or an activity. Evaluating environmental impacts holistically from raw material acquisition, through manufacture, use, and disposal using a life-cycle perspective is gradually being viewed by environmental managers and decision makers as an important element in achieving environmental sustainability. The use of a holistic, life-cycle approach can help industry and government avoid the unintended trading of one environmental problem for another. This paper addresses ongoing research in LCA methodology in the US EPA's National Risk Management Research Laboratory. Two key areas of this research include life-cycle inventory (LCI) data availability and life-cycle impact assessment (LCIA) modeling.

To address the issue of data availability, EPA developed a website entitled LCAccess. Although LCAccess does not itself contain data, it is a searchable global directory to potential data sources. LCAccess also serves as a central source for LCA information. In addition, EPA developed TRACI (the Tool for the Reduction and Assessment of Chemical and other environmental Impacts), which allows the characterization of ozone depletion, global warming, acidification, eutrophication, tropospheric ozone (smog) formation, ecotoxicity, human particulate effects, human health, fossil fuel depletion, and land use effects. TRACI can be used with life-cycle inventory to further analyze and understand the data. © 2004 American Institute of Chemical Engineers Environ Prog, 23: 277–283, 2004

INTRODUCTION

There is a growing awareness that a single-issue approach to an environmental problem may not lead to an effective long-term strategy. Instead, governments and industries around the world are seeing the value and need to look at the entire life cycle of products and processes from cradle to grave and across all media (air, water, and land). In particular, the interest by regulatory bodies is influencing the pace at which the life-cycle concept is being adopted in environmental management approaches. The European Commission's Integrated Product Policy (IPP) is a recent example of how policy makers are recognizing the need for broader-based strategies. There is reasonable consensus that one of the key underpinnings of IPP is that it should be "life-cycle based," as well as "market facing and integrated" [1]. In the US, it is becoming increasingly evident that a holistic view, such as life cycle assessment (LCA), is needed if we are to achieve environmental sustainability [2].

Figure 1 depicts the framework for LCA methodology as specified in the series of International Standards Organization (ISO) standards on LCA. This series is composed of the following documents:

ISO 14040: Principles and Framework
ISO 14041: Goal, Scope, and Inventory Analysis
ISO 14042: Impact Assessment
ISO 14043: Life Cycle Interpretation
ISO 14047: Examples of Impact Assessment

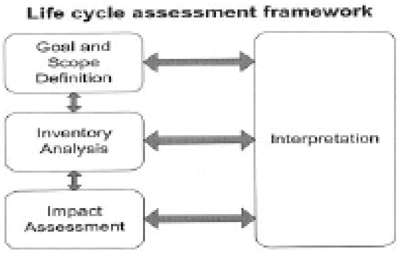


Figure 1. Life-cycle assessment framework.

ISO 14048: Documentation of Format
ISO 14049: Examples of Inventory Analysis

As would be expected from such a wide-encompassing view, LCA encounters practical limitations and barriers that have slowed its widespread adoption in environmental management. The three key barriers are:

1. Lack of awareness of the need to look holistically at the overall impacts of actions.
2. Lack of reliable, publicly available data that can be used to create life-cycle inventories.
3. Lack of an agreed-upon life-cycle impact assessment model.

The EPA's Life Cycle Assessment research program in the National Risk Management Research Laboratory in Cincinnati, OH is addressing the life-cycle inventory data availability and impact modeling issues through its research efforts. Highlights of these efforts are provided in the following sections.

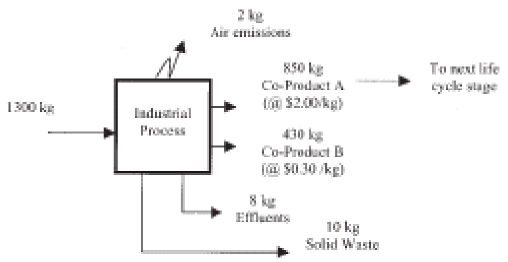
LIFE-CYCLE INVENTORY (LCI) DATA

Completing an LCI requires the acquisition and synthesis of a significant amount of data. Data come from many different sources, such as: proprietary, company data; consultants, labs, and universities; and public databases and sources. However, it is not uncommon for such databases to use different assumptions on how to model the data, such as which units, reference flows, or time periods to use.

An example of such an assumption is the allocation of environmental releases from a facility that produces more than one product. Figure 2 shows three schemes as to how the results for air, water, and solid waste releases from a process that produces two coproducts can vary depending on the basis used for the calculation. Scheme 1 is based on the weight of coproduct A compared to coproduct B. Scheme 2 is based on market value, where coproduct A is being sold at $2 per kilogram. Scheme 3 views coproduct A as the main product intended to be produced by the process and allocates 100% of the emissions to A (for example, mine waste, although not an intended coproduct, can be sold as roadbed material, yet, none of the releases would be attributed to it).

Other issues related to creating LCI data include the following:

- Use of specific vs. general source data
- Use of measured, calculated, or estimated data
- Precision as a measure of variability
- Completeness as a percentage
- Representativeness
- Consistency
- Reproducibility
- Reliability of data source
- Mass and energy balances
- Normative choices made before modeling the product system, such as coproduct allocation
- Choice of data sources and data quality requirements
- Enforcement of the accounting rules
- Processing of data choices
- The LCI calculation method
- Significance analysis
- Presentation of results
- Meeting the goal of the study



Scheme	Air Emissions	Effluents	Solid Waste
1. By weight	1.33 kg	5.31 kg	6.64 kg
2. By market value	1.86 kg	7.47 kg	9.29 kg
3. By "intent"	2.00 kg	8.00 kg	10.00 kg

Figure 2. Three possible schemes for coproduct allocation of life-cycle inventory data (for an assessment of Product A).

The key consideration in creating an LCI is the maintenance of transparency of how the data were arrived at (modeled, estimated, calculated, etc., and the underlying assumptions) so that the end user has full information of the data.

LCI Data Availability

Efforts are currently under way to make inventory data more accessible to users who are attempting to conduct an LCA. There are two notable efforts within the United States: the LCI Database Project and the EPA's *LCAccess* data portal.

LCI Database Project

The LCI Database Project is being coordinated by the National Renewable Energy Laboratory (NREL) and cosponsored by the EPA, GSA, US Army, and members of the auto industry. The project was initiated on May 1, 2001, and gained national prominence later that month at a meeting of interests hosted by the Ford Motor Company. During the meeting, funding agencies and other participants representing industrial, academic, and consulting communities voiced strong support for the LCI database project. As a result of this meeting, an advisory group with 45 representatives from manufacturing, data, government, and nongovernment organizations, as well as LCA experts, worked together to create data development guidelines for the project. The group reviewed the proposed data collection process and provided a plan for the execution of subsequent phases of the database project, including recommended data development priorities and a preliminary cost estimate. In addition, an independent panel produced a review of the data development guidelines.

The life-cycle inventory database provides information as a set of modules. Each module, or unit, quantifies the environmental impact of a process encountered during product manufacturing, use, and disposal. The data will follow a single protocol, be critically reviewed, and be maintained by NREL. Although critical review is yet to be done, several draft data modules are now available through the NREL website: http://www.nrel.gov/lci (see Table 1).

The data are in EcoSPOLD format [3]. The EcoSPOLD format was developed in 1997 as a joint effort of the Society for the Promotion of Lifecycle Development (SPOLD) and the various LCA database and software developers to provide a common format for the exchange of life-cycle inventory data, allowing data to be understood, compared, and exchanged, regardless of how they are stored in their original database. It also is intended to make it easier to download data for use in private software tools (it is assumed that the user already has this capability). A Users Guide for the US Database is available from the NREL website.

The vision for the US Database project is to create a central resource of life-cycle inventory so that future LCAs can be created from a common data set. As can be seen in Table 1, the data modules that have been collected so far have centered on fuel production and combustion, electricity generation, and transportation activities. Given that the project's main sponsor is NREL's Center for Buildings and Thermal Systems, the initial data sets focus on building products and materials. With the participation of the automotive industry, transformation processes relevant to automobile manufacturing, such as aluminum casting, are also included. Data are currently being collected on steel stamping and iron casting.

This is a very ambitious project, and although it has been under way for 3 years, there is much to be done before it is usable for LCA. The data sets available at this time are limited in coverage; in particular, the database does not yet include chemical processes. The effort is intended to eventually include data for many more products and materials (the prioritized list is too long to be included here) but will continue to give priority to building- and automotive-related data. A process for peer review of the data sets has not been developed or conducted on the data sets already created. Also, it is

Table 1. Currently available data modules in the NREL US database.

Anthracite Coal Production
Anthracite Combustion in Industrial Boilers
Anthracite Combustion in Utility Boilers
Bituminous Coal Production
Bituminous Combustion in Industrial Boilers
Bituminous Combustion in Utility Boilers
Cargo Plane Transportation
Crude Oil Extraction
Diesel Combustion in Industrial Equipment
Diesel-Fueled Barge Transportation
Diesel-Fueled Combination Truck Transportation
Diesel-Fueled Locomotive Transportation
Diesel-Fueled Single Unit Truck Transportation
Distillate Oil Combustion in Industrial Boilers
Distillate Oil Combustion in Utility Boilers
Electricity Generation
Electricity Generation from Biomass
Fuels and Energy Precombustion
Gasoline Combustion in Industrial Equipment
Gasoline-Fueled Combination Truck Transportation
Gasoline-Powered Single Unit Truck Transportation
Lignite Coal Combustion
Lignite Combustion in Industrial Boilers
Lignite Utility Combustion
Lost Foam Aluminum Casting
LPG Combustion in Industrial Boilers
Natural Gas Combustion in Industrial Boilers
Natural Gas Combustion in Industrial Equipment
Natural gas Combustion in Utility Boilers
Ocean Freighter—Diesel
Ocean Freighter—Residual Oil
Petroleum Refining
Precision Sand Aluminum Casting
Residual Oil Combustion in Industrial Boilers
Residual Oil Combustion in Utility Boilers
Residual Oil-Fueled Barge Transportation
Softwood Lumber, US Southeast at mill
SPM Aluminum Casting
Uranium Fuel Production
Wood Combustion

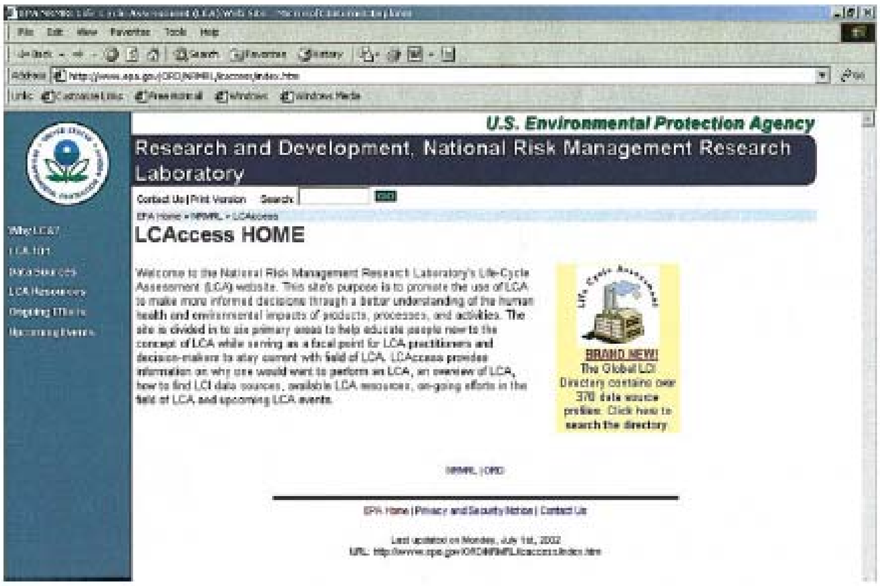


Figure 3. *LCAccess* Homepage. [Color figure can be viewed in the online issue, which is available at www.interscience.wiley.com]

not clear how NREL plans to address the issue of maintenance and upkeep of the database.

The limiting factor is the available support, either through direct funding or in-kind contributions of supplying data. To address this constraint, NREL and its contractors define the effort as a "work in progress" rather than a rigidly defined program [4]. If adequate support is found, this effort has a good chance of being fully successful and will significantly advance the use of LCA as an environmental management tool.

EPA's *LCAccess* Website

LCAccess, shown in Figure 3, is an EPA website dedicated to the promotion of LCA to make more informed decisions through a better understanding of the human health and environmental impacts of products, processes, and activities. *LCAccess* accomplishes this goal by providing information on EPA's role in LCA, the benefits of LCA, what is LCA, and an overview of how to conduct an LCA, how to find LCI data sources (through the LCI Global Directory), available LCA resources (i.e., documents, software tools, other related links), ongoing efforts in the field of LCA (such as EPA, other U.S. efforts, international efforts), and upcoming events.

LCAccess (www.epa.gov/ORD/NRMRL/lcaccess) was mainly created to address the critical need in the LCA community of where to locate existing LCI data. The website serves as a type of data portal; it is not a database itself. It was designed to bridge the need for accessing LCI data while other sources, such as the NREL database, are being developed.

Software Tools

LCA software tools are widely available, either commercially or as freeware from government organizations. Such tools are usually, but not always, accompanied by a database. Table 2 is a list of some of the LCA tools that are currently available on the market (for a complete list see http://www.epa.gov/ORD/NRMRL/lcaccess/resources.htm#Software). These tools are usually fairly straightforward to use, although some training may be needed for the user to become adept at using them. There is usually a subscription or purchase fee associated with these products.

The numerous, underlying assumptions that were applied during data collection are not typically revealed in these prepackaged data programs. That is, it is not always completely clear how the data were modeled to create the data found within the software tools. The user must rely on the reputation of the vendor for assurance on the quality of the data and the methods used to collect them.

Public Databases

Another option is the use of publicly available databases. These databases are often government-sponsored, such as the US EPA's Toxic Release Inventory (TRI) and DOE's Greenhouse Gases, Regulated Emissions, and Energy use in Transportation (GREET) model. They are easily accessible and available at no cost, although these sources do not lend themselves easily to use in life-cycle studies because the data are reported for individual sites or facilities and not as industry averages for a country or a region, as is

Table 2. Examples of LCA software tools.

LCA tool	Developer	Location
Athena 2.2	Athena Institute	Canada
BEES 3.0	National Institute for Standards and Technology	United States
Boustead Model 5.0	Boustead Consulting Ltd.	United Kingdom
CMLCA	Institute of Environmental Science (CML)	The Netherlands
EcoScan	TNO	The Netherlands
GaBi 4	PE Europe & IKP	Germany
IVAM LCA Data 4	IVAM	The Netherlands
KCL-ECO 4	KCL	Finland
LCAiT 4	CIT Ekologik	Sweden
SimaPro 5.1	Pre Consultants	The Netherlands
TEAM 3.0	Ecobilan	France
Umberto 4.3	Institute for Environmental Informatics	Germany

Table 3. Countries creating a national database for life-cycle inventory.*

Argentina	Estonia	The Netherlands
Australia	Finland	Portugal
Austria	France	Poland
Belgium	Germany	South Africa
Brazil	India	Spain
Canada	Italy	Switzerland
Chile	Japan	Thailand
China	Korea	United Kingdom
Chinese Taiwan	Malaysia	United States
Denmark	Norway	Vietnam

From Norris and Notten [5].

Table 4. Common impact indicator categories and example units of measurement.

Impact indicator	Unit of measurement
Resources	kg scarce Resources
Water	m^3 Water
Global warming	kg CO_2 Equivalents
Ozone depletion	CFC-11 Equivalents
Acidification	kg SO_2 Equivalents
Eutrophication	kg PO_4^{3-} Equivalents
Smog formation	kg NO_x Equivalents
Human health	Human toxicity equivalents
Ecological health	Ecotoxicity equivalents
Waste	kg Waste
Land use	Equivalent hectares

needed in an LCI. Often assumptions have to be made about the data to aggregate them to represent an industry sector. Also, data are not allocated by production so additional information is needed to determine releases per product.

Several countries have initiated efforts to create public databases specifically for life-cycle inventory data. A survey reported in 2002 revealed at least 30 countries that are involved in such activities at varying levels of progress (see Table 3).

LIFE-CYCLE IMPACT ASSESSMENT MODELING

Life-cycle impact assessment (LCIA) practice seems to be converging on similar categories (see Table 4). In addition to the categories shown in the table, odor, noise, and radiation effects are found at times in LCA studies, although their occurrence is not as frequent as the ones listed.

As seen in Table 4, typical LCIA practice currently uses midpoint modeling. Midpoint refers to the placement along the stressor–impact (cause–effect) chain where the impacts are modeled. For example, the inventory output data for different greenhouse gases is modeled to indicate potential global warming (expressed in CO_2 equivalents, then added up), not the damage caused by climate change [6]. In general this definition works, but it is not applicable to all impact categories. In particular, the categories of human health and ecological health are not considered to have a common midpoint in the cause–effect chain [7]. This has led to the application of various modeling approaches to these categories.

Although modeling to the endpoint results in a more environmentally relevant and meaningful result, this level of detail would require impossibly large amounts of time, data, resources, and knowledge of how to interpret the results [8]. Analysis at a midpoint is an effective approach to LCIA, in that it reduces the complexity of modeling by minimizing the amount of forecasting and effect modeling. It also results in simplifying communication of the results with fewer categories to report.

TRACI

To promote the use of the best-available impact assessment practices, the EPA developed a software tool called TRACI (The Tool for the Reduction and Assessment of Chemical and other environmental Impacts). The methodologies underlying TRACI reflect state-of-the-art developments for LCIA in the United States [7]. TRACI facilitates the characterization of stressors that have potential human and environmental effects, including ozone depletion, global warming, acid-

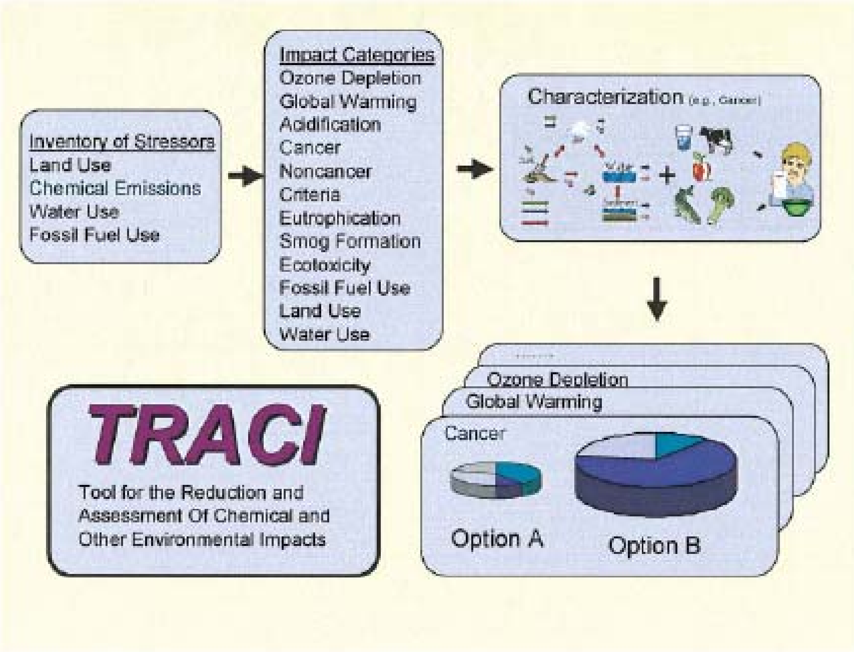


Figure 4. Framework for TRACI (Tool for the Reduction and Assessment of Chemical and other environmental Impacts). [Color figure can be viewed in the online issue, which is available at www.interscience.wiley.com]

ification, eutrophication, tropospheric ozone (smog) formation, ecotoxicity, human health criteria–related effects, human health cancer effects, human health noncancer effects, fossil fuel depletion, and land use effects (see Figure 4). In developing TRACI, impact categories were selected, available methodologies were reviewed, and categories were prioritized for further research.

Most of the impact assessment methodologies within TRACI are based on "midpoint" characterization approaches [7]. The midpoint level was chosen for various reasons, including a higher level of societal consensus concerning the certainties of modeling at this point in the cause–effect chain. Research in the impact categories of acidification, smog formation, eutrophication, land use, human cancer, human noncancer, and human criteria pollutants was conducted to construct methodologies for representing potential effects in the United States. Probabilistic analyses allowed the determination of an appropriate level of sophistication and spatial resolution necessary for impact modeling for each category, yet the tool was designed to accommodate current inconsistencies in practice (site-specific information is often not available).

The UNEP/SETAC Life-Cycle Initiative

In 2000 the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) entered into an agreement to initiate a program to define best practice in the field of Life-Cycle Assessment. Called the Life-Cycle Initiative, this cooperative effort builds on the international standards previously set by ISO. Workgroups have been established to address the topical areas of life-cycle management (LCM), life-cycle inventory (LCI) data, and life-cycle impact assessment (LCIA) (see Figure 5). Descriptions of the activities of these workgroups can be found at the UNEP website: http://www.uneptie.org/pc/sustain/lcinitiative/home.htm.

CONCLUSIONS

There truly is a great need for LCI data and effective models for assessing their related environmental impacts. Although there is currently no consensus on LCI or LCIA methodology, activities are ongoing at the international level that will lead to better LCI data and improved impact-modeling techniques. The US Database effort will benefit from increased support, either through direct financial support to assist in data collection and review or through partnering with NREL in which trade groups collect data in the preset format. It is important for the success of this effort to make appropriate groups aware of it and solicit their support. LCI and LCIA methodologies continue to be the center of discussion and debate, and will continue to evolve as researchers delve further into them.

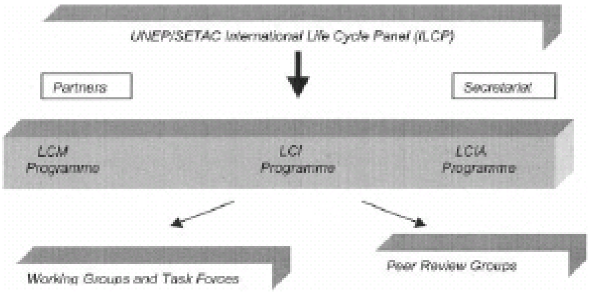


Figure 5. Structure of the UNEP/SETAC Life-Cycle Initiative.

Disclaimer

The US EPA, through its Office of Research and Development, funded and managed the research described here. It has not been subjected to Agency review and, therefore, does not necessarily reflect the views of the Agency, and no official endorsement should be inferred. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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2.6 State-of-the-Practice: Limitations in LCA methodology

The preceding sections (2.2 through 2.5) provide a quick introduction to LCA practice and methodology by describing the basic concept and the general methodological framework. While LCA is recognized as a powerful tool for environmental decision-making, it is apparent that there is not yet a consistent way to conduct LCAs. An early criticism of the methodology is that two or more studies on the same product can yield conflicting results. For example, Guinee refers to one study that favored reusable milk bottles, another that favored the milk carton, while yet a third showed scores for the two types of packaging as being similar (Guinee 1995).

Over the years, many researchers in various countries have attempted to improve the LCA methodology and to reach consensus on how to proceed in the use of the evolving LCA tools. The ISO standard (14000) series has gone a long way in establishing what the term 'life cycle assessment' means as well as in identifying the underlying issues that define the practice. However, without a prescriptive guide or standard for conducting LCAs, variations in practice continue as practitioners make individual judgments and choices on how to collect and model data. The remainder of this section focuses on the key issues and limitations related to LCA methodology. The discussion is divided among the following twelve topics:

- Goal definition
- Setting the functional unit
- Determining the scope and boundaries (cut-off rules and exclusions)
- Inventory data sources (private and public)
- Aggregating data across technologies, ages and controls
- Co-product and input material allocation
- Taxonomy of impact categories and models
- Lacking or inconsistent impact data
- Partitioning (across media and impact models)
- Modeling to the midpoint versus endpoint
- Data Quality
- Interpretation and weighting schemes (normalization, valuation, and uncertainty analysis)

In the following paragraphs, each of these parameters is reviewed.

2.6.1 Goal Definition

The first step in an LCA is to clearly establish the goal of the study. From there, the scope of the analysis can be defined and the boundaries for the study can be drawn. Once the goal is clearly defined, conducting the LCA should be consistent with the stated goal. For example, if the goal is to look at the national impact of a process, such as transportation, then data should accurately reflect the national, versus a regional, average. Currently there is no detailed guidance on how to match the goal with the study. This disconnect can be especially problematic in studies that are intended to be at the screening level, sometimes called simplified LCA, where it is not clear what level of effort should be expended (that is, when are the data 'good enough' for a screening level study?).

Closely related to goal definition is the choice between conducting an "attributional" and "consequential" study. In the October 2001 international workshop on electricity data (presented in section 2.4), participants discussed how inventory data for the electricity generation could and should be modelled. These partipants were the first to make the distinction between "attributional" versus "consequential" modeling which they defined as follows:

- Attributional LCAs attempt to answer "how are things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window?"

- Consequential LCAs attempt to answer "how will flows change in response to decisions?"

For example, an attributional LCA for energy from biomass would be based on the amount of biomass that is needed to generate a given amount of energy over a given time period, and account for all the inputs and outputs of the system in the LCI. A conseqential LCA, on the other hand, would also account for the inputs and outputs associated with the amount of biomass being used as a fuel as well as identify any possible actions that may result elsewhere, such as the need to divert the biomass from the food supply, or the need to convert land to produce the biomass. (Curran, Mann et al. 2005)

The workshop participants did not settle on the appropriateness of attributional versus consequential LCAs for any given application but instead settled on recommending the development of LCI databases that can support either type of model. However, it is easy to see that the choice can have a big impact on the LCI results.

The consequential versus attributional debate continues to spark controversy. What has been referred to as the "small Danish revolution," consequential LCA is far from being perceived as common practice in countries other than Scandinavia, where it has become the officially recognized approach. While it is believed to be at greater risk for misuse, due to the large number of assumptions that are involved, consequential LCA also has the potential to unveil 'hidden' environmental problems and opportunties for system improvement. From this controversy, a guiding principle has emerged: identify and include the processes that are most likely to be affected by the change that is being analyzed (Thrane, Riisgaard et al. 2007).

2.6.2 Setting the Functional Unit

When an LCA is used to compare two or more products, the basis of comparison must be equivalent use, i.e., each system should be defined so that an equal amount of product or equivalent service is delivered to the consumer. For example, if bar soap is to be compared to liquid soap, the logical basis for comparison is an equal number of hand washings. Equivalent use for comparative studies can often be based on volume or weight, particularly when the study compares packaging for delivery of a specific product. A beverage container study might consider 1,000 liters of beverage as an equivalent use basis for comparison, because the product may be delivered to the consumer in a variety of container sizes having different life-cycle characteristics.

The definition of the functional unit is a critical step in LCA because it determines the reference flows and dictates the upstream and downstream process alternatives that are to be included in the study. Although the concept of a functional unit is a straightforward one, clear guidance for how to specify a system's functional unit has not been developed either in the literature or in the ISO standards. The ISO standards only require that the functional unit is dependent on the goal and scope of the study and that it is clearly defined and measurable (Cooper 2003).

2.6.3 Determining the Scope and Boundaries

2.6.3.1 Cut-Off Rules

At times, the system can be limited by including only processes that make relevant contributions to some environmental input or output. The further consideration of processes that are inputs to other processes should be stopped at the point where their contribution is no longer considered to be significant. Determining significance, however, is not always a straightforward process. Some inputs may be easily omitted without much loss of value, such as the impacts of producing a wrench that is used to maintain a farm tractor. In other situtations, one must quantify the input or output in question in order to determine if such data should be included. Furthermore, substances, such as dioxin, may be released in small quantities but carry such large potential risks to human and ecological health that they must be included at all levels.

Resource constraints for data collection may be a consideration in defining the system, although, in no case should the scientific basis of the study be compromised. The level of detail required to perform a thorough inventory depends on the size of the system and the purpose of the study. In a large system encompassing several industries, certain details may not be significant contributors given the defined intent of the study. These details may be omitted without affecting the accuracy or application of the results. However, if the study has a very specific focus, such as a manufacturer comparing alternative processes or materials for inks used in packaging, it would be important to include chemicals used in very small amounts.

2.6.3.2 Capital Equipment and Infrastructure Exclusions

Capital equipment includes the buildings and machinery that are needed to produce the product. The inclusion, or exclusion, of capital equipment has been the subject of debate for many years; however, the common practice is to exclude capital equipment in an LCA. The logic is that an item such as a building or heavy equipment has such a long lifespan that its contribution to the LCA would be insignificant after being apportioned by its years in use, or by the number of products it produced.

The issue of capital equipment exclusion was revisited in the field of nanotechnology. A major concern is that nanotechnology requires large and energy-consuming capital equipment that tends to become outdated quickly due to new developments. Nano-products and nano-materials may be an example where capital equipment impacts cannot be ignored as manufacturers deal with the rapid turnover of technology (Kloepffer, Curran et al. 2007).

2.6.3.3 System Expansion

The dilemma faced by LCA practitioners when modeling an industrial process which produces more than one product, as is often the case, is often settled through the application of system expansion. In system expansion, the boundaries are expanded to include the alternative production of exported functions. That is, a necessary requirement of system expansion is the existence of an alternative way to produce a by-product. While this concept seems reasonable on the surface, it can be problematic. It is often used to 'credit' the system with avoided burdens that are offset by the alternative process. For example, corn mills produce both ethanol and corn oil; since the corn oil displaces similar products such as soybean oil, the corn ethanol system is credited with the amount of energy it would have taken to make soybean oil (Kim and Dale 2002). Not only does system expansion require more data to be collected, it also presents a problem with conveying the results of the study depending upon how the process in question was modeled. It is easy to see how the application of system expansion can have a large impact on study results.

2.6.4 Inventory Data Sources

The lack of readily available inventory data continues to be a major hurdle for LCA practice. Inventory data can be created by collecting primary data directly from the sources, such as material and product manufacturers. More often data are collected from secondary sources such as reports, publications and databases. Data are held either privately, such as in LCA practitioners' software, or in the public domain, such as government sources.

2.6.4.1 Data from Private Sources

Over the years, a number of LCA software tools have become widely available, either commercially or as freeware from government organizations. Such tools are usually, but not always, accompanied by a database. Table 2-5 lists some of the LCA tools that are, or were recently, available on the market. These tools are usually fairly simple to use, although some training may be needed before the user is adept at using them. There is usually a subscription or purchase fee associated with these products.

Table 2-4 LCA and LCI software tools, vendors, and websites

Tool	Vendor	URL
AIST-LCA (JEMAI LCA)	Japan Environmental Management Association for Industry (JEMAI)	http://www.aist.go.jp/index_en.html
BEES 4.0	NIST Building and Fire Research Laboratory	www.bfrl.nist.gov/oae/software/bees.html
Boustead Model 5.0	Boustead Consulting	http://www.boustead-consulting.co.uk/products.htm
CMLCA 4.2	Centre of Environmental Science	www.leidenuniv.nl/cml/ssp/software/cmlca/index.html
Dubo-Calc	Netherlands Ministry of Transport, Public Works and Water Management	http://www.rws.nl/rws/bwd/home/www/cgi-bin/index.cgi?site=1&doc=1785
Ecoinvent Database	Swiss Centre for Life Cycle Inventories	www.ecoinvent.ch
Eco-Quantum	IVAM	http://www.ivam.uva.nl/uk/producten/product7.htm
EDIP PC-Tool	Danish LCA Center	www.lca-center.dk
eiolca.net	Carnegie Mellon University	www.eiolca.net
EMIS	Carbotech	www.carbotech.ch
Environmental Impact Indicator	ATHENA™ Sustainable Materials Institute	www.athenaSMI.ca
EPS 2000 Design System	Assess Ecostrategy Scandinavia AB	www.assess.se/

GaBi 4	PE Europe GmbH and IKP University of Stuttgart	www.gabi-software.com/software.html
GEMIS v4.4	Öko-Institut	www.oeko.de/service/gemis/en/index.htm
GREET	DOE's Office of Transportation	www.transportation.anl.gov/software/GREET/index.html
IdeMAT 2005	Delft University of Technology	www.io.tudelft.nl/research/dfs/idemat/index.htm
KCL-ECO 4.0	KCL	http://www.kcl.fi/page.php?page_id=166
LCAIT 4.1	CIT Ekologik	www.lcait.com/01_1.html
LCAPIX	KM Limited	www.kmlmtd.com/pas/index.html
MIET 3.0	Centre of Environmental Science	www.leidenuniv.nl/cml/ssp/software/miet/index.html
REGIS	Sinum AG	www.sinum.com/htdocs/e_software_regis.shtml
SimaPro 7.1	PRé Consultants	www.pre.nl/simapro.html
SPINE@CPM	Chalmers	www.globalspine.com
TEAM™ 4.0	Ecobalance	www.ecobalance.com/uk_lcatool.php
Umberto	ifu Hamburg GmbH	www.ifu.com/en/products/umberto

(Curran and Notten 2006)

While the use of readily-available software tool makes it easier to conduct an LCA, it is not always completely clear how the data were modeled in order to create the data found within them. The numerous, underlying assumptions that were applied during data collection are not typically revealed in most pre-packaged data programs. Ultimately, the user must rely on the reputation of the vendor for assurance on the quality of the data and the methods used to collect them.

2.6.4.2 Data from Public Sources

Another option for creating life cycle inventories is the use of publicly-available databases. These databases are often government-sponsored, such as the US EPA's Toxic Release Inventory (TRI) and Australia's National Pollutant Inventory (NPI). They are easily accessible and available at no cost. But these sources do not lend themselves easily for use in most life cycle studies because the data are reported for individual sites or facilities and not as industry averages for a country or a region. Often assumptions have to be made about the data in order to aggregate them to represent an industry sector. Also, data are not allocated by production; therefore, additional information is needed in order to determine releases per product (Boguski 2000).

Several countries continue to support efforts to create public databases specifically for life cycle inventory data (listed in Table 2-6). These efforts are at different stages of development.

Table 2-5 Countries creating a national database for Life Cycle Inventory

Argentina	Chile	France	Malaysia	Spain
Australia	China	Germany	Norway	Switzerland
Austria	Chinese Taipei	India	The Netherlands	Thailand
Belgium	Denmark	Italy	Portugal	United Kingdom
Brazil	Estonia	Japan	Poland	United States
Canada	Finland	Korea	South Africa	Vietnam

(Norris and Notten 2002; Curran and Notten 2006)

2.6.5 Aggregating Data across Technologies, Ages, and Controls

The level of specificity that is required in inventory data depends on the goal of the study. For some studies, especially those conducted for internal use within a company, company specific data may be used. But in other cases, the use of average or generic data is needed and is appropriate. How these data are averaged can have a big impact on the output results and tentative conclusions. For example, data can be collected to represent average operations. Or, the worst-case or best-case might be modeled. In addition, processes that make similar products may vary greatly in age or have different permitting requirements depending on their location. These variations result in different types or amounts of environmental releases.

Lacking precise guidance on how to aggregate data across an industry, the guiding principle is that the data should be collected from a reasonable and representative number of plants (U.S. Environmental Protection Agency 2006). In cases where industry averages are applied, current practice dictates that the limitations should be clearly noted.

2.6.6 Co-Product and Input Material Allocation

All industrial processes have multiple input streams and many generate multiple output streams. Usually only one of the outputs is of interest for the life cycle assessment study being conducted, so the analyst needs to determine how much of the energy and material requirements and the environmental releases associated with the process should be attributed, or allocated, to the production of each co-product. For example, steam turbine systems may sell both electricity and low-pressure steam as useful products. When co-products are present, the practitioner must determine how much of the burdens associated with operating and supplying the multi-output process should be allocated to each co-product. The practitioner must also decide how to allocate environmental burdens across co-products when one is a material stream that can be diverted from waste disposal by being sold for other uses. The discussion on allocation is continued in Chapters 4 and 5.

2.6.7 Taxonomy of Impact Categories and Models

The selection of impact categories, or 'areas of protection,' is a critical step in impact assessment. However, determining which impact categories are the most relevant through a pre-defined list has not been done. Therefore, no grand taxonomy of impact categories is available from which to choose. However, it is recognized that different goals and scopes require different categories, data sets, and methodologies (Barnthouse, Fava et al. 1997). There is general agreement that impacts to natural resources, the natural environment, the manmade environment, and human health are the ultimate goal, but the appropriate subdivision of these higher level categories has not been established.

While there is general consensus on how to model the global level impacts for global climate change and ozone depletion, impact models that occur at the regional level (acidification, eutrophication, etc.) or at the local level (human toxicity, ecotoxicity, etc.) vary greatly. In addition reliable models for nuisance impacts (noise and smell) and for radiation are yet to be developed.

2.6.8 Modeling to the Midpoint or Endpoint

Midpoint impact assessment models reflect the relative potency of the stressors at a common midpoint within the stressor-effect chain. Analysis at a midpoint minimizes the amount of forecasting and effect modeling incorporated into the LCIA, thereby reducing the complexity and cost of the modeling and often simplifying communication. Midpoint modeling can minimize assumptions and value choices, reflect a higher level of societal consensus, and be more comprehensive than model coverage for endpoint estimation (Bare, Norris et al. 2003). On the other hand, endpoint modeling leads to discrete, identifiable metrics which may have more relevance and meaning to the decision-maker (Barnthouse, Fava et al. 1997). That is, it is easier to act on information regarding the potential new cases of cancer caused by ozone depletion than on the increase in the impact indicator for ozone depletion which is given in CFC-11 equivalents.

Since the UNEP-EPA-CML workshop in 1999 in Brighton (Bare, Hofstetter et al. 2000), a broad consensus has grown among LCA practitioners and methodology experts that both "midpoint" approaches and "endpoint" approaches have their specific strengths, as well as weaknesses, and can be useful in decision-making.

2.6.9 Lacking or Inconsistent Impact Data

Not only is the lack of quality inventory data a problem, so is the availability of data which can be used to assess the potential impacts of the inventory data. Early attempts to understand life cycle impact assessment relied on the knowledge of experts in the risk assessment field. As could be expected, the focus was on modeling chemical releases (Barnthouse, Fava et al. 1997). While thousands of chemicals are manufactured and used around the world, environmental data are available for only a fraction of them. Data are also needed to model non-chemical stressors, such as resource depletion and nuisance impacts (noise and smell).

2.6.10 Partitioning

It has already been acknowledged that LCIAs do not attempt to directly assess the potential impact of releases, as a traditional risk assessment would do. To conduct the risk assessment of a release, environmental loading information must be subjected to fate and transport analysis to determine how the releases will be transferred to various environmental compartments (air, water, and soil). Furthermore, the exposure pathways to human and ecosystems must be clearly delineated. Since LCIAs do not directly model the fate and effects of releases, the issue of partitioning occurs in two places: partitioning of inventory data across media and across impact categories. The following 2 sub-sections further expand on these two facets of partitioning

2.6.10.1 Partitioning Releases to Media (Air, Water, and Soil)

The life cycle inventory provides information on environmental releases to all media; however, to model potential impacts, it is necessary to know if the release has an affinity to the air, to water, or to the ground. That is, if released to the ground, does it stay in the ground? Or, does the release move to nearby water sources, or evaporate to the air? Actual monitoring or modeled data may be needed in these cases. For many chemicals, the availability of data for evaluating fate and exposure remains limited. Better models and monitoring methods are necessary to develop accurate release estimates.

2.6.10.2 Partitioning across Impact Categories

The assignment of inventory data to different impact categories is referred as classification. In some cases, an environmental release contributes to only one impact category; however, other environmental releases contribute to more than one impact category. For example, nitrogen oxides (NOx) can be assigned to acidification, global warming, and stratospheric ozone depletion. In the real world, a release would result in contributing to a single impact, or possibly splitting across impacts. However, current LCIA models do not account for this partitioning and can assign the full amount of a release multiple times to more than one category, resulting in an overstatement of effects.

2.6.11 Data Quality

Data collected for an inventory should always be associated with a quality measure using data quality goals. No pre-defined list of data quality goals exists for all LCA projects. The number and nature of data quality goals necessary depends on the level of accuracy required to inform the decision-makers involved in the process.

Data quality indicators (DQIs) are benchmarks to which the collected data can be measured to determine if data quality requirements have been met. Similar to data quality goals, there is no pre-defined list of data quality indicators for all LCIs. The selection of data quality indicators depends upon which ones are most appropriate and applicable to the specific data sources being evaluated. Examples of DQIs are accuracy, precision, completeness, representativeness, consistency, and reproducibility. Although formal DQIs are strongly preferred, a description of how the data were generated can also be useful in judging quality.

2.6.12 Interpretation and Weighting Schemes

While the driving force for LCA studies has been the desire to reduce burdens on the environment by altering parts of a product system, benchmarking a product against a competitor or proving that one product is environmentally preferable to another has also been a motivator (Curran 1996). However, moving from the results of the impact assessment to a final decision requires three additional considerations: normalization, valuation, and uncertainty management.

2.6.12.1 Normalization

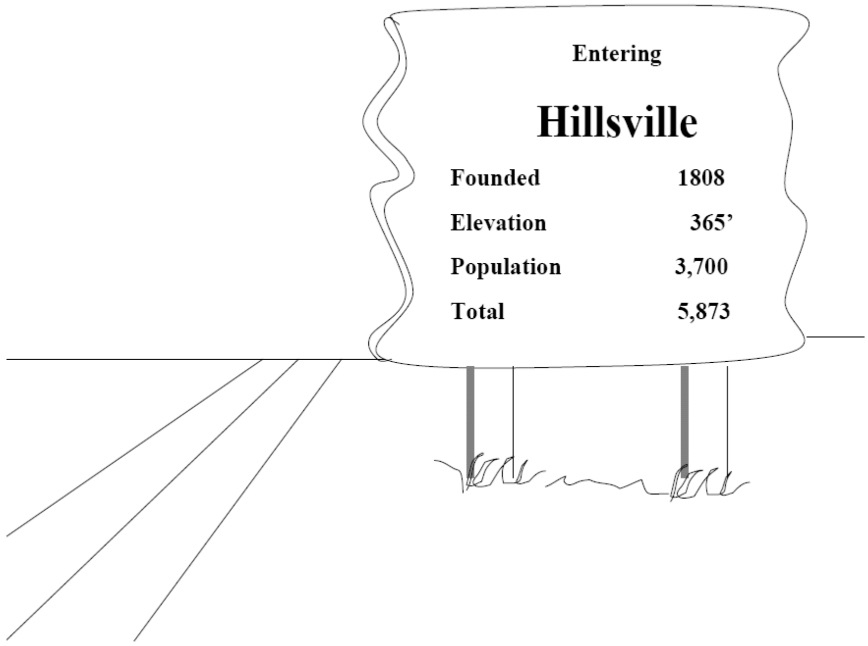
Normalization is applied in order to indicate the relative contribution of an environmental impact. Normalization calculates the magnitude of each impact compared to a reference (or normal) value by dividing the impact category by the reference, such as the total annual environmental load in a country, or the number of inhabitants. There are many different ways that normalization is being conducted worldwide, but no standardized method exists (Udo de Haes, Finnveden et al. 2002).

While normalization can be described as a science-based approach, the underlying assumptions in normalization methodology, such as the choice to use a per capita basis, is a very subjective process, blurring the line between sound science and modeling assumptions (Bare, Pennington et al. 1999). From a decision analysis point of view, the preferable outcome of the normalization step is to convert the different scales of the impact indicators into the same range. This conversion would allow all the impacts to be compacted into a single indicator. But before the numbers can be summed, a valuation step is needed.

2.6.12.2 Valuation

It is not immediately obvious how to compare impact assessment results. For example, how do global warming impacts compare to acid rain results? Studies that aim to identify a "winning" option require the application of value judgments to the LCA results. In order to address this challenge, various weighting methods and approaches have been proposed. However, no single weighting method can meet all decision-makers' needs. All methods contain a high degree of subjectivity. It is up to the individual decision-maker to reflect his or her preference in the interpretation of results.

Figure 2-6 The desire to come up with a single "score" can lead to a meaningless result



2.6.12.3 Uncertainty Analysis for Inventory and Impact Assessment

Uncertainty analysis is the process of determining the variability of the data and the impact on the final results. This variability can be attributed to either errors or fluctuations in the data. Uncertainty applies to both the inventory data and the impact assessment indicators and can have a great impact on how the results are used in decision-making. However, the actual influence of uncertainty on decision-making has not been adequately studied thus far (Geisler, Hellweg et al. 2005).

Even if a comparative LCA study cannot identify a clear "winner," this does not imply that efforts have been a waste of time. LCAs can provide decision-makers with a better understanding of the environmental and health impacts associated with each alternative, where they occur (locally, regionally, or globally), and the relative magnitude of each type of impact in comparison to each of the proposed alternatives included in the study, thus revealing the potential tradeoffs of each alternative.

2.7 Decision Points versus Methodological Assumptions

Some of the issues noted above relate to certain points within the LCA process where either the practitioner must make a decision about how to proceed (i.e. there is no single correct way to proceed) or a choice must be made regarding methodological unknowns (i.e. the correct or preferred course of action has not yet been established through general LCA guidance). Further defining the distinction between the two types of issues, a decision point does not necessarily have a right or wrong way. For example, establishing the goal of the study and the related functional unit are both essential decision points and clearly a matter of choice. But once the choice is made, the following stages of the LCA, data collection through interpretation, must be conducted in a way that is consistent with this choice. Naturally, these decision points should be clearly documented and reported along with the final results of the study.

The second type of issue occurs where variations in current practice exist and the LCA community has not yet reached consensus on best practice (with the assumption that over time consensus can and will be reached or at least a best practice will emerge). Often these variations in methodology are described as "assumptions" and are recognized to have a great impact on the study results (Bare, Pennington et al. 1999). Table 2-7 categorizes each methodological issue described in this section as a decision point or a methodological assumption.

Table 2-6 LCA includes applies various decision points and assumptions throughout the process

Decision Points	Methodological Assumptions
Goal Definition	Aggregation of Inventory Data
Functional Unit Definition	Allocation
Scope Definition	Partitioning
Data Sources	Data Quality
Impact Category Taxonomy	Impact Models
Exclusion of Capital Equipment & Infrastructure	Interpretation and Weighting

2.8 The Holy Grail: Streamlined Life Cycle Assessment

2.8.1 Introduction

Conducting an LCA can be a large and complex, therefore potentially very costly, effort. The analysis of even simple systems frequently demands data from a wide variety of industries. Finding a way to simplify the process has been the quest of LCA practitioners since the early 1990's. Because the problems regarding the availability and quality of data are at the core of all LCA studies, the main goal of simplification, or streamlining, is to reduce the amount of data that must be collected.

In practice, this goal has been accomplished in one of three ways:

1) By making reliable inventory and impact assessment more readily available to users,

2) By reducing the amount of inventory data that are needed by narrowing the boundaries of the study (i.e. omitting one or more stages or targeting pre-selected data), and

3) By limiting the impact categories that are examined.

Current limitations include the lack of site-specific data or data that are linked to specific time periods. In most cases, practitioners resort to limiting the scope (breadth and depth) of their studies in an effort to streamline. However, limiting the scope runs the risk of conducting a study that can no longer be considered an LCA. The practice of scope delimitation (2 and 3 above) is addressed further in the following sections.

2.8.2 Past Efforts that Addressed Streamlined LCA

Throughout the 1990s, several key documents were produced from efforts that addressed the streamlining issue.

• Two internationally-attended EPA workshops were held in Cincinnati, Ohio, in 1995; the results of the discussions from these workshops were published in a journal article (Curran and Young 1996) and in 1997 (U.S. Environmental Protection Agency 1997).

• Two workgroups under the auspices of SETAC created documents on streamlined LCA. The SETAC-Europe workgroup report was entitled “Simplifying LCA: Just a Cut?” (Christiansen 1997). This was followed by a report, entitled “Streamlined Life-Cycle Assessment: A Final Report from the SETAC North America Streamlined LCA Workgroup” (Todd and Curran 1999).

• In 1997, the EPA’s Office of Research and Development funded research which examined the validity of various streamlining techniques. This research mainly focused on variations of streamlining Life Cycle Inventory.

The following is a synthesis of the findings from these efforts and publications.

2.8.3 Terminology

The European LCA practitioners and researchers use the term 'Simplified LCA' more often than 'Streamlined LCA.' In addition, the SETAC-Europe Workgroup identified the following three terms which connote different levels of detail in LCA approaches:

1. *Life Cycle Thinking* is mostly a qualitative discussion to identify the stages of the life cycle and the potential environmental impacts of greatest significance e.g. for use in a design brief or in an introductory discussion of policy measures. The greatest benefit is that it helps focus consideration of the full life cycle of the product or system; data are typically qualitative (statements) or very general and available-by-heart quantitative data.

2. *Streamlined* (*Simplified) LCA* is an application of the LCA methodology for a comprehensive screening assessment, i.e. covering the whole life cycle but superficial, e.g. using generic data (qualitative and/or quantitative), standard modules for transportation or energy production, followed by a simplified assessment, i.e. focusing on the most important environmental aspects and/or potential environmental impacts and/or stages of the life cycle and/or phases of the LCA and a thorough assessment of the reliability of the results.

3. *Detailed LCA* is an application of the LCA methodology for a detailed, quantitative and mostly system-specific life cycle inventory analysis and life cycle impact assessment of all important environmental aspects of the product system, where the scope of the study is the result of an iterative procedure of data collection and scope definition. Simple cut-off allocation rules are not acceptable.

"Streamlining" is not synonymous with "screening." Screening is viewed as an initial, first-run study that may be conducted to determine if additional study is needed and, if so, where the focus of additional study should be placed, for example, the abridged LCA approach developed by Thomas Graedel (Graedel 1998). The results may be expressed qualitatively as well as quantitatively. Screening can be accomplished through a streamlined approach (Todd 1996).

In general, LCA is not viewed as one specific method but rather as a framework for an iterative process that falls along a continuum ranging from mostly qualitative to very detailed (Figure 2-7). LCA applications range from quick screening analyses which identify potentially important issues for further study, to rigorous, detailed studies that are intended to provide authoritative information for public decisions. Overly narrow studies which focus on a particular process and do not consider upstream or downstream processes, such as environmental audits or ecoprofiles (shown to the right of the arrow in Figure 2-7) are not a part of the LCA continuum.

Figure 2-7 The spectrum of LCA practice ranges from full to streamlined

Full LCA | Streamlined LCA | Environmental Audits/Ecoprofiles

Zone of Practical LCA

(Curran and Young 1996)

2.8.4 The Streamlining Process: All LCAs are Streamlined

It is widely recognized that, of necessity, all LCA studies that are conducted are streamlined. Industrial processes are so extensively inter-connected globally that complete consideration of all these interdependencies is impractical. The theoretical definition of a full, detailed LCA would describe a study that accounts for all inputs and outputs (in a balanced analysis) with an exact measure of actual environmental and human health impacts. Taken to the extreme, a full LCA could be construed as a model of the world. Of course, this level of detail is not feasible, so shortcuts are taken. It is not a question of whether or not streamlining is indeed feasible; it is simply a matter of how much streamlining is appropriate and which approaches will lead to accurate results (Curran 1996).

Research Triangle Institute conducted research for the US EPA to develop and demonstrate streamlined LCA approaches for a broad range of applications. A survey of LCA practitioners conducted by the study investigators identified eight major approaches that are used to streamline LCA:

1. Eliminating stages in the total life cycle;

2. Focusing the study on specific environmental impacts or issues from the outset;

3. Analyzing for a limited list of inventory categories;

4. Limiting or eliminating impact assessment;

5. Using qualitative information;

6. Using surrogate data from previous studies;

7. Establishing criteria as "showstoppers" or "knockouts;" and

8. Using 'threshold' levels to curtail analysis at specific points (Weitz, Todd et al. 1996).

In a following phase of the EPA study, the streamlining methods identified above were applied to baseline life cycle inventory (LCI) data for a variety of product systems (Hunt, Boguski et al. 1998). This analysis aimed to identify general guidelines that could be applied to streamlined studies that would yield results of acceptable quality. First, full LCI data were analyzed, and then each streamlining method was applied and new conclusions were drawn. The success of each streamlining method was measured in terms of how well the product system rankings based on the results of the streamlined LCIs matched those based on the results of the full LCIs.

The following four key findings were reported:

- Methods that eliminate stages and do not implement a full life cycle approach may not reach valid conclusions. The more process steps that are included, the greater the likelihood that the streamlined results will match the full analysis.

- The success of streamlining methods was generally not predictable for different products. For some, one method worked quite well, while another did not. This underscores the need for a careful analysis to determine the product system characteristics that will allow us to predict what streamlining scenarios work best.

- The use of surrogate data in place of primary data appears to be valid only if the surrogate processes very closely resemble the actual processes, or if the processes contribute very little to the LCI totals.

- The method with the most promise involves identifying parts of the system that can be eliminated or approximated because they contribute very little to the totals. That is, for those processes which contribute a large percentage of the total, the best data possible is required. For those processes that contribute very little, e.g. less than 1%, estimates or surrogates are acceptable.

Most streamlined methods gave incorrect ranking of products at least half the time or more as compared to full LCIs. The main lesson from the EPA study was that one should not be too quick in cutting materials from LCAs. The most reliable method for streamlining is to conduct approximate LCIs using generic databases.

2.8.5 Discussion

The primary message is that streamlining is an inherent part of the goal and scope definition phase and, furthermore, the goal and scope definition should not be simplified itself. LCA practitioners do not decide *whether* to streamline as much as *where and how to* streamline. The benefits of streamlining, in terms of saved resources, must be *balanced* against maintaining the utility of the results.

It is not feasible to create a one-size fits-all set of streamlining guidelines. In the LCI, streamlining mainly involves the use of readily-available data representing the product system at a general level and not at a process-specific level, or various short-cuts to such data. Studies that explicitly streamline the LCA approach should maintain certain crucial features if they are to convey valid results:

- The study should span *the entire life cycle* from cradle to grave; in a comparative study, stages may be omitted IF it is determined that their inventories are identical and that their omission will not affect the final conclusions; assertions cannot over-state the findings.

- The study should retain a *multi-media perspective* by including natural resource inputs and emissions to air, water, and land.

- The streamlining steps must be *consistent* with the original study goal, i.e. the defined reason for the study, its intended application, and anticipated uses, i.e. target audience.

- The impact assessment should be as *comprehensive* as possible, i.e. not target impact categories of interest, such as studies that assess only global warming. Qualitative data may be useful for impacts that cannot be easily quantified, such as land use issues.

- Life Cycle Thinking is an appropriate starting point, but care should be taken that actions are not limited to the application of *pre-determined notions* of environmental improvement (such as toxic chemical use reduction) without assessing the potential consequences.

- Relative differences between products or processes should be shown along with an indication of the *uncertainty* of the information.

Streamlined LCA presents a bit of a paradox in that while all LCAs are streamlined, attempts to overly streamline LCA by shortcutting the process, such as eliminating stages or releases, should be avoided. Data needs are more of a limiting factor in one's ability to streamline as opposed to methodological aspects. Predictably then, the main emphasis of streamlining LCA in past efforts has been on the reduction of the data collection effort.

To achieve this, the most effective way to simplify the LCA process is to increase the collection, publication, and standardization of LCI data. The Europeans have been successful in creating publicly-available databases through efforts such as the EcoInvent database (http://www.ecoinvent.org) and more recently the European Commission's Platform on Life Cycle Assessment (http://lca.jrc.ec.europa.eu/). The US has seen limited success in creating a national inventory database. The National Renewable Energy Laboratory created the Life Cycle Inventory Database (www.nrel.gov/lci) which now contains approximately 80 data modules. Additional funding is needed to support expansion and updating of the database.

2.8.6 Conclusions

Since a one-size-fits-all streamlined LCA has already been discounted, perhaps a method for specific industry sectors can be created. Such profiles could identify dominant stages that require closer scrutiny, or identify common inputs that may be negligible (however, negligibility cannot be based solely on mass). Such determinations could be made for the industry as a whole, decreasing the effort required in performance of subsequent LCAs. Streamlined methods only apply to specific applications of LCA results in specific systems, so that any "rules of thumb" may apply only to that sector.

Early discussions about streamlining LCA mostly involved minimizing the life cycle effort by cutting out stages or releases. Instead of this type of "whittling down" approach, LCA could be viewed as a "building up" approach by starting at the process level, for example, and then incrementally adding upstream and downstream components as needed. This approach may not appear so overwhelming the users and it would bring them along in the process as the LCA is developed.

Having sufficient reliable data continues to be a major barrier to expanding LCA applications. The NREL LCI database provides a good starting point for providing the data that are needed to complete LCAs. But more is needed. The main benefit of such a national database is that it can provide data to manufacturers for processes that are outside their boundaries. Providing common data sets, such as electricity generation, transportation, waste management, etc. that are easily accessible and contain reliable (peer-reviewed) data to potential users would be a significant boost in enabling users in both the private and public sectors to conduct LCAs, and would, in effect, streamline the LCA process.

2.9 LCA: A Powerful Tool with a Major Weakness

As discussed in Section 2.2.6 Life Cycle Inventory, an inventory analysis produces a list containing the quantities of pollutants released to the environment (after treatment or control) and the amount of energy and materials consumed. At this time, however, LCI methodology is not a straightforward, prescriptive process. Rather, it involves multiple decision points that can greatly influence the outcome of the LCI. Although it would be best to achieve consensus on the methodology, thereby reducing or eliminating variations in the practice, at this time, the best solution is to maintain transparency and to fully document how the data were calculated. That way, even if others may not agree with the approach, it is at least clear what was done.

BLANK

3 Integrating LCA and Sustainability

"When it comes to global warming, we have to look at all sides and impacts. People are just starting to realize that bio-fuels are no silver bullet. Instead, it will take a shotgun with a lot of BB's."

Dick Kempka, Ducks Unlimited

In Ecosystem Marketplace, April 2007

3.1 Chapter Overview

The immediate goal of an LCA is to account for, as precisely as possible, the natural resource inputs and environmental outputs at the boundaries of a system in order to evaluate the overall potential impacts. The larger goal of LCA is to provide broad-based, cradle-to-grave information which can be used to support decision-making by identifying which actions will move us in the direction of sustainability. For example, bio-based materials and products are often touted for their role in achieving sustainability due to their potential to address global warming and to decrease the Nation's dependency on foreign oil, and oil's polluting consequences. However, as discussed previously, there is a potential danger in looking too narrowly at "wicked problems" such as these.

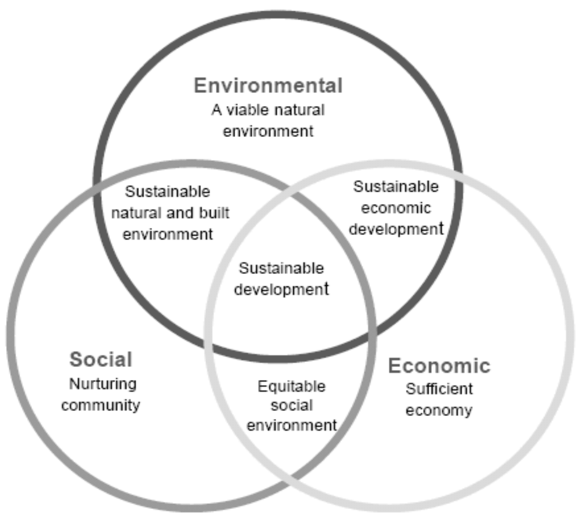
3.1.1 The Wicked Problem of Sustainability

There is a growing recognition that the challenges our planet is facing are increasingly complex and fuzzy. As environmental awareness increases, industries and businesses have responded by providing "greener" products and using "greener" processes. Many companies have found it advantageous to explore ways of moving beyond compliance by using pollution prevention/cleaner production strategies and environmental management systems to improve their environmental performance (Curran 2004). The evolution of environmental strategies over the years has moved from a single-medium, regulatory strategy to one that aims for sustainability across all media for the short and long-term. In this movement, environmental managers have begun to realize the need to look holistically at the impacts of products and processes from cradle-to-grave (or cradle-to-cradle, as it is sometimes called).

Media coverage on issues such as global warming, enhanced by the popularity of Al Gore's documentary film, "An Inconvenient Truth," has helped raise awareness among the general public of the types of far-reaching issues with which we have to deal. The term "wicked problems" has been revisited and adopted to capture these large-scale problems. The concept of the "wicked problem" was first addressed by H. Rittel and M. Webber in 1973 to describe a certain type of complex problem, i.e. one that is messy, circular, or aggressive, in contrast to "tame" problems, such as mathematics, chess, or puzzle solving (Rittel and Webber 1973). Wicked problems have incomplete, contradictory, and changing requirements, and solutions to them are often difficult to recognize because of complex interdependencies. Rittel and Webber stated that attempting to solve one wicked problem may lead to a solution that reveals or creates another, even more complex and thus, more wicked problem. Furthermore, given that there is no single "solution" to a wicked problem, the objective of the work becomes one of coherent action by enabling stakeholders to negotiate shared understanding and shared meaning about the problem and to explore possible actions that may lead to improved "solutions."

The goal of sustainability is to provide satisfactory outcomes for humans and the environment now while supporting the fulfilment of the anticipated needs of future generations. It relates to the interconnectedness of economic, social, institutional and environmental aspects of society and ecology (see Figure 3-1). While this kind of accounting assumes that all the aspects of a system can be measured and counted, it is easy to see how sustainability is a wicked problem that is not easily solvable.

Figure 3-1 The goal of sustainability is to provide satisfactory outcomes for humans and the environment while fulfilling the social and economic needs of current and future generations



3.1.2 Publications Included in this Chapter

While decision-making should consider the complex relationship between economic demands, environmental compatibility and social aspects in order to achieve a more harmonious balance, practical issues require researchers and method developers to dissect the sustainability paradigm into its individual components, forcing us to consider these ideas separately. Only after focusing on the components of sutainability and gaining a better understanding of each one can we then begin to build the global framework that is needed to integrate the disparate pieces and find sustainable solutions. Furthermore, finding and disseminating solutions to these problems is not only a matter of technology, but also of applying a multidisciplinary approach with inputs from both natural and social sciences, and from all sectors of the international community. In this way, we can promote ecogically sound solutions while keeping an eye towards social and economic development.

This chapter includes two published papers that address the environmental challenges presented by the movement towards bio-based materials:

Curran, MA (2005) "Do Bio-Based Products Move Us Toward Sustainability? A Look at Three USEPA Case Studies." *Environmental Progress* 22(4), pp277-295.

Von Blottnitz, H and Curran, M (2006) "A Review of Assessments Conducted on Bio-Ethanol as a Transportation Fuel from a Net Energy, Greenhouse Gas, and Environmental Life Cycle Perspective" *Journal of Cleaner Production*, Vol 15, pp607-619.

As can been seen in the following summaries of these papers, the studies contained therein are limited to evaluating environmental impacts and trade-offs associated with bio-based product alternatives on a life cycle basis. Although these studies focus on the environmental component of sustainability, other factors such as changes in crop production, land use conversion, loss of employment, market supply substitutions, etc., must also be considered at some point in time. Nevertheless, the studies presented here are effective examples of how life-cycle based approaches can generate information regarding the environmental component of sustainable development which can then be integrated into the overall sustainability decision-making process.

The first paper, "Do Bio-Based Products Move Us Toward Sustainability?" presents examples of various studies in which the USEPA has used a life-cycle approach to evaluate the environmental trade-offs of products with bio-based alternatives:

- Motor Oil (Virgin, Re-Refined, and Soybean-Based)
- Wall Insulation (Blown Cellulose, Fiberglass, and Mineral Wool)
- Asphalt Coating (Asphalt Cement and Gilsonite Emulsion)

- Transformer Oil (Silicone-Based, Mineral, and Soybean)
- General Purpose Cleaners (Solvent, Low-Solvent, and D-Limonene)
- Gasoline Oxygenate (Methyl Tertiary-Butyl Ether (MTBE) and Corn-Derived Ethanol)

At the time these studies were conducted, LCA practice, especially LCIA modeling, varied considerably; these examples incorporate various combinations of inventory analyses and impact assessment modeling. The paper concludes that these examples demonstrate how the move towards bio-based products is not an across-the-board "win" for the environment. In all cases, the use of fossil fuel was reported to be reduced when a bio-based alternative was used, and often, but not always, this was accompanied by a reduction in greenhouse gas emissions. Biomass feedstocks usually result in impacts in land use, water use, and eutrophication effects, resulting from the growing of the feedstocks. One exception in the alternatives listed above is the use of d-limonene, a by-product of orange production, in general purpose cleaners. The input of the orange feedstock was adjusted, i.e. allocated, based on the amount of d-limonene needed for production. This amount was approximately $1/1{,}000^{th}$ of an orange, making the impacts in orange production very small (Curran 2003).

The second paper included in this chapter, entitled "A Review of Assessments conducted on Bio-Ethanol as a Transportation Fuel from a Net Energy, Greenhouse Gas, and Environmental Life Cycle Perspective," focuses on bio-ethanol fuel and summarizes reports in the open literature. The paper highlights two recent detailed reviews which present contrasting results: one is generally unfavorable about the use of bioethanol, while the other is more favorable towards its use as replacement for fossil-derived fuel. However, the key aspect to this paper is the recognition that most work that has been done so far has not gone beyond energy and carbon assessments (von Blottnitz and Curran 2006).

The paper cites 47 published assessments that compare bio-ethanol systems to conventional fuel on a life-cycle basis. Most of these assessments focused on net energy and greenhouse gases, and despite differing assumptions and system boundaries, the following general lessons emerged: (i) make ethanol from sugar crops, in tropical countries, but approach expansion of agricultural land usage for bioethanol production with extreme caution; (ii) consider hydrolysing and fermenting lignocellulosic residues to ethanol; and (iii) the LCA results on grasses as feedstocks are insufficient to draw conclusions. It appears that technological choices in process residue handling and in fuel combustion are key, while site-specific environmental management tools should best handle biodiversity issues. Seven of the reviewed studies evaluated a wider range of environmental impacts, including resource depletion, global warming, ozone depletion, acidification, eutrophication, human and ecological health, smog formation, etc., but came up with divergent conclusions, possibly due to different approaches in scoping. These LCAs typically report that bio-ethanol results in reductions in resource use and global warming; however, impacts on acidification, human toxicity and ecological toxicity, occurring mainly during the growing and processing of biomass, were more often unfavorable than favorable. It is in this area that further work is urgently needed.

3.2 Paper IV - Do Bio-based Products Move Us Toward Sustainability?

Curran, MA (2005) "Do Bio-Based Products Move Us Toward Sustainability? A Look at Three USEPA Case Studies." *Environmental Progress* 22(4), pp277-295. DOI: 10.1002/ep.670220416.

Do Bio-Based Products Move Us Toward Sustainability? A Look at Three USEPA Case Studies

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The movement to buy "environmentally-friendly" products was recently reinvigorated by the signing of the 2002 Farm Act that requires all federal agencies to give preference to products made, in whole or significant part, from bio-based material. This paper addresses the reality behind widely held beliefs regarding "green" issues, and shows how complex it can be to choose among alternative products. Examples are presented in which the U.S. Environmental Protection Agency (USEPA) used different approaches, all based on life cycle assessment (LCA), to evaluate the environmental trade-offs of bio-based alternatives. These examples incorporate various combinations of life cycle inventory (LCI) and life cycle impact assessment (LCIA).

The first example presents results of a USEPA Office of Research & Development (ORD) project called the Framework for Responsible Environmental Decision-Making (FRED). The FRED framework uses a set of impact categories in an LCIA. Motor oil, wall insulation, and asphalt coating alternatives were all studied using FRED. The second example is a pilot study in the USEPA's Environmentally Preferable Purchasing program. Using a mix of LCI and LCIA, transformer oil and hard surface cleaner alternatives were evaluated. Finally, the paper presents preliminary LCI results of another ORD effort comparing fuel additives ethanol and MTBE. These examples demonstrate that the move to bio-based products is not an across-the-board "win" for the environment. While LCAs cannot, at this time, provide a definitive answer as to the preferability of bio-based products, it is the best tool to identify environmental trade-offs, thereby providing additional information to support decision-making.

INTRODUCTION

Government agencies, private organizations, and individuals are all being encouraged to purchase products and services that are "environmentally preferable." The main intent of such calls to "buy green" is to purchase products with reduced environmental impact. But how *does* one determine environmental preferability? Often the solution is to insist on buying products or requesting services that require fewer non-renewable resources, hence the requirement for recycled or bio-based products. Green procurement has advanced dramatically in the last two years. However, there is a potential danger in looking too narrowly at product requirements. Any comparison of products to determine preferability must assess all the relevant environmental impacts across the full product life cycle.

FEDERAL INITIATIVES FOR GOING "GREEN"

There are many government mandates in the United States for affirmative procurement, or "buying green," in order to promote the purchase of products and services that have a reduced impact on the environment. In order to meet these mandates, many programs specify single criteria for product selection, such as buying products that are made of recycled materials. Another typical criterion is based on energy expended during the use phase of the product. More recently, a growing emphasis is being placed on the procurement of products made from renewable materials, such as bio-based products. The United States Department of Agriculture (USDA) defines a bio-based product as "a commercial or industrial product, other than food or feed, that utilizes biological products or renewable domestic agricultural (plant, animal, and marine) or forestry materials."

The U.S. Farm Security and Rural Investment Act of 2002, which governs federal farm programs through 2007, was signed into law in May 2002 [1]. The Farm Act requires all federal agencies, when making procurement decisions, to give preference to products made in whole or significant part from bio-based material, unless it is unreasonable to do so based on considerations, such as availability or cost.

At the same time, the federal government recognizes that "environmentally preferable" means "products that have a lesser or reduced effect on human health and the environment when compared with competing products that serve the same purpose" [2]. So, while goals are being set to establish procurement criteria, the government acknowledges that environmental impacts are not limited to single concerns, such as depleting non-renewable resources. Other factors should also be considered, including the energy it takes to convert biomass to a usable feedstock, emissions from transportation, runoff of agrochemicals into water bodies, soil erosion from crop production, etc. In other words, we should also be looking at the entire life cycle of products and services to ensure that our actions do not result in the unintended consequence of simply shifting problems elsewhere or trading off one impact for another, possibly worse, impact.

DEFINING "GREEN" THROUGH A HOLISTIC PERSPECTIVE

Environmental Life Cycle Assessment (LCA) provides a holistic framework for identifying and evaluating environmental burdens associated with the life cycles of materials and services in a "cradle-to-grave" approach. LCA systematically identifies and evaluates opportunities for minimizing the overall environmental consequences of resource usage and environmental releases. Early research conducted by the U.S. Environmental Protection Agency (USEPA) in LCA methodology, along with efforts of the Society of Environmental Toxicology and Chemistry (SETAC), led to the four-part approach to LCA that is widely accepted today:

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

DATA VARIABILITY AND UNCERTAINTY

Before looking at the results of the case studies, some discussion on the variable and uncertain nature of the data is needed. The variability of the actual inventory data may be related to different production methods available to produce the same components or ingredients. Variability may also arise by use of inconsistent grade input materials, differences in process performance based on ambient temperature variations, scrap-rate of the process, ambient air humidity, and numerous other variables that may affect process efficiency and effectiveness. That variation may produce a spread or range of outputs in a data category for the production stage

Different data types used in a life cycle inventory have different validity. Site-specific data are collected by a practitioner at individual sites where the specific unit processes are situated and operating. Non-site-specific data come from other available sources. Surrogate data are collected from different, but reasonably similar, processes, that may be used in absence of primary data. Estimated data represent the LCA practitioner's best judgment as to what the unit operation's environmental releases may be like in reality. The different levels of data source uncertainty associated with values of different data types will affect the assurance one has in the conclusions that can be derived from any given data set.

Combining the data source uncertainty, process data variability, and production variability ranges will provide one with the overall uncertainty/variability range for the data point, which determines its overall "fuzziness." Mathematical methods, such as *error analysis*, can be used to verify that the difference in the values used to determine environmental preferability is appropriate to interpret the results of the study. This natural variability is one reason why comparison between systems should not distinguish between systems that are different by less than an order of magnitude.

LIFE CYCLE CASE STUDIES

The following sections present examples of various studies in which the USEPA has used a life cycle approach to evaluate the environmental trade-offs of bio-based alternatives. Note that since LCA is not yet a prescribed methodology, these examples incorporate various combinations of inventory analysis and impact assessment. In addition, due to the evolving nature of LCA practice, you can see variations in impact assessment methodology. The first section presents the results of a project undertaken by the Office of Research & Development (ORD) called the Framework for Responsible Environmental Decision-Making (FRED) [4]. The FRED framework uses a set of impact categories in an LCIA. The next section summarizes a pilot study performed by the USEPA's Environmentally Preferable Purchasing (EPP) program [5]. EPP uses a mix of LCI and LCIA. The last section presents the preliminary LCI results of a fuel (gasoline) additives screening study [6], also conducted by ORD.

FRAMEWORK FOR RESPONSIBLE ENVIRONMENTAL DECISION-MAKING (FRED)

The approach outlined in FRED was developed to provide guidance on how to conduct a relative comparison between product types to determine environmental preferability [4]. Motor oil, wall insulation, and asphalt coating alternatives were studied using the FRED approach.

As LCA methodology evolves, a variety of environmental impact categories and associated indicators have been used in various models, and more continue to be identified. The categories range from global impacts, such as global warming, to local impacts, such as photochemical smog. After completing a review of the most common categories, eight impact categories were selected for use in the FRED LCA system. These categories were selected based on the goals of the effort, the breadth of the project's scope, and the level of acceptance within the impact assessment community. Table 1 shows the impact indicator models used for each category, the resulting indicators, and the type of LCI data needed for each model.

Table 1. Impact categories and indicator models for the FRED LCA system [4].

Impact Category	Impact Indicator Model	Indicator	Example LCI Data Needed for Model*
Global Warming	Intergovernmental Panel on Climate Control (IPCC)	CO_2 Equivalents (kg)	Carbon Dioxide (CO_2) Nitrogen Dioxide (NO_2) Methane (CH_4) Chlorofluorocarbons (CFCs) Hydrochlorofluorocarbons (HCFCs) Methyl Bromide (CH_3Br)
Stratospheric Ozone Depletion	World Meteorological Organization (WMO)	CFC-11 Equivalents (kg)	Chlorofluorocarbons (CFCs) Hydrochlorofluorocarbons (HCFCs) Halons Methyl Bromide (CH_3Br)
Acidification	Chemical Equivalents	Acidification Potential	Sulfur Oxides (SOx) Nitrogen Oxides (NOx) Hydrochloric Acid (HCL) Hydrofluoric Acid (HF) Ammonia (NH_4)
Photochemical Smog	Empirical Kinetic Modeling Approach (EKMA)	Maximum Incremental Reactivity	Non-Methane Hydrocarbons (NMHCs)
Eutrophication	Redfield Ratio	PO_4 Equivalents (kg)	Phosphate (PO_4) Nitrogen Oxide (NO) Nitrogen Dioxide (NO_2) Nitrates Ammonia (NH_4)
Human Health	University of California -Berkeley Toxic Equivalency Potentials (TEPs)	Benzene, Toluene, TEPs	Toxic Chemicals
Ecological Health	Research Triangle Institute's LCIA Expert (Version 1)	----	Toxic Chemicals
Resource Depletion	Life Cycle Stressor Environmental Assessment (LCSEA) Model	----	Quantity of Minerals Used Quantity of Fossil fuels Used Quantity of Precious Metals

* The following is a sample of typical LCI items for each model. Other LCI items may fall under one category or another.

IPCC

The Intergovernmental Panel on Climate Control (IPCC), referred to in Table 1, is the impact indicator model for global warming. Greenhouse gas data (GHG) obtained for each LCA stage were multiplied by the relevant Global Warming Potential over a 100 year lifespan (GWP_{100}) to produce CO_2 equivalent values. As the equivalency factors are unitless values, any unit of weight can be used, as long as the unit of measurement is stated explicitly and used consistently throughout the calculation. This process is done for each GHG, with the final step being the summation of all CO_2 equivalents. The final sum, known as the Global Warming Index (GWI), indicates the product's potential contribution to global warming for each life cycle stage.

$GWI = \sum w_i \times GWP_i$, where:

w_i = weight of inventory flow i per functional unit of product

GWP_i = Global Warming Potential Equivalency Factor evaluated at 100 years

= weight of CO_2 with the same heat-trapping potential as a gram of inventory flow i

WMO

The World Meteorological Organization (WMO) is the impact indicator model for stratospheric ozone depletion. The *Montreal Protocol Handbook*, a primary guidance document on stratospheric ozone depletion, uses ozone depletion potential, expressed as CFC-11 equivalents, as the indicator of the potential for depletion to occur. The technique used for converting ODCs obtained from LCI data to CFC-11 equivalents is the same as the method demonstrated for global warming: multiply the emissions values by the equivalency factor, and add the resultant equivalencies to arrive at the product's overall potential contribution to stratospheric ozone depletion. The model established by the Montreal Protocol uses the following technique for calculating the equivalency potential (EP):

$EP = \sum w_i \times EF_i$, where:
w_i = weight of inventory flow i per functional unit of product
EF_i = ozone depletion potential equivalency factor
= weight of CFC-11 with the same potential ozone depleting effect as a gram of inventory flow i

Chemical Equivalents

Several indicators exist for acidification. The most common reference substances are hydrogen ions and sulfur dioxide. Either substance can be expressed in terms of the other. The FRED methodology used SO_2 as the reference chemical. The method for calculating the Acidification Index (AI) is similar in approach to GWI and ozone: the LCI substances (Ammonia, HCl, HF, NO, NO_2, NOx, SO_2, and SOx) are multiplied by an equivalency factor (AP) to arrive at SO_2 equivalent quantities. The SO_2 equivalents for each life cycle stage are added to calculate the Acidification Index (AI).

$AI = \sum w_i \times AP_i$, where:
w_i = weight of inventory flow i per functional unit of product
AP_i = Acidification Potential Equivalency Factor
= weight of SO_2 with the same potential acidifying effect as a unit weight of inventory flow i

EKMA

The Empirical Kinetic Modeling Approach (EKMA) is the impact indicator model for photochemical smog. The FRED LCA system used the Maximum Incremental Reactivity (MIR) approach to calculate potential smog formation. The MIR approach is based on the chemical composition of air in 39 urban areas in the U.S., which were modeled by keeping the light and VOC concentrations constant and varying the NO_2 concentration to achieve the maximal ozone production. NO_2 is a catalyst at low concentrations and an inhibitor at high concentrations. MIR values are very useful, as they are valid anywhere on the globe. However, they represent an upper bound of ozone production, and must be viewed in that light. In many northern cities, there is not enough light most of the year to produce the full amounts of ozone indicated by the MIR results. For additional information on the MIR study, see http://www.cert.ucr.edu/~carter/bycarter.htm.

Photochemical Smog Index (PSI) = $\sum w_i \times MIR_i$, where:
w_i = weight of inventory flow i per functional unit of product
MIR_i = Maximum Incremental Reactivity value for inventory flow i

Redfield Ratio

The Redfield Ratio is the impact indicator model for eutrophication, an index that essentially sums up all eutrophication precursors, expressed in the form of phosphate ion (PO_4) equivalents, by multiplying the loading of each with its related equivalency factor. These equivalencies are derived from the work of A.C. Redfield, who discovered that aquatic biomass forms with a Carbon to Nitrogen to Phosphorus (C:N:P) atomic ratio of 106:16:1.

Eutrophication Index = $\sum w_i \times EP_i$, where
w_i = weight of inventory flow i per functional unit of product
EP_i = eutrophication potential equivalency factor
= weight of PO_4 with the same potential eutrophying effect as a unit weight of inventory flow i

UCB TEPs

The University of California-Berkeley (UCB) Toxic Equivalency Potentials (TEPs) was used as the impact indicator model for human health, and is adapted from the Environmental Defense Fund (EDF) Scorecard, (http://www.scorecard.org), developed in conjunction with UC-Berkeley. This is actually a pair of indicators, one for carcinogenic and one for non-carcinogenic effects:

Human Toxicity Index = $\sum w_i \times TEP_i$, where:
w_i = weight of inventory flow i per functional unit of product
TEP_i = toxic equivalency potential
= (for carcinogens) weight of benzene with the same potential cancer-causing effect as a unit weight of inventory flow i
= (for non-carcinogens) weight of toluene with the same potential toxic effect as a unit weight of inventory flow i

RTI's LCIA Expert

Research Triangle Institute's (RTIs) LCIA Expert (Version 1) addresses ecological health. These ecological toxicity equivalency values were based on the model created for the USEPA's Streamlined LCA Model Development and Demonstration Project. The model creates an equivalency value for chemicals based on the persistence, bioaccumulation, and toxicity characteristics it exhibits in the environment. The Ecological Toxicity Index (ECOI) for the product is derived using the following equation:

$ECOI = \sum w_i \times ECO_i$
w_i = weight of inventory flow i per functional unit of product
ECO_i = ecological toxicity equivalency potential.

LCSEA

The FRED LCA system uses the Life Cycle Stressor Effects Assessment (LCSEA) model, developed by Scientific Certification Systems. It calculates the net resource depletion as a function of: 1) the material's relative rate of depletion, and 2) the relative degree of the resource's recycling.

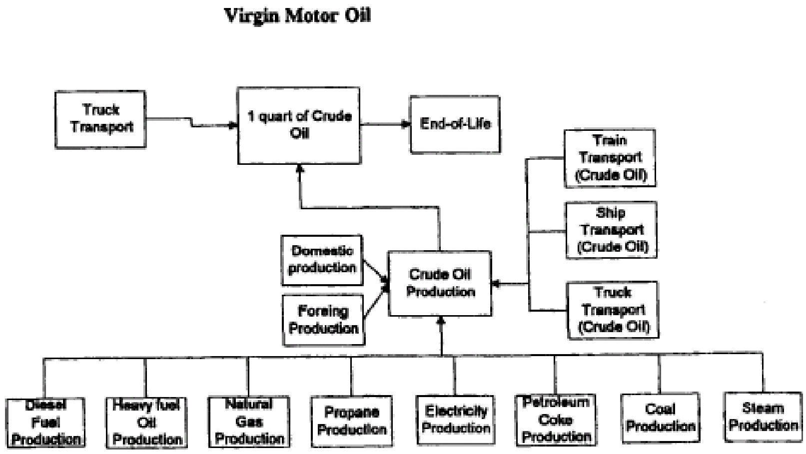

Figure 1. Flow diagram for production and use of virgin motor oil.

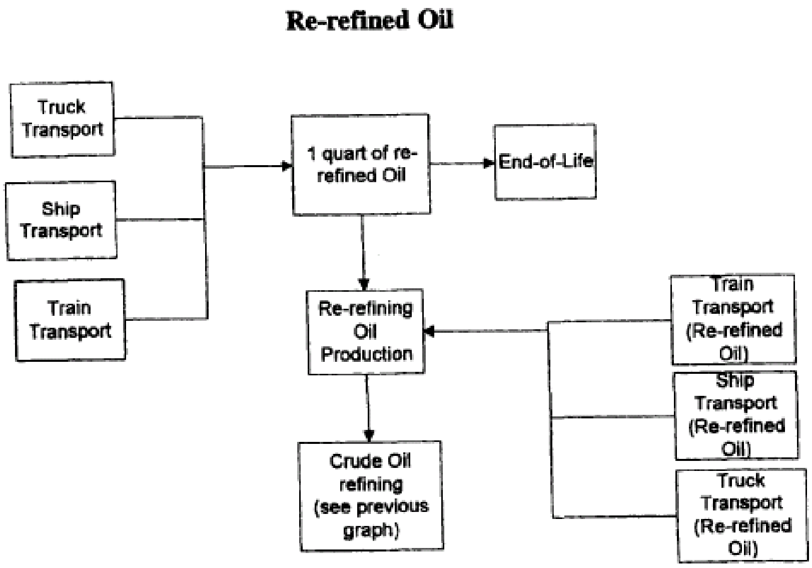

Figure 2. Flow diagram for production and use of re-refined motor oil.

Resource Depletion Indicator (RD) = $\Sigma\, w_i \times RDF_i$, where:

w_i = weight or volume of inventory flow i per functional unit of product

RDF_i = resource depletion factor

$$= \frac{(\text{Waste} - \text{Accretion}) \times T\ (\text{Total Reserve} - \text{Current Reserve})}{\text{Total Reserve} + \text{Recycling} \times T}$$

T is time in years,

Total Reserve is the known maximum extent (i.e., amount exploited over historical time plus current known, unexploited reserves).

For fossil fuel, this model uses the 50-year time horizon to project use (T = 50).

FRED CASE STUDY: MOTOR OIL

Motor oil is used to cool the engine and reduce friction. Historically, motor oil was created by extracting and refining crude oil. Due to technological advances, two alternatives to virgin oil are now commercially available: re-refined oil and "bio-based" oil (See Figure 1). Re-refined oil is essentially used oil that has undergone the refining process a second time, with additives to remove impurities (Figure 2). Bio-based oil is an all-vegetable (i.e., soybean), highly biodegradable oil that performs comparably to petroleum-based oils (Figure 3).

The function provided by the alternative products was automobile engine protection and lubrication for 3,000 miles without viscosity breakdown. The functional

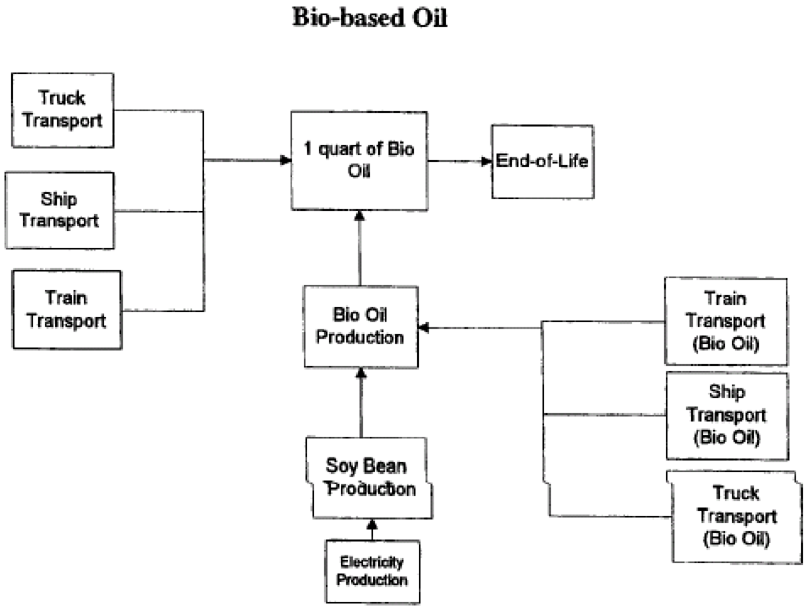

Figure 3. Flow diagram for production and use of bio-based motor oil.

unit was set at one quart, 10W30 motor oil. Data for all three products were supplied by the contractor that worked on the Buildings for Environmental and Economic Sustainability (BEES) model. BEES is a tool developed by the National Institute for Standards and Technology (NIST) to measure the environmental performance of building products by using the LCA approach specified in ISO 14000 standards, and is available online at http://www.bfrl.nist.gov/oae/software/bees.html. According to the contractor, Ecobalance, all data were from secondary sources. Virgin and refined oil data came from petroleum associations representing 90% of manufacturers. Bio-based data were derived from an average of 14 states. Upstream materials and energy use data came from national sources. All data are less than 10 years old.

Identifying Trade-offs

The LCIA results for the three oil systems are presented in Table 2. As one would expect, selecting either re-refined or bio-based oil potentially reduces fossil fuel depletion. Comparing the two alternatives to virgin oil, re-refined oil leads (as preferable) in the categories for eutrophication, photochemical smog, non-cancer, and water use, when looking at order of magnitude differences. Also, a decrease in cancer effects is indicated when moving from selecting virgin oil to either alternative product system. The differences are negligible (per the order of magnitude rule) in the other categories.

FRED CASE STUDY: WALL INSULATION

The goal of this study was to determine the feasibility of evaluating the environmental performance of four different types of wall insulation, with varying levels of thermal resistance: R-13 blown cellulose insulation, R-11 fiberglass batt insulation, R-15 fiberglass batt insulation, and R-12 blown mineral wool insulation. Blown cellulose insulation is produced primarily from post-consumer wood pulp and is treated with fire retardant. Fiberglass batt insulation is made by forming spun-glass fibers into mats of layered material (batts). Blown mineral wool insulation is made from forming fibers from either natural rock or iron ore blast furnace slag. The system studied included all unit processes for the manufacture of the insulation products, as well as the heating/cooling energy requirements associated with their use.The system function for the alternative products is to provide a constant thermal performance of 70° F for both heating and cooling of a house of 9,600 cubic feet, given a typical wood frame-residential construction, when the outside annual temperature is 55° F (average winter temperature, 32° F; average summer temperature, 85° F). The functional unit is the quantity of each insulation product required to maintain the desired thermal performance over a 50-year period. LCI data for this analysis were taken from the BEES program. According to Ecobalance, who supplied the data, it included both primary and industry average data.

Identifying Trade-offs

The results of the LCIA are presented in Table 3. Blown cellulose has a lower indicated impact in the human toxicity category, but the other products have lower indicator results for water use and solid waste. Mineral wool also has the lowest indicator result, by an order of magnitude, for ecotoxicity.

Table 2. LCIA results for motor oil [4].

Indicator	LCIA Totals		
	Virgin Oil	**Re-refined Oil**	**Bio-based Oil**
Global Warming Potential (kg CO_2 equiv)	649	332	353
Ozone Depletion Potential (kg CFC-11)	0	0	0
Acidification (kg SO_2)	5	2	2
Eutrophication (kg PO_4)	2	1	36
Photochemical Smog (kg O_3)	0.74	0.17	7.16
Human Toxicity:			
Cancer	2.12E-04	5.66E-06	9.13E-06
Non-Cancer	2.83E-02	4.29E-03	5.23E-01
Ecotoxicity	8.08E-03	4.31E-03	4.06E-02
Resource Depletion:			
Fossil (tons oil equivalent)	1.70E+00	1.05E-01	3.23E-01
Mineral (equiv tons)	0	0	0
Precious Metal, Mineral (equiv tons)	0	0	0
Other Indicators:			
Land Use (ha)	0	0	0
Water Use (kg)	1.35E-01	3.59E-03	5.89E+02
Solid Waste (kg)	8.19E-01	8.19E-01	8.55E-01

Table 3. LCIA results for wall insulation [4].

Indicator	LCIA Results			
	Blown Cellulose	**Fiberglass R-11**	**Fiberglass R-15**	**Mineral Wool**
Global Warming Potential (kg CO_2 equiv)	986	193	492	153
Ozone Depletion Potential (kg CFC-11)	0	0	0	0
Acidification (kg SO_2)	8	2	5	4
Eutrophication (kg PO_4)	0.1452	0.1169	0.1621	0.0471
Photochemical Smog (kg O_3)	1.08	2.10	6.45	7.62
Human Toxicity:				
Cancer	1.66E-05	3.09E-04	9.27E-04	2.57E-05
Non-Cancer	2.19E-03	6.52E-01	2.14E+00	5.64E-02
Ecotoxicity	1.94E-02	1.49E-02	4.69E-02	2.84E-03
Resource Depletion:				
Fossil (tons oil equivalent)	1.40	.448	.799	.742
Mineral (equiv tons)	0	0	0	0
Precious Metal, Mineral (equiv tons)	0	0	0	0
Other Indicators:				
Land Use (ha)	0	0	0	0
Water Use (kg)	1.15	.253	.473	.300
Solid Waste (kg)	1.26	.225	.377	.326

FRED CASE STUDY: ASPHALT COATING

The products evaluated represented two methods of maintaining roads: applying a thin layer (1.5 inch thick) of asphalt cement, or an asphalt emulsion containing a natural mineral product, gilsonite. Both of these products are applied to asphalt roads before significant deterioration has occurred, usually about 3 to 5 years into its life, and neither adds structural strength. Each extends the life of the road considerably. In the case of the asphalt emulsion, for 3 to 5 years, and, in the case of the asphalt cement thin layer, 7 to 9 years. There are some other specialized methods for maintaining asphalt cement roadways, but these tend to be based on trade secret chemical compositions, and were not included in this study.

Asphalt emulsion is applied by spraying diluted emulsion from a distributor truck that simultaneously spreads sand onto the emulsion. Application is at ambient temperature. A thin layer of asphalt cement is applied by first spreading a tack coat (consisting of a simple asphalt emulsion) with a distributor truck, then applying a layer of asphalt, and finally rolling the layer of asphalt to assure a smooth surface. Typically, the asphalt cement is manufactured near the construction site at a hot-mix asphalt cement plant, which heats the asphalt and mixes it with aggregate. It is then trucked to the road site and applied as above. Asphalt cement must be applied at 165° F or above. Traffic can return 1 to 2 hours after application is complete.

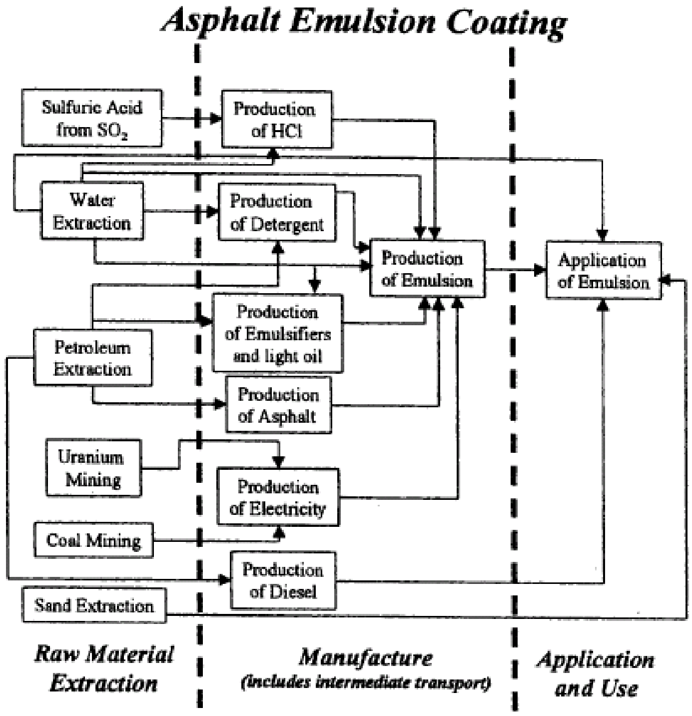

Figure 4. Asphalt emulsion coating system.

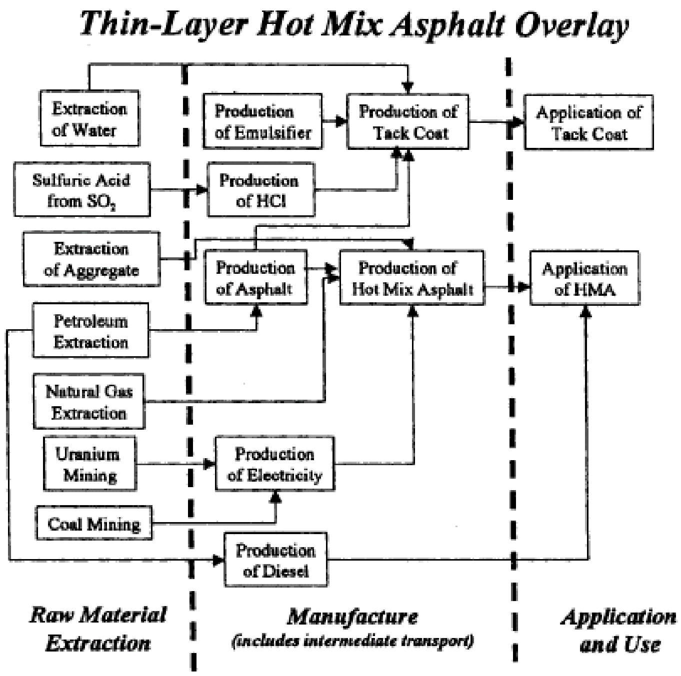

Figure 5. Thin-layer hot mix asphalt overlay system.

Table 4. LCIA results for asphalt coatings [4].

	LCIA Totals	
Indicator	**Asphalt Emulsion**	**Asphalt Cement**
Global Warming Potential (kg CO_2 equiv)	16,547	44,368
Ozone Depletion Potential (kg CFC-11)	0	0
Acidification (kg SO_2)	145	344
Eutrophication (kg PO_4)	0.0065	0.0151
Photochemical Smog (kg O_3)	36	77
Human Toxicity:		
Cancer	.0797	.178
Non-Cancer	2.02	4.51
Ecotoxicity	66,100	2,120
Resource Depletion:		
Fossil (tons oil equivalent)	38,600	85,500
Mineral (equiv. tons)	0	0
Precious Metal, Mineral (equiv. tons)	0	0
Other Indicators:		
Land Use (ha)	0.6	0.6
Water Use (kg)	76,982	2,292
Solid Waste (kg)	31,729	816,165

Table 5. Products in USEPA's Environmentally Preferable Purchasing 2002 pilot study [5].

Product Category	**Representative Products**
Transformer Oil	Mineral oil-based oil Soy-based oil Silicone fluid
General Purpose Cleaner	Solvent-based Bio-based solvent Low-solvent

The function provided by the alternative products is to maintain good quality roads (5 on a scale of 10). The functional unit is 20 years of 1-lane mile. The inventory includes two applications of the thin layer of asphalt cement, and five applications of the asphalt emulsion. Primary data were not available for the asphalt production, but were gathered from published sources. Asphalt Systems, a small manufacturer of asphalt emulsions in Utah, participated in providing site-specific information on the manufacture, application, and use of asphalt emulsions and hot-mix asphalt. Figures 4 and 5 identify the systems under study.

Identifying Trade-offs

We can make several interesting observations about the two products, based on the total indicator values noted in Table 4. Of the 14 indicators and sub-indicators evaluated, the numbers for asphalt emulsion were significantly lower than those for asphalt cement in 11 categories, equal in 2 categories (stratospheric ozone depletion and land use), and greater in one category (water use). However, given the overall uncertainty of these numbers, it is important to also look where an order of magnitude difference occurs—between the results for ecotoxicity (cement is lower), water use (cement is lower), and solid waste (emulsion is lower).

USEPA'S ENVIRONMENTALLY PREFERABLE PURCHASING (EPP) PILOT STUDY

The goal of EPP is to assist the federal government in the purchasing of environmentally preferable products and services. EPP recognizes the need to rely on approaches that minimize or eliminate environmental burdens in a comprehensive manner, and that a life cycle perspective and more defined methodologies, such as Life Cycle Assessment are tools that can be used to achieve this goal [5].

In order to investigate how life cycle-based information can be used to support purchasing decisions, EPP conducted a pilot study covering two product categories: transformer oil and general purpose cleaner (See Table 5). Each category included three alternative products, with one being a bio-based alternative. The intent of the study was to contribute to expanding the availability of life cycle-based information about products and services in a user-friendly format to assist federal purchasers in making better buying decisions from an environmental perspective.

The methodology was designed to identify environmental attributes, based on life cycle data, which government purchasers can use when ordering products acceptable to the EPP program (See Figure 6). An environmental attribute was defined for this study as an environmental characteristic for a product family that has been revealed by a weak link analysis, and

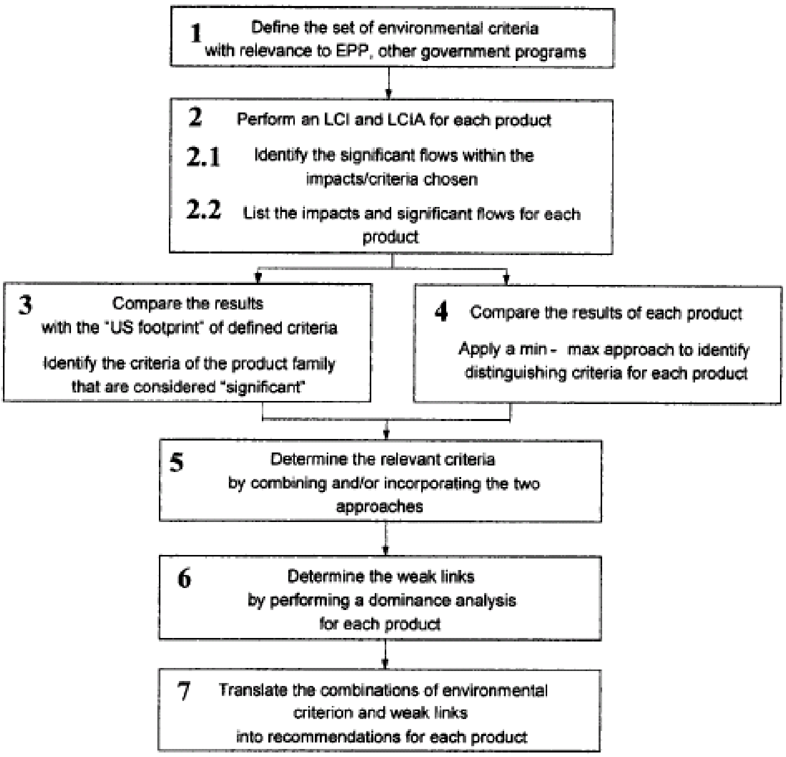

Figure 6. EPP methodology flowchart to identify acceptable products for purchase

becomes a possible criterion for a purchasing decision. A weak link is defined in this study as a process stage within a life cycle that contributes more than the other stages to a specific environmental impact. Both weak link analysis and the dominance analysis that determines the weak link are components of the methodology developed by Ecobalance for this study.

This approach aims to enhance the current EPP framework, and the credibility and reliability of purchasing decisions by:

- Taking a life cycle approach, accounting for the full upstream and downstream and often "hidden" impacts; and
- Using comprehensive data in an approach that simplifies life cycle-related information for purchasers to use.

The approach is in full coherence with the five guiding principles of the EPP methodology (see www.epa.gov/oppt/epp/finalguidance.htm and www.epa.gov/oppt/epp/eppbro.htm), especially since it:

- Examines multiple environmental criteria throughout the complete life cycle of the products;
- Identifies those environmental impacts that are most relevant to a product family. The criteria chosen takes into account the magnitude of the impacts due to the product family, as compared to total impact due to the U.S. economy, and the variance of these impacts within the family;
- Is based on quantitative, measurable, and reportable information. The whole approach is based on the ISO 14040-14043 standards and can be peer-reviewed.

Inventory data were provided by the EPP contractor, Ecobalance. Much of the data were provided through Ecobalance's database, TEAM™ (Tool for Environmental Assessment and Management). Data from other secondary sources were used to supplement the TEAM™ database, such as U.S. market information, trade associations, and the manufacturing industry, as well as government databases, such as the U.S. Bureau of Reclamation, U.S. Department of Agriculture, and various USEPA databases on environmental releases.

DEFINING THE SET OF ENVIRONMENTAL IMPACTS

EPP defined the set of environmental impacts with relevance to the program, as well as other government programs for identifying possible purchasing criteria. The environmental impacts include a set of defined impact and/or set of input and output flows that may be relevant for a product in terms of its environmental significance. Environmental impacts are identified using the following guidelines:

- Impacts relevant to the products, such as natural resources depletion for oil products);
- Availability of indicator methodologies; and/or
- Impacts monitored by the EPP program or other government programs.

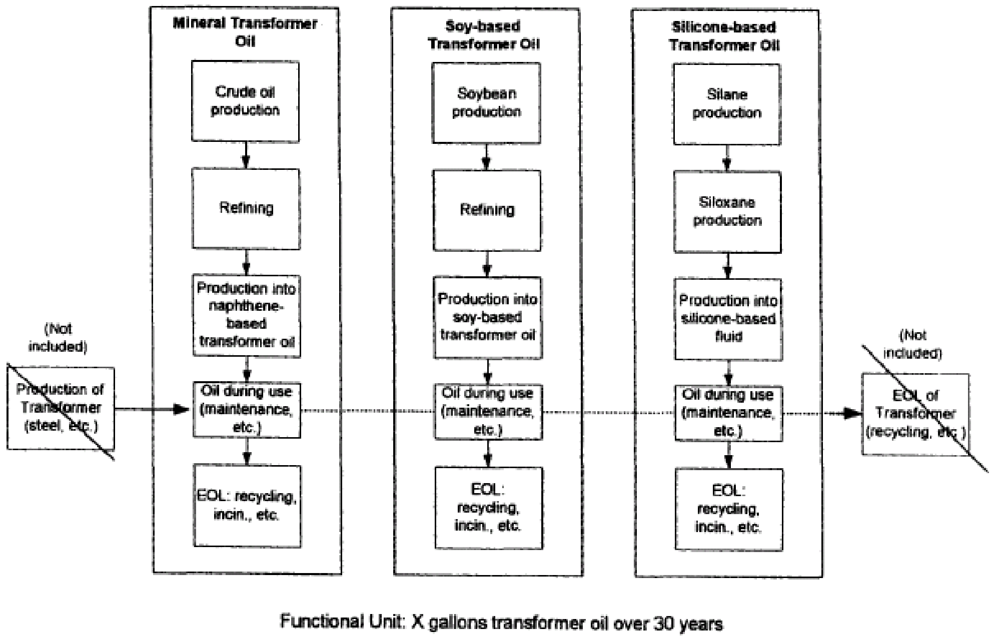

Figure 7. Flow diagram for transformer oils.

The environmental impacts used in other USEPA LCA efforts served as a starting point. These included the FRED approach, which was completed before the start of the EPP pilot study, and the TRACI (Tool for the Reduction and Assessment of Chemical Impacts) model, which was being developed at the same time. In addition, other impacts and key environmental issues (e.g., solid waste) associated with the product systems were included.

TRACI, developed by USEPA, allows potential effects to be characterized, including ozone depletion, global warming, acidification, eutrophication, tropospheric ozone (smog) formation, ecotoxicity, human particulate, human carcinogenic and non-carcinogenic effects, fossil fuel depletion, and land use effects.

To develop TRACI, impact categories were selected, available methodologies were reviewed, and categories were prioritized for further research. Consistency with previous modeling assumptions, especially those of USEPA, was important for every category. The human health cancer and non-cancer categories were heavily based on the assumptions made for the Risk Assessment Guidance for Superfund, and the USEPA's *Exposure Factors Handbook*. For categories such as acidification and smog formation, detailed empirical models, such as those developed by the U.S. National Acid Precipitation Assessment Program and the California Air Resources Board, allowed more sophisticated, location-specific characterization factors to be included. When there was no EPA precedent, assumptions and value choices were, in some cases, minimized by the use of midpoints.

Methodologies were developed specifically for the U.S. using input parameters consistent with U.S. locations for the following impact categories: acidification, smog formation, eutrophication, land use, and human cancer, non-cancer, and criteria effects. TRACI is downloadable at http://www.epa. gov/ORD/NRMRL/std/sab/iam_traci.htm.

EPP CASE STUDY: TRANSFORMER OIL

The main life cycle stages, shown in Figure 7, include:

- Production of the oils, including upstream and ancillary materials production;
- Transformer fluid transportation;
- Transformer fluid use, including maintaining the oil in the transformer; and
- Transformer fluid end-of-life, including recycling, reuse, and incineration.

The functional unit was defined as the number of gallons required to cool a 1,000 kVA transformer, about 500 gallons of fluid, over a 30-year lifetime. The assessment did not account for the production and disposal of the actual transformer. The results of the transformer oil evaluations are presented in Table 6.

N/A is listed in the land use category for mineral oil and silicone fluid because the EPP report states that adequate information was not available to quantify the land used to produce these products. The negative values (shown in parentheses) are the results of offsets due to the use of oil in other energy applications at the end-of-life.

Table 6. LCIA and LCI values for transformer oils [5].

Topics	Categories and Sub-Categories	Unit	Mineral Oil	Soybean Oil	Silicone Fluid
Land Management	Land Use	m^2	N/A	37,145	N/A
Resources	Water Intake	gal	(416)	2.5E+05	2,474
	Total Primary Energy	MJ	1.3E+05	6.6E+04	1.2E+05
	Total Primary Energy (non-renewable)	MJ	1.3E+05	9,301	1.1E+05
	Coal	kg	167	126	734
	Natural Gas	kg	229	220	1,568
	Oil	kg	2,643	(156)	41
	Uranium	kg	9.4E-04	5.6E-04	(0.001)
	Mineral Ores/ Quarry Materials	kg	18	179	772
	Limestone	kg	17	11	136
	Potash	kg		51	
	Perlite	kg	0.65	9.9E-03	289
	Phosphate Rock	kg		89	
	Sodium Chloride	kg	(0.0024)	28	346
	Use of Agrochemicals	kg		2.2	
Air Pollution	Acidification	moles H+ equiv	430	300	783
	Global Warming	kg CO_2	2,154	265	6,112
	Particulates	total DALYs	0.13	0.14	0.12
	Photochemical Smog	g NOx equiv	13	12	64
	Ozone Depletion	kg CFC-11	1.4E-05	1.2E-05	3.4E-02
	NOx	g	4,840	4,455	23,346
	SOx	g	10,263	5,944	8,321
	CO_2	g	2.0E+06	2.2E+05	4.9E+06
	CH_4	g	2,894	1,471	51,624
	PM10	g	557	618	26
	Particulates	g	1,965	894	8,583
	CFC-11	g			15
Water Pollution	Eutrophication	kg N	1,111	215,259	31
	Nitrogen as N	g	276	100,416	3.5
	Phosphate as P	g	(0)	15,763	(0)
Waste	Hazardous Waste (Land Disposal)	kg	6.1	(0.019)	1.8
	Non-Hazardous Solid Waste	kg	52	54	246
Toxicity	Human Health Non-cancer	lbs C_7H_7 equiv	6,181	5,184	55,266
	Human Health Cancer	lbs C_6H_6 equiv	5.1	4.3	16
	Ecotoxicity	lbs 2,4-D equiv	3.9	3.2	185
	Dioxins – Air Emissions	g	1.1E-06	8.9E-07	3.2E-06
	Cadmium – Air Emissions	g	7.4E-03	6.2E-03	2.3E-02
	Lead – Air Emissions	g	0.11	0.10	2.8
	Mercury – Air Emissions	g	7.1E-03	5.9E-03	0.34
	Arsenic – Air Emissions	g	0.11	0.10	0.35

Identifying Trade-offs

The soybean oil product is the largest user of land and water. However, adequate information was not available to quantify the land used to produce mineral oil or silicone fluid. Water pollution (eutrophication) is significantly higher for the soybean oil product.

Mineral oil has the fewest number of impact categories, but shows greater impact in the total primary energy, air pollution particulates (PM10), and hazardous waste (land disposal). The silicone fluid product is higher than the other two alternatives in several of the impact categories. Most notably are the higher levels for air pollution (ozone depletion and CFC-11).

EPP CASE STUDY: GENERAL PURPOSE CLEANERS

The main life cycle stages, shown in Figure 8, include:

- Production of the cleaners, including upstream and ancillary materials production and packaging;
- Transportation; and
- Use.

The functional unit was set at three cups of cleaner, with the assumption that each product cleans a hard surface, such as a kitchen countertop, with the same effectiveness. The end-of-life, i.e., cleaner in the wastewater, was not studied. The results of the general purpose cleaner evaluations are presented in Table 7.

Table 7. LCIA and LCI values for general purpose cleaners [5].

Topic	Categories and Sub-Categories	Unit	Solvent	Bio-Based Solvent	Low-Solvent
Renewable Resources	Water Intake	gal	0.28	0.32	0.38
Nonrenewable Resources	Total Primary Energy	MJ	4.1	4.9	7.6
	Total Primary Energy (non-renewable)	MJ	4.1	4.9	7.5
	Coal	kg	1.7E-02	1.8E-02	4.6E-02
	Natural Gas	kg	3.0E-02	4.3E-02	6.6E-02
	Oil	kg	4.4E-02	4.5E-02	5.8E-02
	Uranium	kg	1.7E-06	2.0E-06	2.4E-06
	Mineral Ores/Quarry Materials	kg	6.3E-03	2.7E-03	8.8E-03
	Limestone	kg	2.7E-04	6.2E-05	3.3E-03
	Sand	kg	4.5E-03	2.2E-03	4.5E-03
	Sodium Chloride	kg	1.6E-03	1.2E-04	9.2E-04
	Use of Agrochemicals	kg	-	-	-
Air Pollution	Acidification	moles H+ equiv	4.1E-02	5.1E-02	7.9E-02
	Global Warming	g CO_2 equiv	140	160	290
	Particulates	total DALYs	2.6E-02	3.1E-021	5.5E-02
	Photochemical Smog	g NOx equiv	26	23	18
	Ozone Depletion	g CFC-11 equiv	1.5E-07	2.9E-08	4.2E-07
	NOx	g	0.69	0.95	1.4
	SOx	g	0.83	1.0	1.5
	CO_2	g	137	159	281
	Hydrocarbons (unspecified)	g	26	21	15
	Particulates (unspecified)	g	0.28	0.35	0.67
Water Pollution	Eutrophication	g N equiv	0.04	0.06	0.08
Waste	Hazardous Waste (Land Disposal)	kg	2.0E-05	7.06E-06	5.5E-03
	Nonhazardous Waste (Land Disposal)	kg	2.7E-02	3.0E-02	4.7E-02
Toxicity	Human Health Non-Cancer	g C_7H_7 equiv	333		
	Human Health Cancer	g C_6H_6 equiv	0.13	0.08	0.29
	Ecotoxicity	g 2,4-D equiv	2.7	1.4	1.5
	Arsenic – Air Emissions	g	2.9E-06	6.1E-07	2.0E-05
	Dioxins – Air Emissions	g	2.6E-11	4.8E-12	6.9E-11
	Lead – Air Emissions	g	7.3E-06	7.8E-07	3.5E-05
	Mercury – Air Emissions	g	9.0E-06	3.9E-08	4.0E-06
	Mercury – Water Emissions	g	4.7E-06	3.4E-10	2.6E-06
	Phenol – Water Emissions	g	1.8E-04	1.9E-04	2.4E-04

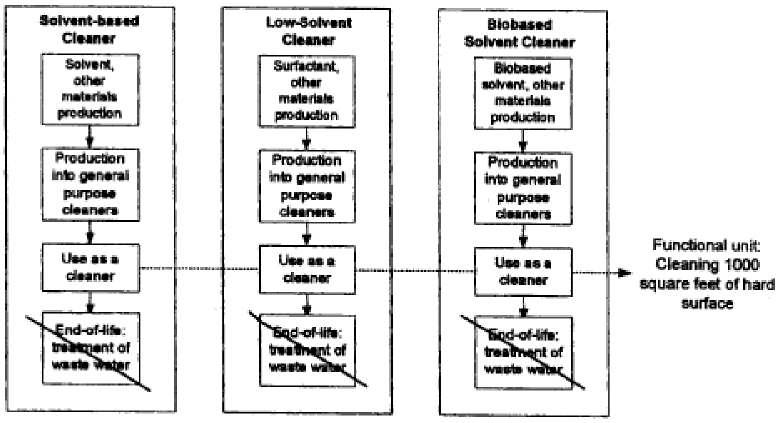

Figure 8. General purpose cleaner system boundaries.

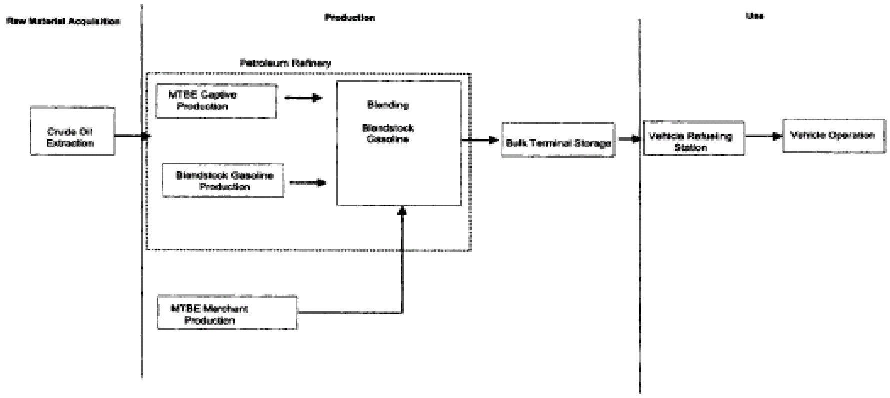

Figure 9. Life cycle stages of gasoline with MTBE.

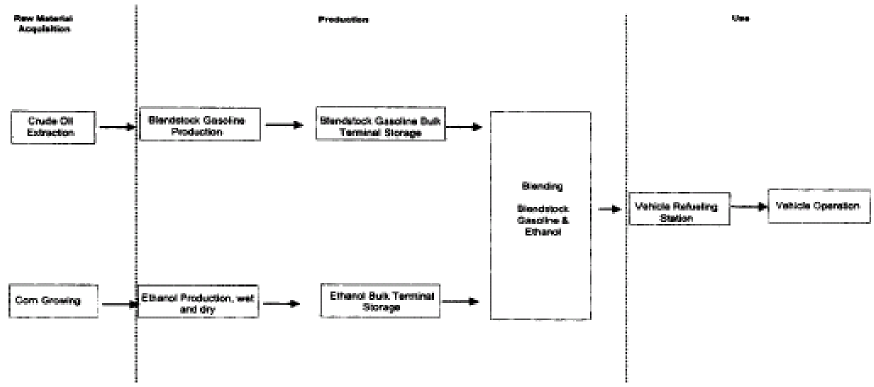

Figure 10. Life cycle stages of gasoline with ethanol.

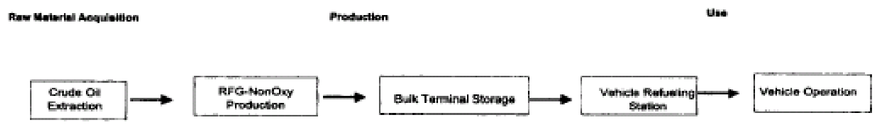

Figure 11. Life cycle stages of non-oxygenated gasoline.

Identifying Trade-offs

In looking at the results in Table 7, one can see the same types of trade-offs between these product systems as were found with the transformer oils, but with a major difference. The impacts attributed to biomass production (land use, water use, eutrophication, etc.) do not appear under the bio-based cleaner. This difference is attributed to the modeling assumptions that were used to collect the data. D-limonene, the major component in the bio-based cleaner, was considered to be a co-product of orange production. Approximately 1 pound of d-limonene is produced for every 1,000 pounds of whole oranges, but only 1/1000 of upstream orange production was allocated to generating d-limonene. Therefore, the impacts attributable to the amount of d-limonene production become negligible from a modeling standpoint.

SCREENING LCA OF FUEL ADDITIVES

The EPA's Office of Research and Development (ORD) was tasked by the Agency's Office of Air and Radiation to conduct screening LCA's of selected automotive fuel (i.e., gasoline) systems [1]. Although this project is still an on-going effort, the preliminary results of the inventory data collection phase of the project are presented.

A screening LCA is based on the application of the life cycle concept and uses a mix of qualitative and quantitative generic data. The intent of

Table 8. Impact indicators used in LCA screening of fuel additives [1].

Air Emissions (lb)
Metals in Air (lb)
Water Effluents (lb)
Metals in Water (lb)
Solid Waste (lb)
Fossil Fuel Use (Btu)
Non-Fossil Fuel Use (lb)
Water Use (gal)
Land Use (acres)
Transportation (miles)
Agrochemical Use (lb)
CO_2 Uptake (lb)

screening is to provide "directional" information regarding the environmental trade-offs between alternatives, and highlight where the more significant impact areas occur. The three fuel systems being studied include the following:

- Gasoline with methyl tertiary-butyl ether (MTBE),
- Gasoline with ethanol (EtOH), and
- A non-oxygenated reformulated gasoline.

Figures 9, 10, and 11 depict the life cycle stages included in the evaluation of the three fuel systems. The function under review is providing fuel for a generic 3,200-pound passenger automobile in the United States driving 12,000 miles over a one-year period (base year: 1999). The functional unit (i.e., number of gallons of gasoline and additive) was calculated for each system based on this function. To achieve this function, approximately 547 gallons of the fuel with MTBE are needed. For the fuel with ethanol, about 552 gallons of fuel are needed, and for the non-oxygenated fuel about 535 gallons are needed.

A screening level approach was applied to the preliminary inventory data to assess the inputs and outputs for the three systems being studied. The screening approach included the development of criteria that would determine which inventory data could be used to give some indication of potential environmental impact. For releases to the environment, the selection of these impact indicators was based on three criteria:

1. Releases to the environment and use of resources that are known to cause damage to human health or the environment;
2. Releases to the environment and use of resources suspected of causing damage to human health or the environment, and are highly visible; and
3. Releases that occur in large quantity relative to the entire system.

The first criterion is straightforward, saying if we know the cause and effect of a particular input or output, then it should be included. Mostly these are the regulated pollutants, for example, benzene. This criterion also covers reported data that cannot be traced to specific environmental releases, such as BOD and VOCs.

The second criterion follows the precautionary principle by saying that anything highly suspect, but not wholly backed by scientific evidence or agreed on by the science community, should also be listed. This criterion allows for "hot topics" that are of particular concern to whoever is conducting or interpreting the LCA. Emission of greenhouse gases is one such example.

The third criterion is based solely on quantitative measure. Any release that is large compared to the level of the other data in the inventory is reported.

In addition to these criteria for releases, the simplified approach also allows for categories of impact at a more general level. For example, instead of accounting for the emissions and effluents caused by the various modes of transporting fuels, miles traveled by ocean tanker, pipeline, barge, rail, and truck are provided. Also, since the use of pesticides, herbicides, and fertilizers is significant to growing corn in the ethanol system, but information on how much runs off into surface waters is not readily available, "agrochemical use" was added as an indicator.

Applying these decision rules to the inventory for the three gasoline systems resulted in the identification of 12 impact indicators (See Table 8) [6].

Identifying Trade-offs

Because the results of this study are still preliminary and have not undergone peer review, not all the life cycle inventory are presented in numeric terms. Instead they are discussed generally and presented as a description of how the product systems compare to each other. We, once again, applied the rule of thumb of using an order of magnitude to define differences.

Air emissions, water effluents, and solid waste: The data show no real differences (i.e., order of magnitude difference) between the systems for air emissions, water effluents, and solid waste. This lack of distinction between the three systems may be attributed to the dominance of the gasoline portion in each product (i.e., the additive is a small percentage).

Fossil Fuel and Non-Fossil Fuel Use: These indicators reflect the impact on natural resource depletion and include the amounts of fossil fuel (in Btu) and non-fossil fuel (in pounds) needed for each product system. Again, the results show no real difference.

Water Use: Approximately 25,900 gallons of water are reported for the ethanol system, the majority of it used for growing corn. Water use data was not readily found for the other systems.

Land Use: Land use is also reported only for corn growing (approximately .07 acres) but for a different reason than water use. Basing the amount of land by allocating across the products coming out of a system, such as a refinery, makes it negligible for purposes of the study.

Transportation: The total number of miles for all transportation activities within each system is very similar (14,000-18,000 miles). However, it is important to go beyond the total number of miles and investigate the transportation mode(s) assumed to be used and the associated environmental releases. For example, the ethanol system was modeled to include hauling of ethanol by truck to bulk terminal stations where blending with blendstock gasoline occurs. Likewise, the conventional systems (MTBE and non-oxy) reflect many miles in the importing of oil from overseas via ocean tanker. A closer look at the potential impact of these different types of transportation is needed.

Agrochemical Use: Agrochemicals used in corn growing include nitrogen, phosphate, and potash as fertilizers, insecticides, and pesticides. For the stated base year, 1999, the amount of these materials needed to grow .07 acres of corn was estimated to be about 37 pounds.

CO_2 Uptake: Carbon dioxide that is taken in by growing corn was approximated at 337 pounds (for the .07 acres).

These results are an interesting first step in approximating the inputs and outputs associated with the three systems. While it is apparent that the activity of growing corn sets the ethanol system apart, quantification of water use in the petroleum-based systems, runoff of agrochemicals from corn fields into waterways, and impacts from transportation is needed in order to better understand the potential impact of selecting one of these additives.

CONCLUSIONS

The USEPA's report on the EPP program states that, "LCA, and especially its most developed component, LCI, is a tool that provides quantitative and scientific analyses of environmental impacts of products and their associated industrial systems. By providing an unbiased analysis of entire industrial systems, LCA has shown that the reality behind widely-held beliefs regarding "green" issues was often more complex than expected [5]. The preceding examples of how the USEPA has applied various life cycle approaches verify this notion of complexity.

Information generated by LCAs can be used as a basis for supporting purchasing decisions by highlighting the difference in environmental impacts among choices. LCA does not, however, make the decision, although, at times, the best choice may be obvious. Other factors need to be included in the final decision-making process, including the consequences of diverting the use of land from growing crops for food to growing crops for fuel, or dedicating additional acreage to crop production. The full implementation of any LCA depends entirely on the values that the decision-maker holds, and the values, such as the importance of local impacts versus regional or global concerns, that are made when choosing among alternatives.

So, is the move to bio-based products better or not? As was seen in the preceding examples, a clear "winner" does not always appear from an LCA. Since LCA is being used to identify key environmental attributes of products and services, perhaps the best use of LCA information is to support the decision-making process and help us determine if moving toward a bio-based system is the right direction, that is, in the direction of sustainability.

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DISCLAIMER

The USEPA, through its Office of Research and Development, funded and managed the research described here. It has not been subjected to Agency review and, therefore, does not necessarily reflect the views of the Agency, and no official endorsement should be inferred. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

3.3 Paper V - A Review of Assessments Conducted on Bio-Ethanol as a Transportation Fuel from a Net Energy, Greenhouse Gas, and Environmental Life Cycle Perspective

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A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective

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Abstract

Interest in producing ethanol from biomass in an attempt to make transportation ecologically sustainable continues to grow. In recent years, a large number of assessments have been conducted to assess the environmental merit of biofuels. Two detailed reviews present contrasting results: one is generally unfavourable, whilst the other is more favourable towards fuel bio-ethanol. However, most work that has been done so far, to assess the conversion of specific feedstocks to biofuels, specifically bio-ethanol, has not gone beyond energy and carbon assessments. This study draws on 47 published assessments that compare bio-ethanol systems to conventional fuel on a life cycle basis, or using life cycle assessment (LCA). A majority of these assessments focused on net energy and greenhouse gases, and despite differing assumptions and system boundaries, the following general lessons emerge: (i) make ethanol from sugar crops, in tropical countries, but approach expansion of agricultural land usage with extreme caution; (ii) consider hydrolysing and fermenting lignocellulosic residues to ethanol; and (iii) the LCA results on grasses as feedstock are insufficient to draw conclusions. It appears that technology choices in process residue handling and in fuel combustion are key, whilst site-specific environmental management tools should best handle biodiversity issues. Seven of the reviewed studies evaluated a wider range of environmental impacts, including resource depletion, global warming, ozone depletion, acidification, eutrophication, human and ecological health, smog formation, etc., but came up with divergent conclusions, possibly due to different approaches in scoping. These LCAs typically report that bio-ethanol results in reductions in resource use and global warming; however, impacts on acidification, human toxicity and ecological toxicity, occurring mainly during the growing and processing of biomass, were more often unfavourable than favourable. It is in this area that further work is needed.
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Keywords: Bio-ethanol; Life cycle assessment; Energy balance; Greenhouse gas; Sustainable transportation

1. Introduction

Ethanol derived from biomass is often advocated as a significant contributor to possible solutions to our need for a sustainable transportation fuel. Kim and Dale [1] estimated that the potential for ethanol production is equivalent to about 32% of the total gasoline consumption worldwide, when used in E85 (85% ethanol in gasoline) for a midsize passenger vehicle. Such a substitution immediately addresses the issue of reducing our use of non-renewable resources (fossil fuels) and the attendant impacts on climate change, especially carbon dioxide and the resulting greenhouse effect, but it does not always address the notion of overall improvement. For instance, it is well understood that the conversion of biomass to bio-energy requires additional energy inputs, most often provided in some form of fossil fuel. The life cycle energy balance of a biofuel compared to conventional fossil fuel should be positive, but

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depending on the processing choices, the cumulative fossil energy demand might, at times, only be marginally lower or even higher than that of liquid fossil fuels (e.g., [2,3]). Also, ethanol in gasoline may result in decreased urban air quality, and be associated with substantive risks to water resources and biodiversity [4].

Bio-based systems have other possible ecological drawbacks. Agricultural production of biomass is relatively land intensive, and there is a risk of pollutants entering water sources from fertilisers and pesticides that are applied to the land to enhance plant growth. A very large number of researchers have recognised this conundrum and have attempted to analyse bio-ethanol systems in an effort to describe their environmental sustainability and to determine whether bio-based fuels, i.e. biofuels, are helping us to achieve the goal of providing environmentally sustainable transportation. Two recent reviews have attempted to summarise the findings. One focused on ethanol alone and presents generally unfavourable recommendations [4]; the other review looked at biofuels more generally and presented more favourable results for ethanol but cautioned with respect to some of its environmental impacts [5]. It must be noted that a number of studies that looked specifically at the North American corn-to-ethanol route were very critical as to its environmental sustainability [3,6,7].

Whilst the issue of sustainability is complicated, one that encompasses human and environmental health as well as societal needs, it is clear that our efforts to identify solutions should be broad in scope to avoid shifting problems from one place to another [8]. A large number of authors have studied liquid biofuel production systems, both current and projected, with the aim of determining whether the currently accepted premise that such systems contribute to environmental sustainability is valid. In this paper we review previous evaluations of bio-ethanol (as a transportation fuel) that used life cycle thinking or life cycle assessment as the basis for the evaluation. It is assumed that the reader has a fundamental knowledge of bio-ethanol production systems, so such background information is not provided here. The paper begins with a brief review of the study approach, then provides an overview of the evaluations that were found in a search of the open literature, and concludes with a summary of key findings and recommendations both for policy on bio-ethanol projects, and for further studies.

2. Approach used in this study

2.1. Objective

The objective of the study was to review recent evaluations of bio-ethanol, made from varying feedstocks for use as a transportation fuel, compared to conventional fuels on a life cycle basis. The effort consisted of a literature search and a desk study, followed by an analysis of the methods and assumptions used, and findings obtained to detect if any trends could be identified in the results when viewed by the type or location of the feedstock.

2.2. Scope of the search

An online search of publicly available papers and reports was conducted to find studies that have been published in recent years (1996–2004). The focus of the search was on ethanol from biomass for use as a transportation fuel (a gasoline replacement). The search included completed, published assessments that claimed to be life cycle based and that were environmental in nature. Cost analyses were not part of the main focus of the study. Only those reports that are available in English were subjected to further analysis; 47 reports were included in the analysis.

This area of research is still of significant interest worldwide and studies on biofuels continue to be conducted. Although additional studies have been published since the completion of the literature search, this paper includes the assessments that were available at that time.

2.3. Defining the life cycle

Life cycle management is quickly becoming a well-known and often used approach for environmental management. A comprehensive environmental assessment of an industrial system needs to consider both upstream and downstream inputs and outputs involved in the delivery of a unit of functionality. A life cycle approach involves a cradle-to-grave assessment, where the product is followed from its primal production stage involving its raw materials, through to its end use. The diagram in Fig. 1 illustrates a generic biofuel life cycle scheme; it shows the main sub-processes, and identifies the flows of importance for describing environmental performance.

The main stages A–E can be studied in order to determine the holistic performance of the system, depending on the goals of the study. It is at this point that differences in studies that are called life cycle assessments can be seen. Some studies include cradle-to-grave boundaries but evaluate limited input or output data. Most often, studies on energy and carbon balances, as well as greenhouse gas emissions, are found in the literature. The goal of a life cycle assessment (LCA), on the other hand, is to model all potential impacts to human health and the environment across all media – air, water and solid waste (see Appendix at the end of the paper for a longer discussion on LCA). A distinction can then be made between studies that are life cycle based versus those that aim to be fuller life cycle assessments.

3. Overview of published studies

The online literature search led to a recent review study that was conducted by the Institute for Energy and Environmental Research (IFEU) with a similar objective [5]. This study analyzed and compared all international, publicly accessible publications about biofuels that are currently used for transportation (e.g., bio-diesel and bio-ethanol as well as those potential fuels like biomass-to-liquid, BTL). The literature search uncovered additional references that were not part of the IFEU review. The integration of these efforts resulted in

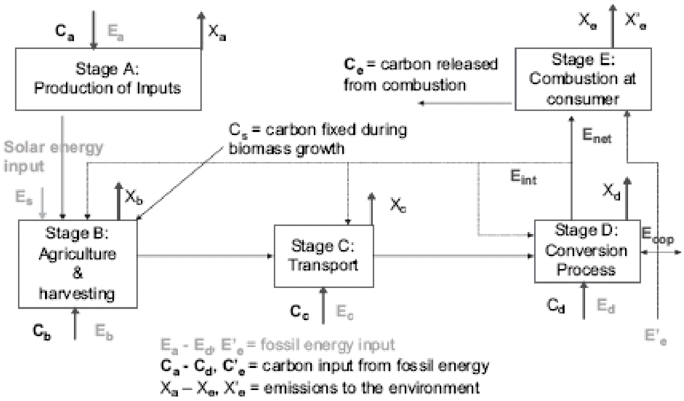

Fig. 1. Material flow and environmental interventions across the life cycle stages in a biofuel system.

47 publications, in English, that address bio-ethanol (see Table 1). Note that whilst several studies encompassed the entire life cycle as depicted in Fig. 1, many studies did not extend beyond ethanol production. It was nevertheless possible to compare studies with such differing system boundaries, at least for the carbon and energy analyses, by developing a spreadsheet to reflect all the E and C streams in Fig. 1. For those studies that exclude life cycle stages such as fuel distribution, storage and combustion (in use), it was then assumed that the carbon and energy flows associated with these stages were similar to results documented in other studies.

To date, the emphasis in life cycle based studies of bio-ethanol has strongly, but not exclusively, focused on North America and Europe, and the few full LCAs completed also do not cover the full range of possible or promising options.

4. Key results from selected bio-ethanol system assessments

Results are discussed in three categories of special interest to the question of environmental sustainability: (1) reducing dependence on fossil fuels through energy balance assessments; (2) reducing emissions of greenhouse gases (GHGs); and (3) reducing health and environmental impacts throughout the life cycle. Each interest area is discussed in more detail in the following sections.

4.1. Energy balance assessments

Almost all studies on biofuels consider the question as to whether such fuels achieve the desired net effect of lowering the amount of fossil fuel needed to propel standard and near-future vehicles powered by spark-ignition internal combustion engines. As discussed in the appended methodological discussion on energy assessments, a variety of indicators has been developed for this purpose, and it is important to norm these to the few most appropriate ones. The IFEU review [5] does this well.

Whilst this type of analysis is often inspired by the controversial results of Pimentel on ethanol from corn in the United States (e.g., [9]), the bulk of the studies report moderate to strong fossil fuel substitution effects for bio-ethanol systems. This is evident from Fig. 2 and Table 2, that present results for two of the most commonly used energy balance indicators.

4.1.1. Net replaced fossil energy

This indicator can be reported relative to the achieved transportation effect (e.g., per kilometre driven) or relative to the land area used, as is done in Fig. 2. It must be noted that no additional land is needed when by-products (e.g., molasses) or lignocellulosic residue are used as feedstock for fermentation. In this regard, for these latter feedstocks, Fig. 2 indicates the potential amounts of replaced fossil energy per hectare of land, but this is not an additional land requirement as it is for the food crops in the lowest section of the figure.

Of the possible sources of bio-ethanol, sugar crops are most land-efficient in replacing fossil energy, and here tropical sugarcane significantly outperforms sugar beet in temperate regions. Our interpretation of the Brazilian studies (at about 250 GJ/ha a) appears somewhat less conservative than that of Quirin et al. [5], which may be due to the inclusion of by-product electricity credits.

Starch crops, such as maize (corn), potatoes, wheat and rye, replace significantly less fossil energy. The IFEU study [5] reports a range of 35–50 GJ/ha a; studies which we are citing compare well with this range, at 35–40 GJ/ha a for potato and wheat [10], and a projected 27–56 GJ/ha a for wheat and rye winter crops [11]. An unusually high result is reported by Hanegraaf et al. [12], who reported 124 GJ/ha a for winter wheat, but yielding heat and power in addition to ethanol. For corn in North America, our analysis of the definitive USDA study [13] yields a fossil energy replacement of 38 GJ/ha a – a number of much debate, although its poor

Table 1
Studies of biomass-to-fuel ethanol categorised by feedstock, location and scope of the evaluation (energy/GHG or multiple criteria/LCA)

		Farmed feedstock		Waste feedstock	
		Energy/GHG	Multiple criteria/LCA	Energy/GHG	Multiple criteria/LCA
Corn	North America	Pimentel, 2003 [3] IEA, 2003 [28]; Graboski, 2002 [29] USDA, 2002 [13]; Berthiaume et al., 2001 [7] Pimentel, 2001 [9]; GM, 2001 [30] Schneider and McCarl, 2001 [31]; Levelton Engineering Ltd., 2000 [32] Wang et al., 1999 [33]		Levelton Engineering Ltd., 2000 [32] (corn stover)	Sheehan et al., 2004 [19] (corn stover)
	Europe	JRC, 2003 [34]; Jungmeier et al., 2003 [35]; Schmitz, 2003 [36]; TU München, 2003 [37]			
Wheat	North America	IEA, 2003 [28]; (S&T)2, 2003 [38]		Elsayed et al., 2003 [24] (wheat straw); Levelton Engineering Ltd., 2000 [32] (wheat straw)	
	Europe	Elsayed et al., 2003 [24]; EUCAR, CONCAWE, and JRC, 2003 [39]; JRC, 2003 [34]; Jungmeier et al., 2003 [35]; LowCVP, 2004 [40]; Schmitz 2003 [36]; TU München, 2003 [37]; Thrän and Kaltschmitt, 2004 [41]; ADEME/DIREM, 2002 [42]; CONCAWE, 2002 [43]; Rosenberger et al., 2001 [11]; Richards, 2000 [44]; Hanegraaf et al., 1998 [12]; Gover et al., 1996 [45]	Kaltschmitt et al., 1997 [10]; Steltzer, 1999 [16]; IFEU, 2002 [46]		
	Australia	CSIRO, 2001 [47]			
Potatoes	Europe	JRC, 2003 [34]; Schmitz, 2003 [36]	Kaltschmitt et al., 1997 [10]; Steltzer, 1999 [16]; IFEU, 2002 [46]		
Cassava	China	Hu et al., 2004 [18]	Hu et al., 2004 [18]		
Ligno-cellulose	Australia			CSIRO, 2001 [47] (wood)	
	North America	IEA, 2003 [28]			
	Europe	EUCAR, CONCAWE, and JRC, 2003 [39] (wood); IEA, 2003 [28] (unknown); Jungmeier et al., 2003 [35]; Schindler and Weindorf, 2003 [48]; CONCAWE, 2002 [43] (wood & grass); EST, 2002 [49] (wood); GM 2002 [50] (various); JRC, 2002 [51] (wood & grass); Fromentin, 2000 [52]; Levelton Engineering Ltd., 2000 [32] (switchgrass & hay)		GM, 2002 [50] (crop residue); EUCAR, CONCAWE, and JRC, 2003 [39] (wood & straw); Altmann, 2002 [53] (wood)	
	Philippines		Tan and Culuba, 2002 [20] (agricultural)		
Sugarcane	North America	Bastianoni and Marchettini, 1996 [54]			
	South America	Moreira, 2002 [55]; Macedo, 1998 [25]			
	India			Prakash et al., 1998 [22] (molasses)	Kadam, 2002 [14] (bagasse)
	Australia			Enerstrat, 2003 [56] (molasses); CSIRO, 2001 [47] (molasses)	
	South Africa			Theka, 2003 [57] (molasses)	

Table 1 (*continued*)

		Farmed feedstock		Waste feedstock	
		Energy/GHG	Multiple criteria/LCA	Energy/GHG	Multiple criteria/LCA
Sugar beet	Europe	Elsayed et al., 2003 [24] EUCAR, CONCAWE, and JRC, 2003 [39]; IEA, 2003 [28] JRC, 2003 [34]; Jungmeier et al., 2003 [35] Schmitz, 2003 [36]; Thrän and Kaltschmitt, 2004 [41] TU München, 2003 [37]; ADEME/DIREM, 2002 [42]; CONCAWE, 2002 [43]; GM, 2002 [50]; Altmann, 2002 [53]; Fromentin, 2000 [52]; FfE, 1999 [58]; Hanegraaf et al., 1998 [12]	Steltzer, 1999 [16]; Kaltschmitt et al., 1997 [10]; IFEU, 2002 [46]		
	Australia	CSIRO, 2001 [47]			

performance relative to the sugar crops is not doubted by any commentators.

For ethanol made from a waste product taken to carry no environmental burden, a fossil energy replacement can also be determined on a per hectare basis. However, in interpreting these, it must be remembered that this is not the additional land area needed, but rather an additional bio-energy contribution that can potentially be harnessed from land already in use. Results will differ on a case-by-case basis, depending on how efficiently wastes and by-products are already used, and how the industrial systems are configured. The two diverging results for molasses illustrate this: the Indian case yielding 30 GJ/ha a is for a distillery fully integrated into a sugar mill, where excess low pressure steam is used; whereas the South African case yielding 5 GJ/ha a is for a distillery distant from sugar mills, relying on coal and grid electricity for its energy needs.

For ethanol from lignocellulosic feedstocks, the contribution to fossil energy replacement is of a similar magnitude to that of the starch crops. Our interpretation of studies on sugarcane bagasse, corn stover and wheat straw here agrees well with the range reported by Quirin et al. [5], at 25–90 GJ/ha a. It is important to note that the three studies we refer to are all for waste lignocellulosic material. Dedicated energy cropping (e.g., of grasses) is a future possibility that needs to be considered too.

4.1.2. Energy yield ratios

The ratios relating energy output of the resultant biofuel to the fossil energy input into its production are also often used to test the sensibility of making a particular product. Table 2 summarises our analysis of key studies for a range of feedstocks and locations in this regard. Again, the tropical sugarcane-based ethanol production outperforms that from starch crops in temperate regions by a significant margin. Several commentators have questioned whether the energy yield ratio for ethanol from corn in the US is at all positive, though the balance of evidence seems to indicate it is, if only marginally [13].

In the case of molasses utilization, the two studies which we cite yield very diverging ratios – the physical differences

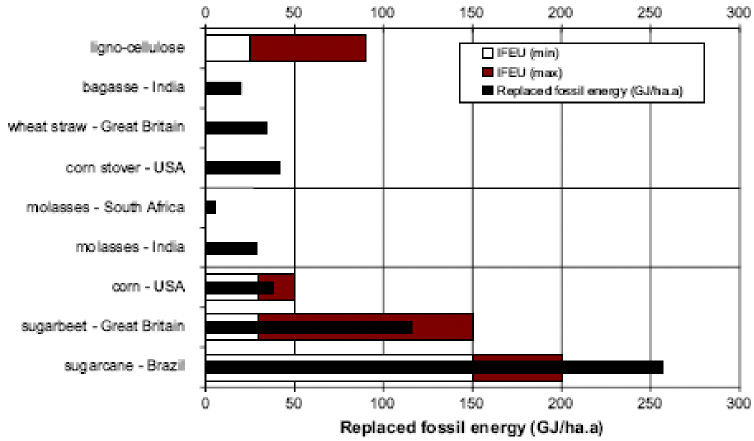

Fig. 2. Agricultural land efficiency of bio-ethanol in replacing fossil energy for transportation.

Table 2
Bio-energy yield to fossil energy input ratios for bio-ethanol systems

Feedstock and country	Energy yield ratio
Sugarcane, Brazil	7.9
Sugar beet, Great Britain	2.0
Corn, USA	1.3
Molasses, India	48
Molasses, South Africa	1.1
Corn stover, USA	5.2
Wheat straw, Great Britain	5.2
Bagasse, India	32

between the two cases have already been discussed above; in addition, it is probable that the Indian study has neglected several non-factory inputs of fossil energy into the system.

In the case of utilization of lignocellulosic wastes through hydrolysis and fermentation, the cited studies all project very encouraging bio-energy yields in relation to the required fossil energy inputs.

Concerning the effectiveness of bio-ethanol to replace fossil energy, Quirin et al. [5] have concluded that the desired effect is generally achieved, and our more limited review confirms this. It is, however, also clear that tropical sugarcane-based production is most effective from this vantage point.

4.2. Greenhouse gas assessments

With scientific evidence now increasingly mounting that climate is changing, and that this can be attributed to the large-scale use of fossil fuels, the potential of biofuels to deliver transportation energy in a carbon-neutral way is receiving increasing attention. Most studies on bio-ethanol systems have thus, also investigated at least their CO_2 balance, and often also those of the other major greenhouse gases methane and nitrous oxide. Again, a multitude of different indicators are used, and results are often not immediately comparable.

4.2.1. Avoided CO_2 equivalent emissions from bio-energy systems

Closely related to the replaced fossil energy indicator is the avoided emission of greenhouse gases (GHGs). It is dominated by CO_2 flows, but the nature of the replaced fossil fuels (coal, oil, and gas) does introduce a degree of divergence from the energy indicator, as these fuels are characterised by different fossil carbon intensities. Careful accounting for the two next important GHGs, i.e. CH_4 and N_2O, may exacerbate this variation, with global warming potentials of 21 and 310 times those of CO_2, respectively.

Again, the avoided CO_2 indicator can be derived relative to a kilometre driven, or to the land area used. Fig. 3 presents the results of our limited evaluation of avoided GHG emissions per hectare cropped and year, for the same studies as in Fig. 2, and compares our results with those by Quirin et al. [5].

Sugar-based production systems again achieve much higher effects per hectare of cropped land than starch-based systems, and tropical sugarcane is again by far the most efficient crop. Our analysis yields a much higher figure for avoided GHGs than that of Quirin et al. [5], again because of our inclusion of the substitution effect of bio-based process heat and electricity. For the other feedstocks, our interpretation of the selected studies agrees well with the more general results of Quirin et al. [5].

4.3. Health and environmental impact assessments

Only seven of the reviewed studies listed in Table 1 evaluate impacts that are more expansive in scope than the studies described in the previous sections. Whilst these studies all account for energy (as resource demand), CO_2 and greenhouse gas emissions, they go beyond these measures and include additional impact indicators. As each of these studies had a somewhat different objective and therefore, also a different scope, we have not attempted to harmonise the results as in the previous sections, but have opted to rather individually summarise each of them in the following paragraphs and in Table 3. Full citations are included in the References section.

Table 3 summarises the findings of these seven LCA studies by indicating for 13 impact categories and six related inventory categories whether the study reports an increased or decreased impact for bio-ethanol compared to conventional fuel. A dash indicates no change. In cases where only inventory data were provided, the relevant impact category was applied and interpreted as an increase, decrease or no change. As one scans across the lines of this table, it becomes evident that there is not much consensus on the environmental benefits of fuel bio-ethanol beyond the broad agreement that they do avoid to some extent the use of fossil energy carriers, and consequently also reduce GHG emissions.

Kadam (2002). Environmental benefits on a life cycle basis of using bagasse-derived ethanol as a gasoline oxygenate in India [14].

Feedstock: Bagasse
Location: India
Basis: 1 dry tonne of bagasse to produce 10% by volume ethanol in gasoline (E10).
System description: This study compares the conventional practice of burning bagasse in the field and using conventional fuel (Scenario 1) to a hypothetical process of converting bagasse into ethanol for use in E10 (Scenario 2). Boundaries include bagasse transport, ethanol production, use and excess electricity.
Impacts:
- Non-renewable resource depletion
- Greenhouse effect
- Air acidification
- Eutrophication
- Human toxicity
- Waste generation
- Air odour

Findings: The author concludes that there are significant benefits in diverting excess bagasse to ethanol production as opposed to the current practice of open-field burning.

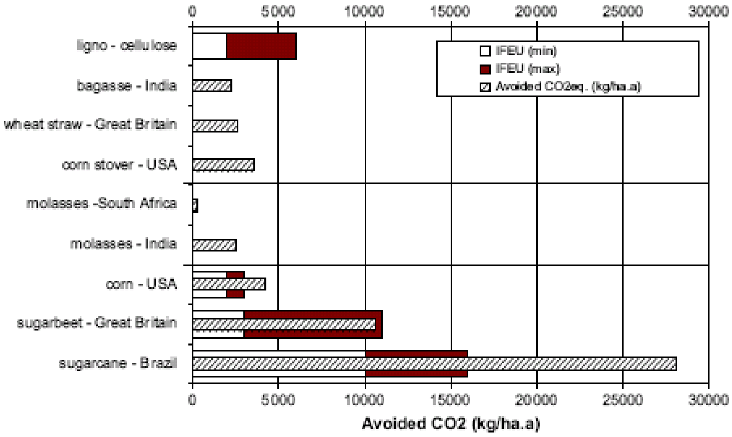

Fig. 3. Avoided GHG emissions for different bio-ethanol systems.

Scenario 2 leads to a decrease in carbon monoxide, hydrocarbons, SO_x, NO_x, particulates, carbon dioxide, methane and fossil fuel consumption. COD (from ethanol raw material production) is significantly higher. Non-methane hydrocarbons are from ethanol production. Lime, ammonia and sulphuric acid occur only in Scenario 2. Electricity credits result in negative CO_2 and CH_4 emissions and lower solid waste.

Kaltschmitt, Reinhardt & Steltzer (1997). Life cycle analysis of biofuels under different environmental aspects [10].

Feedstock: Sugar beet, wheat, and potato
Location: Germany
Basis: 1 ha
System description: This study compared bio-based systems, including cultivation and harvesting of raw materials, through energy use, to fossil systems, including mining and processing of raw materials through energy use.
Impacts:

- Finite energy
- Global warming potential (CO_2 equivalents)
- Nitrous oxide
- Acidification potential
- Sulphur dioxide
- Nitrogen oxide

Findings: The study shows some clear ecological advantages of bio-ethanol over fossil fuels, such as conserving fossil energy sources and reducing global warming potential, but bio-ethanol also has some definite disadvantages; in particular N_2O and NO_X emissions are higher. SO_2 emissions and, correspondingly, acidification potential show no discernible change.

Puppan (2002). Environmental evaluation of biofuels [15].

Feedstock: Sugar beet, winter wheat, and potato
Location: Germany
Basis: Summary of a German study on E5 fuel versus gasoline [16].
System description: Not provided
Impacts:

- Depletion of abiotic resources
- Climate change
- Stratospheric ozone depletion
- Acidification
- Human and ecotoxicity

Findings: For the bio-ethanol portion of the paper, Puppan cites a German study [16] that shows that E5 (5% ethanol) fuel has lower impacts for depletion of abiotic resources and climate change, but higher impacts for stratospheric ozone depletion (acidification and human toxicity impacts were mostly unchanged). Puppan states that the LCA study proved the environmental benefit of biofuels during the combustion in the engine, but also emphasised the environmental drawbacks that occur during the agricultural phase, such as pollution of ground and groundwater by fertilisers and pesticides as well as the creation of monocultures. Puppan concludes that it is apparent that the net environmental impact significantly depends on the agricultural conditions.

Reinhardt and Uihlein (2002). Bioethanol and ETBE (ethyl tertiary butyl ether) versus other biofuels for transportation in Europe: an ecological comparison [17].

Feedstock: Sugar beet, wheat and potato
Location: Europe
Basis: Per kilometre
System description: The study includes fertiliser, fuel, and pesticide production; cultivation; sugar extraction; ethanol production; and consumption (use in the vehicle).
Impacts:

- Resource demand (natural gas, mineral coal, brown coal, uranium ore)

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Table 3
Common life cycle impact categories and inventory releases for bio-ethanol compared to conventional fuel from a review of recent literature (1996–2004)

	Agricultural Feedstocks				Waste Feedstocks		
	Kaltschmitt 1997 [10] Sugar beet Wheat Potato	Puppan 2001[15] Sugar beet Winter wheat Potato	Reinhardt 2002 [17] Sugar beet Wheat Potato	Hu 2004 [18] Cassava	Kadam 2002 [14] Waste Bagasse	Sheehan 2004 [19] Corn Stover	Tan & Culuba 2002 [20] Agricultural Cellulosic Waste
	Germany	Germany	Europe	China	India	USA	Philippines
Resource Depletion	↓	↓	↓	↓	↓	↓	↓
Global Warming	↓	↓	↓	NA	↓	↓	↓
CO2	NA	NA	↓	↓	NA	NA	NA
Acidification	—	—	↑	NA	↓	↑	↑
SOx	↑	NA	↑	NA	NA	NA	NA
NOx	↓	NA	↑	↑	NA	NA	NA
Eutrophication	NA	NA	↑	NA	↓	NA	↑
Human Toxicity	NA	—	NA	NA	↓	NA	↑
CO	NA	NA	↑	↓	NA	NA	NA
PM	NA	NA	↑	↓	NA	NA	NA
Ecological Toxicity	NA	—	NA	NA	NA	NA	NA
Photochemical Smog	NA	NA	↓	NA	NA	↑	↓
HC	NA	NA	↓	↓	NA	NA	NA
Solid Waste	NA	NA	NA	NA	↓	NA	NA
Land Use	NA	NA	NA	NA	NA	—	NA
Water Use	NA	NA	NA	NA	—	NA	NA
Ozone Depletion	↑	↑	NA	NA	NA	↓	NA
Odour	NA	NA	NA	NA	↓	NA	NA

NA – Not Assessed ↑ – Increased impact for bio-ethanol
— – No significant change ↓ – Decreased impact for bio-ethanol

- Greenhouse gas emissions (CO_2, CH_4, and N_2O)
- Acidification
- Eutrophication
- Photochemical smog (N_2O)
- Human toxicity (reported as LCI)
- Ecotoxicity (reported as LCI)

Findings: For all life cycle comparisons, resource demand and greenhouse gas effect are in favour of biofuels, whereas most of the other parameters are in favour of the fossil fuels. Ethanol from sugar beets has advantages over wheat and potato.

Hu et al (2004). Economics, environment, and energy life cycle assessment of automobiles fueled by bio-ethanol blends in China [18].

Feedstock: Cassava
Location: China
Basis: 200,000 km driving distance
System description: Cassava, from the Guangxi Province, is converted to E85 fuel for use in a five-passenger vehicle.
Impacts: The environmental impacts are reported as inventory releases of CO_2, CO, hydrocarbons (HC), NO_x, and particulate matter (PM).
Findings: The cassava-based E85 fuel has lower life cycle CO_2, CO, HC, and PM pollutants than gasoline fuel; however, it has higher NO_x emissions. The combined environment indicator is calculated to be 20% lower for bio-ethanol.

Sheehan et al (2004). Energy and environmental aspects of using corn stover for fuel ethanol [19].

Feedstock: Corn stover
Location: USA (Iowa)
Basis: 1 ha of land and 1 km travelled using 85% ethanol in gasoline (E85) versus gasoline.
System description: Sheehan describes a hypothetical system of using corn stover to make E85. The processes include stover production and collection; transport; ethanol production; distribution; and use. The system also includes the gasoline system, with which the ethanol is blended, from crude oil extraction through use.
Impacts:
- Fossil energy use
- Greenhouse gas emissions
- Air quality (ozone precursors; CO; NO_x)
- Land use (soil health)
- Cost

Findings: Findings are presented in the paper for a few key metrics:
- Fossil energy use is 102% and greenhouse gas emissions are 113% lower for E85.
- 2.91 MJ/km avoided non-renewable energy.
- Air quality impact is mixed with emissions of CO, NO_x, and SO_x substantially higher. NO_x emissions result mainly from farm soil. SO_X emissions result from the combustion of lignin residue at ethanol plants. Hydrocarbon ozone precursors are reduced.
- Stover can be removed from the field whilst maintaining or increasing soil carbon.

Tan & Culuba (2002). Life cycle assessment of conventional and alternatives fuels for road vehicles [20].

Feedstock: Cellulosic agricultural waste using enzymatic hydrolysis and fermentation
Location: Philippines
Basis: Per kilometre
System description: The LCA encompasses extraction of raw materials and energy resources; conversion of these resources into the desired product; the utilization of the product by the consumer; and the disposal, reuse, or recycling of the product after its service life.
Impacts:
- Resource depletion (oil, coal, and natural gas)
- Human toxicity potential (PM10)
- Nutrification
- Photochemical ozone
- Acidification
- GWP (CO_2, CH_4, and N_2O)
- Air emissions (VOC, CO, NO_x, PM10, and SO_x)

Findings: For Scenario A, using Philippine Department of Energy projections for the year 2009, the use of bio-ethanol in place of gasoline is expected to yield significant gains particularly with respect to fossil fuel depletion and greenhouse gas emissions. The total impacts for bio-ethanol are significantly lower than those of gasoline, primarily due to sharp reductions in CO_2 emissions (and global warming potential) and fossil fuel consumption. Tan and Culuba state that impacts of biofuels in other impact categories remain roughly comparable to those of conventional fuels (Table 1 shows acidification, nitrification and human toxicity potentials that are slightly larger and photochemical oxidation potential slightly less than conventional fuel).

5. Findings and recommendations

Published life cycle based assessments of the sustainability of bio-ethanol systems have investigated a wide variety of feedstocks (as presented in Table 1). An array of different metrics has been used to convey their results, sometimes complicating comparisons. Methods have varied from simple energy and carbon accounting to attempts to be more inclusive in addressing sustainability. Much of the focus has been to determine if the use of biomass to make fuel is a net loss or a net gain regarding energy input versus output.

Two factors emerge as dominating the energy performance of bio-ethanol systems: crop/climate productivity, and nature of the feedstock. With regard to both of these, it is highly significant that both tropical sugar crops (by far the most productive) and cellulosic feedstocks (potentially most sustainable and abundant), have, to date, received the least amount of attention in bio-ethanol sustainability assessments that go beyond energy and carbon analysis.

The overriding conclusion of the studies that looked at energy balances was that the use of bio-ethanol in place of conventional fuels or as an additive leads to a net gain. That is, the prevailing data indicate that it takes less energy to make and distribute ethanol than can be delivered by the fuel. The results of the studies that evaluated other environmental impact categories beyond energy and greenhouse gases were mixed. Acidification, human toxicity and ecological toxicity impacts, mainly occurring during the harvesting and processing of the biomass, were more often unfavourable than favourable for bio-ethanol. The IFEU study had similar findings and concluded that for all life cycle comparisons, resource demand and GHG effect are in favour of biofuels, whereas most of the other parameters they evaluated are in favour of fossil fuels [17].

Our recommendations for future sustainability assessments of bio-ethanol are as follows:

1. It is not necessary to repeat detailed energy and GHG assessments. Depending on crop and geographical location, in many cases it will be possible to obtain a sufficiently reliable estimate from previous work (e.g., [5] or from the Biomitre website [21].
2. Studies should be selected to fill the critical gaps: full life cycle assessments are needed on ethanol from tropical sugar crops, and on 2nd generation bio-ethanol from cellulosic cropped feedstocks, such as perennial grasses or short rotation forests.
3. The assessments must be cradle-to-grave, as significant air quality impacts may be associated with the bio-ethanol used in internal combustion engines.
4. Attention must be paid to gathering the data needed for the disputed environmental categories of acidification, eutrophication, photochemical smog, human and ecotoxicity, as well as land use and its effects on biodiversity. Put another way, the safeguard subjects of human and ecological health need to feature more prominently next to those of climate change and resource depletion concerns.
5. Data gaps for life cycle assessments of corn to bio-ethanol in the United States should be addressed and filled, to address shortcomings of studies, to date, in accordance with recommendations 3 and 4.

6. Conclusion

Moving toward sustainability requires a re-thinking of our systems of production, consumption and waste management and an increased awareness of the need to avoid shifting of problems, as often occurs with isolated measures. The ecological advantages should outnumber, or outweigh, the disadvantages to the environment and human health. Numerous studies have been done in recent years evaluating the life cycle impacts of bio-ethanol, and there is now strong evidence that all bio-ethanol production is mildly to strongly beneficial from a climate protection and a fossil fuel conservation perspective. Fuel ethanol produced from sugar crops in tropical settings appears by far the most efficient in these categories from a land-use perspective. However, whilst over 40 studies have been life cycle based, only seven were identified which could be said to approach life cycle assessments. These studies do not, of course, cover the full range of possible feedstocks and geographies, and their results in the standard impact categories diverge. Further assessments should thus, take energy and carbon performances as understood, work on the less studied but highly promising feedstocks and locations outside Europe and North America, and pay more attention to the safeguard subjects of human and ecological health. We caution against basing fuel production policy on environmental sustainability studies that are life cycle based in the sense of extending from the crop to the wheel, but that ignore issues other than fossil fuel depletion and GHG emissions; such practices are likely to result in detrimental shifting of burdens.

Acknowledgements

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Appendix

Energy analysis approaches: input versus output

The energy analysis approach evaluates all the fossil fuel inputs in upstream processing steps like agriculture, transportation and processing, and these are compared against the delivered energy of the product biofuel. Referring to the terminology in Fig. 1, the net energy available from a fuel, E_e, is equal to $(E_G - E_{e1})$, where E_G is the gross energy produced by the fuel during combustion and E_{e1} is the total feedback energy in the fuel production process.

Proposed in the literature are *energy yield ratios*, e.g., the ratio of gross energy output to energy input $(E_g/E_i$, when there is no fossil energy input as in the case described by Prakash et al. [22] or $E_g/(E_i + E_b + E_c + E_d)$ in the more general case).

Similarly, a *fossil energy ratio* is proposed by Sheehan et al. [23], defined as

$$\frac{(E_{net} + E_{cop})}{(E_A + E_B + E_C + E_D)}$$

To avoid any confusion, we will here call this the bio-energy yield to fossil energy input ratio (or ByFi ratio). This relates the energy retrieved from a product biofuel, weighed against the fossil energy input involved in its life cycle, particularly in its production and conversion, and the related upstream processes. It is observed that for fossil energy ratios greater than 1, the system approaches renewability, which is theoretically only feasible for no fossil energy requirements (ratio of infinity). It might be more useful to describe this ratio as a "*bio-energy ratio*," as its value increases as the fossil energy input to the system decreases.

The use of fossil energy replaced should also be of interest – especially when comparing liquid fuel options to other bio-energy scenarios, such as electricity generation, and it is reported more frequently in recent studies. This measure is the total energy needed to provide an equivalent of amount of gasoline less all fossil energy uses needed to produce the bio-ethanol:

$$\text{Avoided fossil energy} = E_E + E_{cop} = (E_A + E_B + E_C + E_D)$$

Carbon balancing approaches

Carbon dioxide is the key greenhouse gas responsible for environmental issues of climate change. The production and use of agro-based fuels, however, mitigates the presence of carbon dioxide in the atmosphere, because this carbon dioxide is used by the crops in photosynthesis, converting the carbon released back to biomass, in a complete carbon cycle.

The emissions of CO_2 from fossil energy use, and of other greenhouse gases (notably N_2O in fertiliser manufacture and use, and CH_4 from agricultural and processing operations), should remain as low as possible. These *total CO_2 equivalent emissions*, documented in detail in the studies of Elsayed et al. [24] and Sheehan et al. [19] are:

$$C_{\mathrm{eq.emm.}} = \frac{44}{12}(C_A + C_B + C_C + C_D) + \sum_j GWP_j \left[\sum_{i=a}^{e} X_{ij} \right]$$

A related approach analyses *avoided emissions*, where the use of biomass used as fuel replaces a quantity of fossil fuel that may have been used, or improved efficiency in energy utilisation results in a reduction in fossil fuel use. The CO_2 that may have resulted from its combustion is classified as "avoided emissions", and these figures would vary depending on the energy savings calculated, as well as the measure of relativity on which they are based (e.g., per annum, per kWh electricity produced, per hectare of land, per kilometre travelled, etc.) [25].

$$\text{Avoided } CO_2 \text{ emissions (kg } CO_2 \text{ eq.)} = [C'_E + C_{\mathrm{cop}}]\frac{44}{12} - C_{\mathrm{eq.emm.}}$$

Life cycle assessment

Life cycle assessment (LCA) is a tool for the systematic evaluation of the environmental aspects of a product or service system through all stages of its life cycle. It provides a holistic, i.e. cradle-to-grave, approach to evaluate environmental performance by considering the potential impacts from all stages of manufacture, product use (including maintenance and recycling), and end-of-life management. LCA provides an adequate instrument for environmental decision support. Life cycle assessment has proven to be a valuable tool to document the environmental considerations that need to be part of decision-making towards sustainability. A reliable LCA performance is crucial to achieve a life cycle economy. The International Organization for Standardization (ISO), a worldwide federation of national standards bodies, has standardised this framework within the ISO 14040 series on LCA [26]. There are four basic elements involved in conducting an LCA: (1) definition of the goal and scope of the study; (2) identification and quantification of environmental loads involved; e.g., the energy and raw materials consumed, the air emissions, water effluents, and wastes generated (inventory); (3) evaluation of the potential environmental impacts of these loads (impact assessment); and (4) assessment of available options for reducing these environmental impacts (interpretation).

Whilst LCA is not a single uniform approach at this time, life cycle impact assessment (LCIA) methodology seems to be converging on similar categories [27]. The 10 most common are listed below with brief descriptions. In addition, odour, noise and radiation effects are sometimes included, but their occurrence is not as frequent. Typical LCIA practice employs midpoint modelling. Midpoint refers to the placement along the stressor-impact (cause-effect) chain where the impacts are modelled. For example, the inventory output data for different greenhouse gases is modelled to indicate potential global warming (expressed in CO_2 equivalents, then added up), not the damage caused by climate change. In general this definition works, but it is not applicable to all impact categories. Especially, the categories of human health and ecological health are not considered to have a common midpoint in the cause-effect chain. This has led to the application of various modelling approaches to these categories. Although modelling to the endpoint results in a more environmentally-relevant and meaningful result, this level of detail would require impossibly large amounts of time, data, resources and knowledge of how to interpret the results. Analysis at a midpoint is an effective approach to LCIA in that it reduces the complexity of modelling by minimizing the amount of forecasting and effect modelling. It also results in simplifying communication of the results with fewer categories to report.

Acidification potential: Acidification results when sulphur dioxide and nitrogen oxides reach the atmosphere and react with water vapour to form acids. These acids fall to earth and can damage plants, animals, and structures. Acid deposition can occur through wet (e.g., rain, snow, and sleet), dry, or cloud water deposition (e.g., fog). Acidification compares the capacity of substances to release hydrogen and is expressed in SO_2 equivalents.

Ecological toxicity potential: Ecotoxicity characterization provides a relative prediction of the potential of chemicals to cause harm to plant and animal life. Whilst determining an ecotoxicity potential for a single chemical in a known environment is a difficult task, expanding the list of chemicals and environments to which the modelling is applicable makes this task even more difficult, especially since impacts of the stressors on plant and animal species can have multiple components. A reference chemical is often selected for the comparison, e.g., 2,4-Dichlorophenoxyacetic acid (2,4-D), and thus the units of the ecotoxicity potentials are expressed in kg 2,4-D equivalents/kg emissions.

Eutrophication potential: Eutrophication occurs when fertilisers move from land to surface waters and cause an increase in the aquatic plant growth. This is followed by a chain of other events including fish death, decreased biodiversity, and foul odour and taste. The limiting nutrient is often phosphorus for freshwater systems and nitrogen for estuaries and coastal waters, and thus the location of the release often makes a significant impact on the relative potential for damage.

Global warming potential: Global warming refers to the potential change in climate that may occur with increasing concentrations of "greenhouse gases" which trap heat that would have otherwise passed out of the earth's atmosphere. Resultant effects may include increased droughts, floods, loss of polar ice caps, sea-level rise, soil moisture loss, forest loss, change in wind and ocean patterns, and changes in agricultural production. Greenhouse gases include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_xO), as well as some compounds that are not naturally occurring (hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulphur hexafluoride (SF_6), etc.). The impact of a greenhouse gas is compared to the warming potential of carbon dioxide, so global warming potential is expressed in units of CO_2 equivalents.

Human toxicity potential: Human toxicity characterization provides relative comparisons of a large number of chemicals which may have the potential to contribute to cancer or other negative human health effects. The focus of this category is not on the localised use of chemicals within a work environment (e.g., industrial hygiene), but the long-term exposures to chemicals in the regional and global environment.

Ozone depletion potential: Ozone depletion is the reduction of the protective ozone layer within the stratosphere caused by the emissions of ozone-depleting substances (such as freon, chlorofluorocarbons, carbon tetrachloride, methyl chloroform, etc.). Models often adopt the ozone depletion potentials published in the Handbook for the International Treaties for the Protection of the Ozone Layer where chemical scores are based on CFC-11 as the reference compound.

Photochemical ozone creation potential: Also known as ground-level smog, ozone is formed within the troposphere from a variety of chemicals including nitrogen oxides, carbon monoxide, methane, and other volatile organic compounds in the presence of high temperatures and sunlight. High concentrations of ozone lead to negative impacts on human health and the environment. POCP is often measured relative to ethylene and is expressed as C_2H_4 equivalents.

Natural resource depletion: There are several ways currently being used to analyse resource use, but no method is currently recognised as the standard methodology. Resources are any naturally occurring material, such as ores and fossil energy sources. This category may also include land use and water use.

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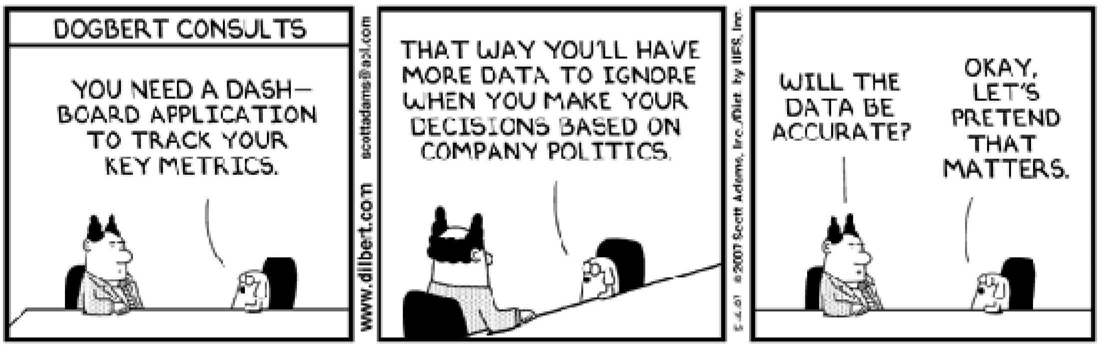
3.4 Discussion



3.4.1 Assessing the Environmental Preferability of Bio-Based Feedstocks

As demonstrated by the two preceding papers, the issue of environmental impacts related to bio-based materials, including bio-fuels, is a complicated one. There is no simple answer to the question "are materials from bio-based feedstocks environmentally preferable?" Bio-fuels, for example, appear to be effective in reducing some aspects (such as fossil fuel use) while increasing others (such as water quality impacts). It brings into question how we define and measure 'sustainability.' Whichever metrics are chosen to measure sustainability, the analysis must be on a life cycle basis.

The pressure to assess the holistic environmental characteristics of products and processes has been more advanced in Europe than elsewhere (Curran 1996). European regulators continue to seek to implement the life cycle concept in policy directives, such as the Integrated Product Policy (IPP) Directive. The lack of an integrated approach across facilities, industries, and media has long been a problem recognized by the U.S. EPA (McCulloch 2003). Current regulations are typically single-media based and do not address environmental problems from the short and long-range perspective.

3.4.2 The Hype on Bio-Fuels – A Modern Day Gold Rush

Amid the current media hype to promote the production and use of bio-fuels, industries are ramping up production of ethanol to meet anticipated future demands. Most of the studies found in the bio-ethanol review focused on the question "do bio-fuels have a net positive energy gain?" As seen in the discussion, this is the topic of much debate. It is likely that the cause for much of the variation is found in the assumptions behind the calculations of the input and output numbers. Some researchers assign energy credits to the bioethanol system for producing co-products, such as corn oil, corn meal, dried grains, etc.

Energy efficiency seems to be the logical question with the underlying question being "Are we getting more out than we put in?" According to the literature, the response, with few exceptions, seems to be an overwhelming 'yes!' (http://www.iowacorn.org/ethanol/documents/energy_balance_000.pdf). But energy balance asks the wrong question. This debate is irrelevant and misleading since, as Dr. Bruce Dale puts it, all BTUs are not created equal. We pay approximately 12 times[3] as much for a BTU of electricity as we do for a BTU of coal since electricity has a much higher utility (it should also be pointed out that current fuel prices do not cover most of the externalized impacts or costs, adding to the complexity of the comparison). A net energy balance assumes that one BTU of energy available from any energy carrier is equal to a BTU from any other energy carrier. This assumption is invalid since we do not value energy *per se* but rather we value the service it offers (Dale 2006).

3.4.3 Identifying Opportunities for Improvement

Instead of debating the merits of energy efficiency, another view that could be taken is from the field of Pollution Prevention (also called Cleaner Production). A Pollution Prevention Opportunity Assessment identifies where an industrial system can be improved, usually at the plant level (a gate-to-gate approach). Expanding this approach would allow the use of the results of the LCAs to identify opportunities for improvement at the system level. For example, organic farming would greatly reduce the impacts in the corn growing phase related to run-off of pesticides, fertilizers and herbicides into water sources.

But caution must be used, along with a life cycle view, when alternatives are identified, for example, the switch to so-called organic solutions. Using animal manure to fertilize crops is a cost-effective replacement for commercial fertilizers; however, any fertilizer can have undesirable effects when it enters nearby streams and lakes. Pathogens in manure can make water unsafe to drink or use for recreation. The nitrogen and phosphorus that are needed to grow crops can lead to eutrophication and undesirable algae blooms when they run off into the water. These effects not only hinder recreational use and aesthetics, they also deteriorate the ecosystem (http://www.dnr.state.wi.us/org/water/wm/nps/ag/waterquality.htm). For the use of any type of soil amendment, timing, application rates, soil types, slope, etc., must be taken into consideration in order to optimize effectiveness and reduce potential risks.

[3] A short tonne of coal costs approximately $40 and has an energy value of 20,754,000 BTU while at 8¢ per KWH, an equivalent amount of electricity (6,083 KWH) would cost $486.

3.4.4 Integrating LCA with Other Aspects Needed in Decision-Making

The U.S. Congress is moving rapidly to spur production of bio-fuels in an attempt to reduce oil dependency. Governmental agencies, private organizations and individuals are being encouraged to purchase products and services that are “environmentally preferable.” The main intent of such calls to “buy green” is to purchase products with reduced environmental impacts. But how *does* one determine environmental preferability? Often the bottom-line is to insist on buying products or requesting services that require fewer non-renewable resources, hence the requirement for recycled or bio-based products. The fight for green procurement has advanced dramatically in recent years; however, there is a potential danger in looking too narrowly at product requirements. Any comparison of products to determine preferability must assess all the relevant environmental impacts across the full product life cycle.

Sustainable development requires the consideration of the economic, environmental and social aspects of products and product systems. Therefore, responsible decision-making in public policy, industry and related fields should consider those issues for present and future relevance. Currently there is no single program or technique that is capable of delivering an overall answer with regard to environmental decision-making. Research is needed to develop a framework which integrates results of LCAs into the decision-making process along with other pertinent factors, especially societal needs.

If used properly, bio-fuels can *help* us meet our energy needs while maintaining ample supplies of food, animal feed, and clean water supplies. To make this happen, well thought out national bio-fuels policies that support the best options are needed for both the short and long-term future.

4 Allocation Methodology in LCA

4.1 Chapter Overview

Chapter 2 (Section 2.6) identified 12 key issues within LCA methodology where the decisions and assumptions that are made by the LCA practitioner can have a great effect on the outcome of the study. This chapter focuses specifically on the topic of co-product allocation. It describes the events that led to the current focus on allocation research as presented in Section 4.2. It then defines allocation methodology in Section 4.3, and includes a paper that reports on the findings of a literature review that was conducted by this thesis author ("Co-Product and Input Allocation Approaches for Creating Life Cycle Inventory Data: A Literature Review") in Section 4.4. A discussion of the literature review is presented in Section 4.5. Finally, in Section 4.6 an overall discussion and conclusions for Chapter 4 are presented.

4.2 Events Leading to the Current Research on Allocation

In May 2001, efforts to develop a U.S. national life cycle inventory database were initiated. In July 2001, a meeting of the advisors to the US Database effort was held in Washington, D.C. A number of issues surfaced during this meeting and those issues dominated the discussion throughout the *Guidelines* development process. Examples are: a. the appropriate methods for co-product allocation, and b. the proper allocation for materials that are reused or recycled. Issues such as these are controversial because different industries can be affected differently by alternative choices, in part, because the LCA experts themselves may strongly disagree on the choice of methodologies, and, in part, because there can be different approaches to LCA (e.g., attributional versus consequential) which may dictate different methods to a task like co-product allocation (National Renewable Energy Laboratory 2003).

One particularly vociferous participant persistently supported the use of economic allocation (letter from N. Howard to M. Deru, Nov 2004 – see text box below), but this suggestion was met with much resistance. This debate made it clear that while the various practitioners had preferences on how to apply allocation, additional research on this topic was needed. Specifically, research had apparently not been done to systematically compare allocation techniques.

November 1, 2004

Michael Deru
National Renewable Energy Laboratory

Re: US LCI Database Project Meeting Minutes

Dear Michael,

Thank you for the minutes of the meeting. I'm sorry that I wasn't able to stay for the full time, but those that relate to the part that I was present at appear to be a good record of the meeting in that they do report what people said. However, some of the statements themselves are either misleading or factually incorrect. I would like to comment on a few key ones, which are of course related to allocation. This is what you all expect from me and I don't want to disappoint.

I would be grateful if you could circulate my responses to the whole group.

Co-Product Allocation – bullet 3 states:
"Economic value rarely correlates to the true environmental impacts, and physical or energy parameters are often preferable for allocation (e.g. surface area is a better predictor of paint use than is value)"

The example in this comment is not helpful. The first step in ISO 14041 is to try to avoid the need for allocation and it is ridiculous to assert that allocation is needed for painted goods – you can always directly measure the amount of paint used and would never need to revert to allocation in this over-simplistic example. Moreover, surface area might be a badly inaccurate allocation method if the porosity of the surfaces being painted was different. Furthermore, if the porosity of the surfaces were the same, economic allocation would give exactly the same answer as area allocation because the value of paint used would be the same for both articles.

Overall, though, this comment reveals an even deeper misunderstanding. This misunderstanding lies at the heart of why I have found this project

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so frustrating to participate in. The comment assumes that there is some TRUE PRIMARY STANDARD of allocation that economic allocation and the myriad of possible physical allocations can all be compared to. This is completely tautological thinking – advocates of physical methods will SEE TRUTH in their approach, advocates of economic allocation will SEE TRUTH in their approach. Sensitivity analysis only reveals the differences between different CHOICES, but reveals nothing about which is TRUE. The CHOICE made by the LCA practitioner is an arbitrary and philosophical choice and not objectively determined. The choice is between:

- Use of physical units chosen case by case by the practitioner because they believe that the chosen unit best represents the CORRECT allocation. It is an arbitrary choice and needs to include a delineation of what constitutes waste or co-product. To have any credibility, these choices need to be supported by peer review. The approach is intrinsically inconsistent between process stages and between products, because it is universally impossible to use the same physical property for all stages up and downstream for any real process.

Or,

- Use of economic allocation completely consistently throughout with no arbitrary choices made and no need to define wastes (cos they automatically have no value and attract no burdens) from co-products and a completely consistent handling of recycled and down-cycled materials. The method is objective and philosophically sound in that investment (cause) in the process causes, the impacts (effects) from the process which must then be allocated in proportion to the way they generate return on investment from sales of the product stream outputs.

In a nut-shell – there is no TRUE in allocation - just better or worse choices. The discussion should be about whether economic allocation is better or worse than arbitrary free-for-all in physical allocation, not trying to pick holes in the fabric of economic allocation when in physical unit allocation the holes are bigger than the fabric.

Co-Product Allocation – bullet 11 starts:
"How do you handle price changes to materials from older plants with rigid technology that cannot adjust production outputs to reflect changing values?"

I was not able to respond at the meeting, but the answer is that you simply accept the values as they fall irrespective of the plant's ability to adapt. It remains stubbornly true that the return that the plant owner gets from his/her investment in this rigid technology is less than it would have been if he/she'd invested in more adaptable plant. SO WHAT? is the answer – the economic allocation still works philosophically and practically.

Co-Product Allocation – bullet 13 starts:
"On the other hand, LCA is data intensive, and we must rely on other databases. Fairness is important, including uniform allocation, but there must be a balance. Sometimes, we have to accept what we can get, and cannot discount or throw out data because it is based on other allocation methods."

The whole object of the US LCI Database project was, from my understanding, to get it right without compromise, but on a unit process basis to constrain the problem. I think that this proposal represents an unacceptable compromise that just perpetuates the myriad of inconsistent data and doesn't provide a level playing field for building material, product and system comparisons. I hope to do better with the LCA into LEED project.

Co-Product Allocation – bullet 15 starts:
"One route to consistency is to use the same allocation method within material families. That kind of consistency is supported when practitioners draw common data from the database project"

At first sight, this seems an OK compromise for specific product choices – window A vs window B - but it falls down badly in building systems where the choices are made between materials/products as composites coming from very different industries. It was this factor that forced the UK stakeholders to decide that economic allocation was the ONLY viable fair level playing field method for consistent comparison.

I hope everyone is not too cross with me!

Regards,

Nigel Howard.

Nigel Howard
VP LEED & International
US Green Building Council

During the annual conference on LCA (InLCA) held by the American Center for LCA in September 2003, in Seattle, Washington, a discussion group was convened to identify issues regarding methodology for collecting, modeling, calculating and estimating life cycle inventory data in order to develop more uniform international practices. The intent of this meeting was to determine if there was general agreement for the need to develop "rules" for collecting LCI inventory ("data collection" includes, at a minimum, measuring, estimating, modeling, and calculating data). The strong turnout (over 50 participants joined in the discussion) indicated the underlying need for better guidance. A summary of this workshop was subsequently presented at the "International Workshop on Data Quality" in Karlsruhe, Germany, in October 2003 (Curran 2003).

Through a combination of material that was prepared in advance of the Seattle meeting and the presentations and discussions that were held during the meeting, the following "Decision Points in LCI" were identified by the workshop attendees as areas in which precise guidance is needed in order to make LCI practice more uniform internationally:

- Co-product allocation;
- Recycling allocation;
- Exclusion of small amounts;

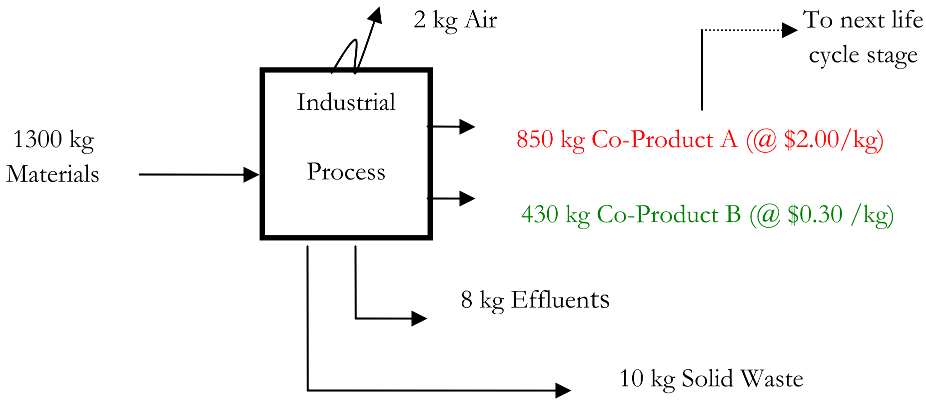
- Exclusion of spills and losses;
- Age-appropriateness of data;
- Surrogate and estimated data;
- Inventory for impact assessment (both, non-specific (e.g., VOC's) and too specific);
- Matching the goal to the method;
- Collecting primary data (primary = first source; secondary = published);
- Report format;
- Iterative procedure of data collection;
- Choosing boundaries;
- Capital equipment and infrastructure exclusions; and
- Time and location meta data.

Although this list was not prioritized during the session, co-product allocation is clearly an important aspect of LCI methodology and in need of further research. Therefore, this thesis author now focuses upon the issue of Co-Product Allocation.

4.3 Defining Co-Product Allocation

Industrial processes typically make more than one product. When conducting an assessment, the LCA practitioner must decide how to allocate the environmental releases emitted by the process across each co-product. This approach is illustrated in Figure 4-1 using a hypothetical product A and co-product B. The figure shows how the results for air, water and solid waste releases from a process that produces two co-products can vary depending on the basis used for the calculation. Scheme 1 is based on the weight of co-product A compared to co-product B. Scheme 2 is based on market value, where co-product A is sold at $2 per kilogram and co-product B is sold at $0.30 per kilogram. Scheme 3 views co-product A as the main product intended to be produced by the process and allocates 100% of the emissions to A.

Figure 4-1 Three possible schemes to allocate emissions and effluents from an industrial process to Product A based on weight, market value and intent



Allocation Scheme	Air Emissions	Effluents	Solid Waste
1. By weight	1.33 kg	5.31 kg	6.64 kg
2. By market value	1.86 kg	7.47 kg	9.29 kg
3. By primary product	2.00 kg	8.00 kg	10.00 kg

The International Standards Organization (ISO) deals directly with allocation and calls for practitioners to avoid allocation if possible, by either 1. Modeling the sub-processes involved in production (i.e. collect more detailed data), or 2. Expanding the system boundaries to include additional processes that relate to the co-product(s). There is general agreement that avoiding allocation through sub-process modeling and system expansion is an appealing way to handle this seemingly intractable problem. Both approaches cause the model to get larger and more complicated, requiring the collection of more data in order to complete the analysis. Collecting more data means more time and effort which brings the practicality of the approach into question. Also, larger systems run the risk of being less transparent in that there is more information on how the data were arrived at than can be easily communicated. So, although the answers that would be obtained through sub-process modeling would be more relevant to sustainability and more useful in helping decision-makers make better decision, allocation may not always be avoidable, especially if the data for the sub-processes or for the expanded system cannot be easily acquired.

The following section includes a paper that summarizes a review that was conducted by this thesis author in order to determine the level of general understanding and knowledge about co-product allocation that is present in the current literature. A literature search was conducted (and completed in December 2005) in order to find publications that address the variety of allocation schemes that are applicable to creating LCIs specifically for industrial processes. The reviewer found that while the need to match the goal of the LCA to the approach is an often recurring theme in papers and publications, general guidance is not yet available for how or when to apply allocation rules.

4.4 Paper VI - Co-Product and Input Allocation Approaches for Creating Life Cycle Inventory Data: A Literature Review

Curran, MA (2007) "Co-Product and Input Allocation Approaches for Creating Life Cycle Inventory Data: A Literature Review." Springer/Kluwer Academic Publishers, *International Journal of Life Cycle Assessment* 12 (1) pp65-78. DOI: http://dx.doi.org/10.1065/lca2006.08.268.

Review Articles

Co-Product and Input Allocation Approaches for Creating Life Cycle Inventory Data: A Literature Review

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Abstract

Goal and Background. Allocation methodology for creating life cycle inventories is frequently addressed, yet the practice continues to be in a state of flux. Clearly there are many ways in which allocation can be carried out with no single method providing a general solution. ISO 14041 identifies a methodological framework, although it does not provide specific guidance on when and how to apply the steps that are outlined in the standard. An expansion, or elaboration, of the current ISO framework for allocation is needed.

Method. A literature search was conducted on the various allocation schemes that are used to create life cycle inventories, with a focus on industrial processes. The results are grouped by 'general guidelines' and 'industry-specific' applications.

Results and Discussion. The search uncovered procedures that support larger efforts, such as the U.S. Database Project and CML's Operational Guide. Other researchers conducted industry-specific studies to examine the effect of that varying inputs has on outputs. These studies typically attempt to demonstrate how allocation depends on a system's operation.

Conclusions. A recurring theme is the need to match the methodological choice with the goal of the study. However, guidance remains lacking. While system expansion is the preferred approach and avoids allocation altogether, it leads to a larger, more complicated model that requires more data. Data accessibility, time, and effort become significant and bring the practicality of applying system expansion into question

Recommendations and Perspectives. It would be useful to develop the range of allocation approaches aligned with different applications (i.e. goals). These approaches should be tested in various case studies for further discussion within the LCA community. Emphasis in these tests should be on the effect of modelling variations on the decision outcome when analyzing an entire system and not focus at the single process level.

Keywords: Co-product allocation; goal definition; life cycle assessment (LCA); life cycle inventory (LCI)

Introduction

Creating a life cycle inventory by collecting input and output data to represent a technology or an industry sounds like a straightforward process; however, how one models an industrial process or technology can vary widely depending on the underlying assumptions that are used to calculate the inputs and outputs. In particular, the choice of an allocation approach can have a profound effect on the results of the data that are entered into the inventory. Allocation in a life cycle inventory includes the allocation of products over different functions, the allocation of aggregated environmental data over individual processes, the allocation of emissions over different environmental compartments, and the allocation of emissions over parallel or serial environmental mechanisms [1].

An important element in these discussions is the distinction that has been made in life cycle assessment practice between foreground and background data (**Fig. 1**). The foreground refers to the part of the system that is of primary concern, i.e. the system that the decision maker can influence directly and without relying on market forces. The background is the part of the system that delivers energy and materials to the foreground as aggregated data sets in which individual plants and operations are not identified. The selection of foreground or background data decides if either marginal or average data are to be used [2].

In 1996, SETAC took the aforementioned distinction one step further and categorized changes as marginal, average, or discrete, depending on the goal of the study:

1. **Marginal changes to a specific technology** are relevant when the performance of the specific process is analyzed and when changes around the system of interest are incremental. This would be the case when analyzing a waste in-

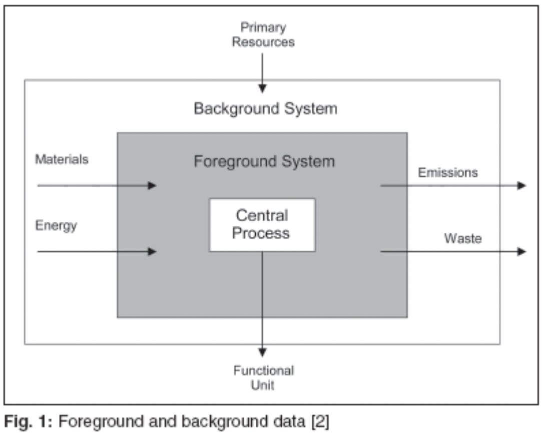


Fig. 1: Foreground and background data [2]

cinerator operation in relation to marginal changes in chlorine content or calorific value, while other variables are held constant. It is also relevant in the co-product and recycling systems, where outputs can be changed independently of each other and the system is analyzed on marginal changes in these outputs.

2. **Average changes to a specific technology** are applicable to comparing a new product to an existing one that would lead to a change in an existing specific technology. In this case the aim of the study is to compare the average behavior of a new and existing technology, rather than marginal changes in the behavior of the existing technology. In the case of a waste incinerator, an example would be assessing the average changes in the system that would occur due to the addition of the air emission control equipment to the incinerator.

3. **Average changes to an average technology mix** are relevant if different processes or products with a similar function are compared. In this case, average changes due to a shift to a different technology mix are considered; e.g., comparing different waste management options, such as incineration and recycling.

4. **Discrete changes in technology mix** are applicable when fundamental changes in society are considered that would influence a large number of technologies. One such discrete change would be, for instance, a shift to a chlorine-free economy which would mean phasing out all products that contain chlorine and introducing a completely new mix of technologies for producing alternative products [3].

1 Life Cycle Inventory Methodological Issues

Life cycle inventory methodology has been the subject of several workshops and workgroups in an effort to make the practice more transparent and uniform so that methods are not selected on an arbitrary basis [4–6].

During the 2003 InLCA/LCM conference in Seattle, Washington, a session was organized with the specific intent of initiating open discussion on inventory methodology and determining if there was support behind the idea of developing international procedural guidelines for inventory, going beyond the ISO 14040 and 14041 guidance [7]. The general consensus of the group in Seattle was that there is a need and desire for more detailed guidance, especially around the following list of suggested key decision points within life cycle inventory:

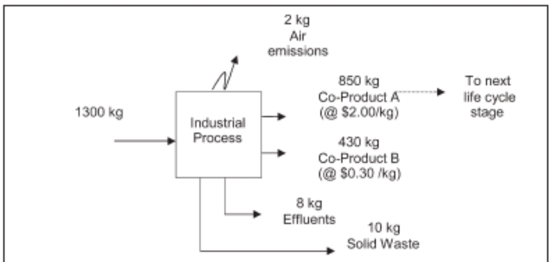
- Co Product Allocation
- Recycling Allocation
- Exclusion of Small Amounts
- Exclusion of Spills and Losses
- Age-Appropriateness of Data
- Surrogate and Estimated Data
- Inventory for Impact Assessment
- Matching the Goal to the Method
- Collecting Primary Data
- Report Format
- Iterative Procedure for Data Collection
- Choosing Boundaries
- Capital Equipment/Infrastructure Exclusions
- Time and Location Meta Data

Although the discussion did not prioritize the items on this list, allocation related to co-products and recycling was clearly a significant concern.

1.1 Co-product allocation

All industrial processes have multiple input streams and many generate multiple output streams. For example, steam turbine systems may sell both electricity and low pressure steam as useful products. Usually only one of the outputs is of interest for the life cycle assessment study being conducted, so the analyst needs to determine how much of the energy and material requirements and the environmental releases associated with the process should be attributed, or allocated, to the production of each co-product. The practitioner must also decide how to allocate environmental burdens across co-products when one is a waste stream that can be sold for other uses.

Fig. 2 depicts a hypothetical industrial process that generates Products A and B and assigns air emissions, effluents and solid waste to Product A, the product of interest, using different bases for allocation. In this simple example, weight, market value and intent are demonstrated as ways to allocate environmental releases.



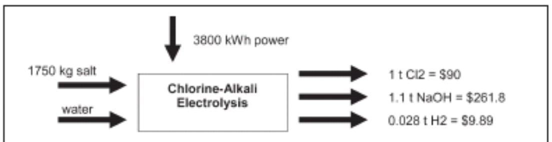
Burdens assigned to Product A

Scheme	Air Emissions	Effluents	Solid Waste
1. By weight	1.33 kg	5.31 kg	6.64 kg
2. By market value	1.86 kg	7.47 kg	9.29 kg
3. By primary product	2.00 kg	8.00 kg	10.00 kg

Fig. 2: Three possible schemes for co-product allocation of life cycle inventory data (for an assessment of product A) [8]

1.2 Material input allocation

A parallel issue regarding allocation involves the assignment of material and energy into a process (**Fig. 3**). This issue does not receive as much attention as co-product allocation although it can have just as much of an impact on the results.



Allocation results:

Process-Input	Complete Process	Allocation Basis	1 ton Cl_2	1.1 ton NaOH	0.028 ton H_2
Power	3800 kWh	Mass	1786	1965	50
		Market Value	945	2750	105
Salt	1750 kg	Mass	823	905	23
		Market Value	435	1267	48

Fig. 3: PE's example of allocating power and salt inputs to a chlor-alkali electrolysis process by mass and market value (PE Europe Gmbh, 'Software training 3: Allocation – Basics, example and tasks', on-line)

2 ISO Standard on allocation

The guidance provided by the International Standards Organization (ISO) recognizes the variety of approaches that can be used to treat the allocation issue and, therefore, requires a step-wise approach (see text box on ISO 14041). The standard calls for practitioners to avoid allocation if possible; and secondly, to model approaches which reflect the physical relationships between the process outputs and its inputs. Proper application of the ISO guidelines on allocation requires a good understanding of the physical relationships between co-products in a process.

ISO 14041: 6.5.3 Allocation Procedure [9]

On the basis of the principles mentioned above, the following stepwise procedure shall be applied.

Step 1: Wherever possible, allocation should be avoided by:

1) Dividing the unit process to be allocated into two or more subprocesses and collecting the input and output data related to these subprocesses.
2) Expanding the product system to include the additional functions related to the co-products, taking into account the requirements of (function, functional unit and reference flow).

Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way which reflect the underlying physical relationships between them, i.e. they shall reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system. The resulting allocation will not necessarily be in proportion to any simple measurement such as mass or molar flows of co-products.

Step 3: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way which reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products.

For allocation in open-loop recycling, ISO 14041 [9] requires that the same procedure be followed although it allows for a variation to Step 1. If recycling does not cause a change in the inherent properties of the material, allocation can be avoided by calculating the environmental burdens as if the material were recycled back into the same product. Otherwise, allocation is based on physical properties, economic value, or the number of subsequent uses of the recycled material.

The international standard fails to deal with all the aspects that are involved in allocation methodology. It does not include information on the effect of the different methods on the life cycle modelling, for example the feasibility of the methods, the amount of work required, or the type of information that results from the application of the methods. Furthermore, ISO's solution to the allocation problem has been criticized for its failure to account for different situations (e.g., 10, 11, 12). Ekvall & Weidema [13] suggest that there may be a need to refine the methodology, both by adjusting the ISO standard and by creating additional recommendations. The following sections highlight key issues related to allocation.

2.1 System expansion and the avoided burden approach

As noted previously, the preferred approach identified by ISO 14041 is system expansion. Azapagic [2] provides a good description of the system expansion approach and explains how it can be approached in two different ways, depending on the goal of the study (note, however, that for both the intent is to compare systems).

One approach for system expansion broadens the system boundaries and introduces a new functional unit to make the two systems being compared equal in scope (**Fig. 4**). Take for example Product A which is produced by Process AB along with co-product B. Product A is to be compared to Product C which is the only product that is produced by Process C. Using system expansion, an alternative way to produce Product B, i.e. Process D, is added to Process C. The comparison is now between Process AB and Process C plus Process D.

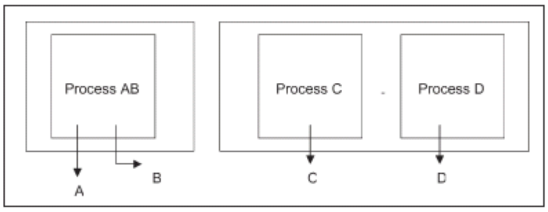


Fig. 4: Expanding boundaries to compare processes making products A, B, C and D

Another approach to applying system expansion is by subtracting the environmental burdens of an alternative way of producing Product B (using the same example as before) so that only Product A is compared to Product C. This approach is also referred to as the *avoided burden* approach since it is reasoned that the production of any alternative products is no longer needed and the associated environmental burdens are avoided. The environmental burdens allocated to the product of interest are then calculated as the burdens from the process minus the burdens of an alternative co-product. For example, a process that also generates heat, such as a refrigerator, offsets some of the need for space heating which would be supplied by some other source. The emissions avoided through this reduced demand might include emissions such as CO_2, SO_2, NO_x, CO and hydrocarbons that are typical from power generation. This process can result in negative accounting of burdens if the subtracted releases do not occur in the main product system.

System expansion simplifies modelling and limits the assumptions that the modeler needs to make. However, broadening the system boundaries makes the process of data collection much more complicated. Not only are more data needed, but appropriate data are needed.

2.2 Underlying physical relationships

A physical relationship, or physical causality, is an important distinction in allocation for justifying if a real industrial situation exists. Physical relationship exists when the amounts of the individual co-products are interdependent. Weidema and Norris [14] assert that "allocation by product mass is justifiable in co-production when this mass is actually determining the volume of flows of the co-producing process". That is, an increase in output of a specific co-product causes an increase in production in direct proportion. Similarly, they apply this logic to economic allocation which is a justifiable approach when the volume of the co-producing process varies in proportion to the changes in economic revenue to the process from the different co-products. An example of this is joint production of metals from ore in which there are no alternative production routes or substitutes for the metals in question. The authors conclude that since allocation by mass may be justifiable in one situation although not in another, no allocation key seems to have global applicability.

3 In Search of a General Solution

Although avoiding allocation is favored by the ISO standard, it is not always possible to expand systems in all cases. And, as alluded to, allocation cannot be totally avoided even in a system expansion approach (this is discussed later). Therefore, other options must be used.

Mass has been most often used as a basis for allocation. Allocation by volume is done in a similar way. Methods based on market value usually include expected economic gain based on gross sales. However, none of these methods offers a general solution. Allocation may seem impractical in cases where one product far outweighs another. Although market value in most cases reflects the use of energy and therefore many of the associated burdens, allocation on this basis covers only one aspect of the system. Also, market value is highly variable over time, sometimes up to 50% in a short time period. Allocation on an equal basis (50/50) or on an 'all or none' basis (100% to one product) can be considered to be a highly arbitrary choice [2].

Environmental burdens related to the alternative systems must still be modeled using an appropriate method where co-products are generated. A lot has been published in the open literature on the subject in an effort to better understand the consequences of allocation choices.

Klöpffer [15] reviewed different allocation rules for recycling that are proposed in seven sources that he found in the open literature [4,16–21]. While the approaches described by these sources vary from a simple to more complicated, he concludes that two trends in allocation methods for open-loop recycling emerge:

1. A trend toward scientifically unambiguous solutions, regardless of their practicability.
2. Simple rules, which are arbitrary to some extent, but feasible also in the analysis of complex systems.

Further, Klöpffer suggests the following three criteria for establishing a reasonable allocation rule:

3. Mathematical neatness, internal logic, no double counting.
4. Feasibility at a low level of information with regard to the actual use or origin of the secondary raw materials.
5. Justice and incentive for producers and users of secondary materials.

Klöpffer applying these criteria to three selected allocation rules: 50:50 split, i.e. allocating environmental burdens equally across co-products; shifting the secondary raw materials outside the boundaries, i.e. crediting the originating process with the amount of waste that is avoided; and system coupling, later known as system expansion. He concludes that, the second rule (shifting the burden outside the boundaries) is the most feasible for the most general case of complicated systems where little information is available regarding the origin and fate of secondary raw materials.

Klöpffer identified just a few reports that deal with the allocation issue. Since his 1996 review, many other pieces have been written and published. The following sections summarize the key findings of a search of the recent literature exploring methodology dealing with input and output allocation in creating a life cycle inventory of industrial processes.

4 Highlights of a Literature Review on Input/Output Allocation Methodology for Life Cycle Inventory of Industrial Processes

A literature search was conducted (and completed in December 2005) in order to find publications that address the variety of allocation schemes that are applicable to creating life cycle inventories specifically for industrial processes. The publications that were chosen for this study either indicate current state-of-the-practice by providing general guidance or they provide side-by-side comparisons of alternative allocation approaches. The search uncovered several papers that address the allocation issue directly, although in general terms. Procedures that were developed in support of larger efforts, such as the U.S. Database Project [22] and CML's Operational Guide [23], provide general guidelines for life cycle inventory methodology and give good insight into how allocation issues are being addressed in the development of larger datasets. Other researchers looked at industry-specific studies that dealt with the creation of foreground data thus allowing for the modelling of varying inputs to examine the effect on the outputs, or vice versa (varying the outputs to examine the effects on the inputs). These studies typically attempt to demonstrate how burden allocation depends on the way the system is operated.

The more significant writings found in the open literature are briefly described below, under the headings 'General Guidelines and Studies' and 'Industry-Specific Applications'. The following discussion is not intended to be a comprehensive review of all the writings on allocation that were found in the literature. Instead, selected references are presented in order to highlight the key themes, commonalities and differences found throughout these publications.

4.1 General guidelines and studies

4.1.1 EPA's life cycle assessment: Inventory guidelines and principles

One of the earlier documents that provided guidance on how to conduct life cycle assessment was published by the US EPA [19]. The preparation of the manual relied heavily on the experience of Franklin Associates. This is apparent through statements such as the following on page 57:

"If a process produces several different chemical products, care must be taken in the analysis. It will be necessary to write balanced chemical equations and trace the chemical stoichiometry from the raw materials into the products. A simple mass allocation method frequently gives reasonable results, but not always. In calculating energy, heat of reaction may be the appropriate basis for allocating energy to the various co-products. These calculations can become quite complex, but if the chemical products being produced are similar, *experience shows* (emphasis added) that a simple mass allocation very closely approximates the results of even more complex calculations."

The guide includes statements such as "A basis for allocation needs to be selected with careful attention paid to the specific items calculated", and "...each industrial system must be handled on a case-by-case basis. No allocation basis exists that is always applicable". The guide goes on to address allocation of material inputs. Different bases for allocation are compared, such as for an electrolytic cell in which 1 mole of hydrogen (2 grams) and one-half mole of oxygen (16 grams) are produced from water. The allocation of electricity on a molar basis yields two-thirds of the electricity input being attributed to the hydrogen; on a mass basis the allocation would be one-eighth. Citing conservation of mass, it is concluded that a mass allocation would be appropriate for materials calculations.

The EPA guide allows for a combination of allocation bases to be used in constructing a life cycle inventory, yet it very clearly endorses the use of mass allocation, referring to it as common practice based on chemical engineering, chemistry and physics and a reasonable modelling technique. The guide addresses the notion of using selling price as a basis for co-product allocation, concluding that this approach is not entirely satisfactory because the selling prices of the co-products vary greatly over time. Although mass basis is not ideal, it is widely recognized and produces a predictable and stable result.

4.1.2 Argonne's greenhouse gases, regulated emissions, and energy use in transportation (GREET) model

Since 1995, the US Department of Energy's Argonne Laboratory has been developing the Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) model which is intended to serve as an analytical tool for estimating fuel-cycle energy use and emissions associated with alternative transportation fuels and vehicle technologies [24]. Although GREET is an energy-based tool, its modelling deals with the same allocation issues that are encountered in a life cycle assessment. The GREET method for allocation is based on what is called the 'replacement value of co-products'. Using a corn mill as an example, the developers of GREET provide the following reasoning behind its selection.

"The co-products used in this analysis include distillers dried grain solids (DDGS) from dry milling, and corn oil, corn gluten meal, and corn gluten feed from wet milling. There are basically four ways to estimate energy credits for co-products. First, the energy content of co-products can be used to estimate energy credits. For example, a pound of corn gluten meal or corn gluten feed has a caloric content of 8,000 Btu. This results in about a 40-percent co-product energy credit. The disadvantage of this method is that calories are a measurement of food nutritional value and are not a good proxy for energy in a fuel context.

A second method of estimating co-product energy values is to use the relative market values of ethanol and its co-products. For example, if energy used to produce ethanol is allocated between ethanol and co-products based on their 10-year average market values, about 30 percent of energy used to produce ethanol should be assigned to the co-products. The problem with this method is that prices of ethanol and ethanol co-products are determined by a large number of market factors that are unrelated to energy content.

Third, one can allocate energy use among multiple products on an output weight basis, regardless of the operation's purpose or the co-products' economic values. If energy used to produce ethanol is allocated between ethanol and co-products based on the output weight, about 48 percent of energy used to produce ethanol should be assigned to the ethanol and 52 percent to co-products. The problem with this method is that weight of a product is not always a good measurement of its energy value.

A fourth method, based on the replacement value of co-products, is the method chosen for our final results. Energy credits are assumed to be equal to the energy value of a substitute product which the ethanol co-product can replace. For example, in the case of corn gluten meal and corn gluten feed, soybean meal can be used as a substitute, and soybean oil can replace corn oil. Using this method, about 20 percent of the energy used to produce ethanol would be assigned to co-products. This method has appeal because the co-product value is measured by energy units unlike the other methods that use calories, economic value, and weight to represent energy value. Also, since energy replacement values result in fewer energy credits than the other methods, it can be considered a conservative estimate [25]."

4.1.3 NREL's US database development guidelines

The U.S. Department of Energy's National Renewable Energy Laboratory (NREL) is heading up an effort to develop a database for life cycle inventory. The ultimate objective of the effort is to make data modules for commonly used materials, products and processes publicly available for use in the creation of life cycle inventories. The purpose is three-fold: 1) to support public, private, and non-profit sector efforts to develop product life-cycle assessments and LCA-based decision support systems and tools; 2) to provide regional benchmark data for generating or assessing company, plant, or new technology data; 3) and to provide a firm foundation and broad data resource base for conducting life cycle assessments generally.

The development of the data follows the ISO three-step hierarchy for allocation and is described in the project development guidelines [22].

Step 1 is to *avoid* allocation altogether. When the 'unit process' actually consists of multiple parallel sub-process chains, then using greater modelling detail may avoid the need for allocation. The avoidance approach described by ISO can require significant extra data gathering and modelling if there are multiple displaced products and/or production routes, and if the data for these displaced products are not already available. Therefore, it is recommended that system expansion be used only when there is a dominant, identifiable displaced product; and when there is a dominant, identifiable production path for the displaced product.

Step 2 allocation basis must reflect the 'underlying physical relationships' between the elementary flows and the output shares. The result of such system study or regression analysis may turn out to be equivalent to apportioning based on mass or some other physical characteristic of the product outputs, but ISO explicitly states that Step 2 is not the same as *a priori* apportioning of elementary flows to co-products according to their mass or molar shares.

Step 3 is necessary if Steps 1 and 2 cannot be done. Step 3 is to use some *other relationship* between the elementary flows and the product output as a basis for allocation. Here, ISO identifies economic allocation as the first preference among Step 3 options, and use of the economic value shares among the product outputs is recommended where feasible.

In summary, the NREL guidelines prescribe that allocation practice should follow three steps:

1. Be fully consistent with the ISO hierarchy among methods; avoid allocation if possible; use physical relationships that approximate the causal influence of output share variations upon elementary flows; or use other cause-approximating relationships.
2. Be fully consistent with the ISO requirement to document the reasons for every choice of allocation method, and provide enough transparency to enable interested life cycle assessment practitioners to conduct their own tests of the sensitivity of results to the selection of different allocation approaches.
3. Where possible, carry out sensitivity analyses to illustrate the variability in results for alternative allocation methods.

The guidelines address the implications for both the producers and users of scrap materials and the issue of 'incentive' to recover material. The approach to allocation depends on if the scrap is treated as a waste or as a co-product. The two approaches and related incentives are described in the project guidelines as follows.

4.1.3.1 Scrap as waste method

Under the waste method, the primary product carries all of the inventory elementary flows, with the reduction in solid waste assigned to the primary product as the only environmental incentive for the producing facility to find or create markets for the scrap. On the other hand, there is more incentive to recover materials by potential secondary users, since they are 'free' of all upstream burdens for production.

4.1.3.2 Scrap as co-product method

When scrap is treated as a co-product, the elementary flows that otherwise would have been allocated solely to the primary product(s) are now allocated among the scrap and primary product(s). This might encourage the producer to sell the scrap, rather than having it discarded as waste, in order to reduce the elementary flows allocated to the primary products. On the other hand, potential users of recovered materials with allocated elementary flows may be discouraged from doing so.

4.1.3 Recommendation for economic allocation

Given these alternatives and their implications, data developers may want to carry out sensitivity analyses to determine the range of values resulting from the different methods. However, the modular, generic approach of the U.S. LCI Database Project does not support the provision of ranges or alternative sets of results in the final posted datasets.[1] Subsequent versions of the database format may support data quality ranges, but at this point, the project requires one uniform approach.

In principle, by-products such as trim scrap could be subjected to the full ISO hierarchy, as for any co-product. In practice, however, it is unlikely that step 1 will apply or that a causal relationship will be found to satisfy the Step 2 criteria. It is, therefore, recommended that economic allocation be used for by-products, as per Step 3, unless there is a clear reason to undertake the additional analysis required for Steps 1 or 2 in the hierarchy. The ultimate rationale for applying a more onerous Step 1 or Step 2 approach is a high probability that such an approach will strike a better balance among the data criteria of accuracy, intelligibility, persuasiveness/credibility to users, and affordability (in time and money terms).

4.1.4 Ecoinvent

Ecoinvent is a database system that was designed to accommodate unallocated multi-output processes and their derived single (co-)product output processes [26]. The system is fairly flexible in its guidance for applying allocation. Allocation factors are separately recorded and may be adjusted accordingly. The user may choose between the following allocation schemes: undefined, physical causality, economic causality, and 'other' method. Allocation factors are not necessarily between 0 and 100%. They can be negative or greater than 100% as long as the sum of the set of allocation factors is 100%. In most cases allocation using 'other relationships' (according to the ISO three-step approach) is used.

The EcoInvent methodology avoids system expansion due to reservations on principle by the co-authors toward the approach. Owing to the fact that the avoided burden approach always attributes 100% of the avoided burdens to the product of interest, this approach is seen as an extreme case of the traditional allocation approach. As long as all avoided burdens are attributed to the multi-output process at issue, the authors feel that the avoided burden approach possibly leads to poorly balanced life cycle inventories.

4.1.5 CML operational guide to ISO

The CML guide recognizes allocation as one of the most sensitive issues in life cycle assessment methodology and gives the issue wide coverage [27–29]. While the guide notes that ISO standards provide an important international reference with respect to principles, framework and terminology for conducting and reporting life cycle assessment studies, the

[1] Sensitivity results, ranges and optional data sets corresponding to different approaches are more feasible for client- or product-specific studies. In the case of generic LCI data modules, which may be rolled up and separately presented at an aggregated level for a specific product, or may be used as input datasets for a product LCA, the use of ranges or alternatives could results in an unmanageable proliferation of cases.

standards do not provide step-by-step operational guidelines. Toward this end, economic allocation is advised as a baseline for all allocation in a detailed life cycle assessment. The guide addresses the various issues related to the use of economic information for generating inventory data and offers possible solutions. Unstable prices, for example, are compared to unstable emissions data. Most processes do not emit regularly over time, in that they show daily, weekly, monthly and seasonal variations and long-term trends. For processes, similar variations may occur due to the business cycle. The proposed solution is to use three consecutive annual averages, or use prices from futures markets.

4.1.6 The eLCie system

Sylvatica [30] summarized the results of a task performed for the International Design Center for the Environment (IDCE) to create a methodology for 'Streamlined Life Cycle Assessment'. IDCE commissioned this work to develop a sound methodology that it can apply in implementing its 'eLCie' system for identifying environmentally preferable building materials. The scope of the task included a review of relevant developments related to applying life cycle assessment (LCA) in identifying environmentally preferable products.

Sylvatica recognizes that the application of the ISO standards in addressing allocation still allows for flexibility on the issue, in two ways. First, the issue of 'feasibility', as determined by the practitioner (or the client), is raised. System expansion may require the gathering of additional data (e.g., on the life cycles of the avoided products). The budget allotted for the study may not allow this extra effort and may influence whether system expansion is feasible.

Second, the report raises the issue of 'fairness' and refers to the ISO standard which says that economic allocation can be used 'for example' but also allows for the use of mass or energy. In an effort to avoid 'gaming' of the modelling by each participant to suit his or her own interests, Sylvatica describes two ways to achieve consistency in allocation modelling. One is *ubiquity*: using the same method for allocation for all processes in all life cycle models. The other is *consistency-in-practice*: using the same method for the same kind of process in all life cycles being compared. Sylvatica sees consistency-in-practice as a way for the LCA community to move ahead in conducting LCAs that allow valid comparisons while engaging the full wealth of databases and practitioners. However, the consistency-in-practice approach is yet to be validated.

4.1.7 Avoiding allocation through system expansion

Weidema has published widely on how to avoid allocation in life cycle inventory by applying the system expansion approach [references 4 and 31 for example]. He has identified four obstacles to system expansion and why this option has not generally been accepted in *retrospective* life cycle assessments:

1. There is typically *no possibility for system expansion*. Retrospective studies typically seek to describe a status quo situation, in which there are no changes in production volume. This obviously excludes the possibility of system expansion, because an expansion involves balancing a change in output volume of a co-product in one system with an equivalent change in the other systems to be compared, in order to maintain comparable product outputs from the systems.
2. It has been *regarded as too difficult*, too uncertain, or even impossible to identify which processes are affected when balancing a change in demand for (or supply of) a specific co-product.
3. Because system expansion *may involve processes that also have multiple products*, it has been suggested that there are situations in which system expansion would be impossible because it would involve an unending regression.
4. When a *by-product does not substitute for another product*, system expansion may be regarded as incompatible with the requirement that compared systems must have identical functions [1].

Weidema's focus is on the use of life cycle assessment as a way to model as closely as possible all the possible external consequences of the potential change in demand for a product. Weidema concludes that allocation can (and shall) always be avoided in *prospective* assessments. His reasoning is that in order to correctly study the effects of a potential product substitution, it is necessary that the studied product systems are comparable, which means they must provide the same function, thus reflect the substitution that is really expected to take place. He continues by saying that since the ultimate goal of so-called non-comparative, retrospective life cycle assessments, such as product declarations and hot-spot identification, is to support the choice among several products, this choice should be based on environmental consequences and not on historical impact. Therefore, one might as well do a prospective life cycle assessment in the first place. In *retrospective* assessments, it is not possible to declare which allocation procedure to use, but avoiding allocation may still be an option [1].

Weidema proposes rules for treating the co-production issue in the form of four steps [1] which he later simplified to 3 [31].

Avoiding Co-Product Allocation [31]

Step 1) The co-producing process (and its exchanges) shall be ascribed fully (100%) to the determining co-product for this process (product A).

Step 2) Under the conditions that the dependent co-products are fully utilized in other processes, product A shall be credited for the processes that are displaced by the dependent co-products. The intermediate treatment shall be ascribed to product A. If there are differences between a dependent co-product and the product it displaces, and if these differences cause any changes in the further life cycles in which the co-product is used, these changes shall likewise be ascribed to product A.

When the conditions of Steps 1 and 2 are not fulfilled then Step 3 applies:

Step 3) When a dependent co-product is not utilized fully (i.e. when part of it must be regarded as a waste), the intermediate treatment shall be ascribed to product B, while product B is credited for the avoided waste treatment of the co-product.

Note that in order to perform Step 2, data that are specific to intermediate treatment are needed.

Ekvall has also addressed system expansion, which he compares to the ripples caused by throwing a stone in a lake [32]. He asserts that the propagation of consequences should be mapped down the cause-and-effect chains to where the effects are large enough to be significant for the assessment of the action. In order to provide a decision-maker with comprehensive information regarding the environmental impacts of a specific action, the study should focus on the activities for which the environmental impacts are most affected by the action. This is regardless of whether they are located within or outside the life cycle of the product investigated. Ekvall recognizes that such an approach requires a different focus than is typically used in a life cycle assessment study. Furthermore, he recognizes that the expanded system philosophy which he advocates is radically different than an effects-oriented life cycle assessment and should be considered as an entirely new approach.

Ekvall refers to early SETAC definitions which state that a life cycle assessment should provide information about the environmental burdens associated with a product, process or activity but are vague because the association – that is, the connection in the mind – between environmental burdens and the product being investigated is subjective. Different methodological boundaries, allocation methods, and/or data sources will be used by people with different associations.

Ekvall, along with Finnveden, explores the merits of the system expansion approach in a broader context in a later paper; they conclude that system expansion is possible in a wide range of life cycle assessment applications [33]. The authors suggest that other approaches to the allocation problem are adequate only where the effects on the inventory results are small. However, system expansion requires the collection of more data from additional activities and is often based on inaccurate data, or improper assumptions, for the effects on the external functions. Further research is needed to establish what data should be used in system expansion.

4.1.8 Open-loop recycling

Open-loop recycling is often regarded as a special case under co-product allocation, and is regarded as such by ISO 14041. The key distinction is that modelling of recycled material should reflect changes in the inherent properties of the material, i.e. declining quality. According to Kim et al. [34], the only way to avoid the allocation for open loop recycling is with system expansion. They assert that credit accrued by recycling and using recycled material should be considered in connection with the material quality. The application of the proposed method only requires data for the life cycle under study and information on the recycling process.

The handling of open-loop recycling appears to be especially significant to the building industry for entire buildings as well as for individual building products and materials. For example, Vögtlander et al. [35,36] propose an Eco-Costs – Value Ratio (EVR) model for economic allocation which they demonstrate in office and warehouse buildings applications.

Borg et al. [37] posit that economic value is the appropriate indicator for remaining quality of recycled materials:

$$L_{forward} = RF \times L_{vo \rightarrow vn} \times T_m / S_n$$

where:

$L_{forward}$ = environmental load into the future
RF = fraction available for recycling
$L_{vo \rightarrow vn}$ = all environmental loads beginning with product V_0 through the production chain to product V_n
T_m = market value for recycled material
S_n = market value for virgin raw material or intermediate product

Werner and Richter [38] applied economic allocation to aluminum window frames. It was decided that the environmental burdens should be allocated to only those products which are the 'aim' or the 'intended output' of a process. Therefore, release data were allocated entirely to the window.

Jungmeier et al. [39,40] address the treatment of allocation in wood-based products. They conclude that different allocation factors, e.g., mass or economic value, are allowable in the same LCA. They suggest the following allocation schemes that they consider to be the most practical: forestry – mass or volume; sawmill – mass or volume and proceeds; wood industry – mass and proceeds.

4.2 Industry-specific applications

4.2.1 Ammonia production

Kim and Overcash [41] explored three ways to define an industrial manufacturing process: a macroscopic, a microscopic, and a quasi-microscopic approach. The macroscopic approach does not subdivide any of the sub-processes within a plant. The microscopic approach fully separates all sub-processes while the quasi-microscopic approach allows for joint sub-processes that cannot be technically separated. The authors claim that although modelling at the sub-process level requires more information, it may not be very much more time or cost intensive. The three approaches were applied to a plant that co-produces ammonia and carbon dioxide. The quasi-microscopic and microscopic approaches, which model the emissions around the ammonia synthesis sub-process, gave similar results. The macroscopic approach, which splits emissions between the two co-products, resulted in higher results for most of the emissions (Fig. 5).

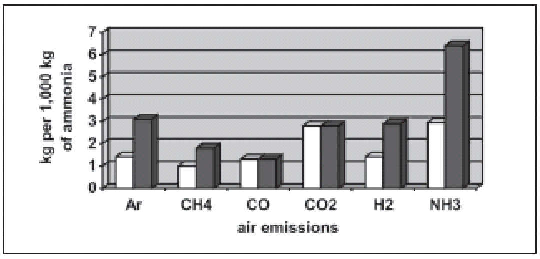


Fig. 5: Allocation of selected emissions from ammonia production using a macroscopic (macro) and a microscopic/quasi-microscopic (micro) approach [41]

Kim & Overcash assert that the use of the sub-division procedure minimizes the need for allocation, but it does not completely avoid it for background data. However, the authors propose that the solution is for every industrial sector to perform the allocation procedure by the quasi-microscopic/microscopic approach.

4.2.2 Borax production

Azapagic [2,42–44] explored the use of linear programming to find opportunities for system improvement in the production of 5 boron products (anhydrous borax (AB), boric acid (BA), anhydrous boric acid (ABA), 5 mol borate and 10 mol borate). Linear programming is a traditional system analysis approach that is used to help identify optimum operations by accounting for the complex interactions and exchanges between different parts of a system. When environmental releases are viewed as one of the exchanges, linear programming can be applied to co-product allocation. Azapagic's work deals with system analysis where incremental changes to a specific system, i.e. marginal changes, were the goal of the study, so that the linear representation of a system model is appropriate. Linear programming accommodates changes to the system and reflects changes in environmental burdens that results from these changes.

This work is based on the notion that it is not always obvious what kind of causality exists in a system. In order to identify the type of causality to be used as the basis for allocation the system operation must be well understood and *detailed data on the subprocesses in the system must be available*. Whichever method is used, Azapagic warns that it has to reflect the real behavior of the system under study.

The case study on the production of boron products was based on foreground data that were obtained from U.S. Borax and included the process data for extracting raw materials and making the boron products. The majority of the data for the background system were obtained from external (to borax production) databases. This included operations such as production of packaging, explosives, ammonia, and nitric acid. Linear programming was not applied to background data. The allocation method for these processes is not specified by Azapagic. The linear programming model of the boron products is composed of six submodels: mine and secondary crusher, primary process, boric acid, anhydrous boric acid, anhydrous borax, and packaging and shipping.

Azapagic compares the results of applying the marginal allocation approach with what she calls the 'arbitrary allocation methods' most commonly used in co-product systems. Fig. 6 shows the results for CO_2 emissions calculated on a marginal, aggregated mass (i.e. facility level) and disaggregated mass (i.e. sub-process level), market value and boric oxide (B_2O_3)-content basis.

Azapagic concludes that allocation methods are often chosen arbitrarily since they do not always reflect natural causality. While allocation by B_2O_3 content may be expected to increase the allocation coefficients as the degree of processing of the different products increases to give higher boron content, the increase is actually linearly-related to the content of B_2O_3 in the products and not to the burdens associated with their processing. The situation is different for mass-based allocation that is done using disaggregated data since disaggregation takes into account the differences in the processes used to make the different products. In that case, allocation on marginal and mass bases gives the same results. Therefore, in some cases it is correct to allocate on the basis of physical quantity as long as the choice is based on natural causation and is not an arbitrary one. A correct type of causality can only be identified if the system operation is well understood and detailed data on sub-processes are available.

4.2.3 Corn ethanol production

Kim and Dale [45] applied the system expansion approach to the production of ethanol from corn using dry and wet milling. The alternative product system was assumed to be soybean milling (i.e. soybean oil and soybean meal). Urea production for animal feed was also included in the analysis. It was assumed that these five products were mutually dependent in that the amount of one product produced influences the production of another product that meets the same need (for example, increased corn oil production results in reduced soybean oil consumption).

The underlying assumption in the system expansion approach is that product systems with an equivalent function have the *same environmental burdens*. Hence, the environmental burdens associated with ethanol from dry milling are assumed to be equivalent to those associated with ethanol from wet milling. A series of equations were generated to compute the correlations between competing co-products

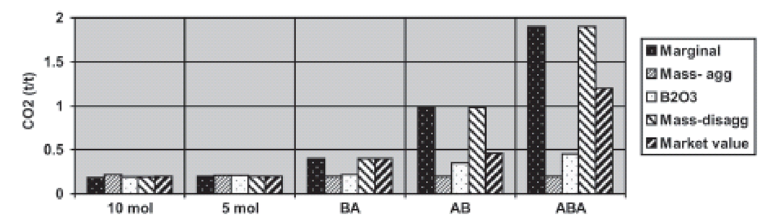


Fig. 6: Comparison of different allocation methods for CO_2 emissions from the production of five boron products [2]

related to how much one product is capable of displacing another product. For example, the displacement ratio of distillers dried grains (DDGS) to soybean meal, in mass, is 0.823.

$$E_{DDGS} = b_{DDGS/soybean\ meal}\ n\ E_{soybean\ meal} + {}_{DDGS/corn}\ n\ E_{corn}$$

where:

- $E_{soybean\ meal}$ = Environmental burdens associated with producing one kilogram of soybean meal
- E_{corn} = Environmental burdens associated with producing one kilogram of corn
- $b_{DDGS/soybean\ meal}$ = Displacement ratio between DDGS and soybean meal (Value is in mass of DDGS per mass of soybean meal, which is equal to 0.823)
- $b_{DDGS/corn}$ = Displacement ratio between DDGS and corn (value is in mass of DDGS per mass of corn, which is 1.077)

This exercise results in nine linear equations, including the environmental burdens associated with producing corn grain and urea. These equations are then solved simultaneously to calculate the environmental burdens per kilogram of the co-products (e.g., ethanol, DDGS, corn oil, soybean meal, etc.). Since all the co-products in this system match each other, the foreground system is a closed system. While the system expansion approach is used to avoid the allocation procedure in the foreground system of ethanol production from corn grain, traditional allocation is applied in the upstream processes, such as ammonia production and petroleum fuels. Although the study focused on net energy and greenhouse gas emissions, the authors suggest that this method is applicable to other environmental burdens.

To identify the effects of the choice of allocation procedure, allocation based on mass, market value, energy content, and a subdivision approach were investigated. Fig. 7 shows the impacts of the different approaches on the net energy calculations. Dry milling has a higher net energy in all allocation methods except in one case of system expansion. No significant differences in the final conclusion were found even though the values of emissions assigned to ethanol do vary with the allocation methods.

Kim and Dale conclude by saying, "The choice of the allocation procedure depends on the goal of the study. Each allocation method has its advantages and disadvantages. For instance, the system expansion approach can evaluate effects of changes in the foreground system, but is a data-intensive process. The mass basis allocation method is easily applicable and can identify the key sub-processes, but is unable to determine effects of key process parameter changes."

4.2.4 Petroleum refining

Recognizing that an aggregate approach at the refinery level is unable to account for energy use and emission differences associated with individual products (fuels), Wang, Lee, and Molburg set out to capture these differences by modelling at the sub-process level [46]. They studied 'well-to-pump' greenhouse gas emissions and energy use for four fuels (gasoline, diesel, LPG & naphtha) under five cases:

1. The energy content based allocation at the refinery level;
2. The energy content based allocation at the refinery level with rule-of-thumb adjustments;
3. The mass based allocation at the process level;
4. The energy content based allocation at the process level; and
5. The market-value based allocation at the refinery level.

The greenhouse gas patterns are similar to those of energy use (Fig. 8). While the refinery level results showed some variation, the three results at the process level are similar.

Wang, Lee, and Molburg conclude that process level allocation should be used in life cycle assessment since it can reveal additional energy and emission burdens associated with certain refinery products that would otherwise be overlooked.

4.2.5 Combined heat and power

Frischknecht emphasizes that one cannot determine whether a particular process will create an allocation problem, therefore allocation cannot be defined at the process level but must be done at the system level [47]. The author applies this thinking to a small-scale, gas-fuelled combined heat and power (CHP) plant. Inventory analysis for decision-making: scope dependent inventory system models, and context-specific, joint product allocation. It is postulated that companies can perform allocation in order to optimize the

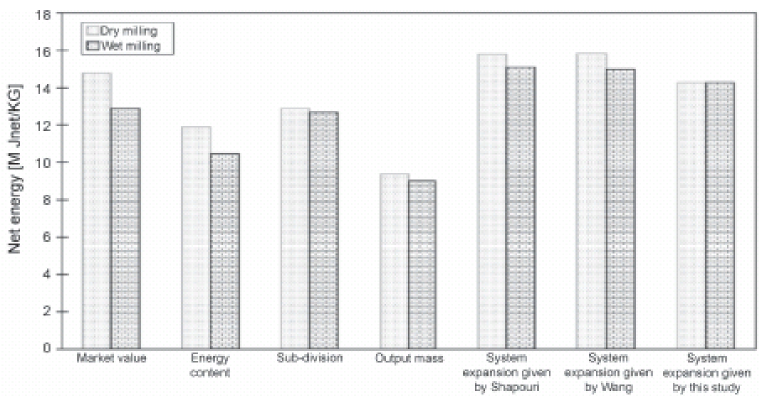


Fig. 7: Effect of the choice of allocation procedures on the net energy for ethanol produced from corn [45]

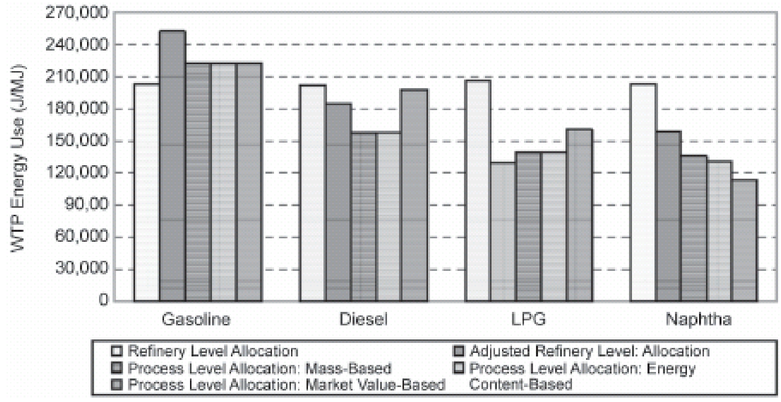


Fig. 8: Well-to-pump energy use for fuel production: Joules per MJ of fuel available at the pump [46]

economic and/or environmental performance. Frischknecht identifies the task at a hand as allocating costs *and* environmental impacts to the jointly produced goods in order to achieve performance that is superior to the competition.

System enlargement is viewed as a simply a special case of an allocation factor and suggests moving system expansion from Step 1 to Step 3 in ISO 14041 and use system expansion in a way similar to the use of economic and other causalities.[2] Furthermore, Frischknecht suggests that context be considered more frequently when carrying out joint product allocation since context highly influences both the procedure and the factors to be applied:

"The burdens of an alternative, single-output process solution of one coalition partner are subtracted from the burdens of the whole coalition of, e.g., two partners. The difference shows the minimum burdens to be attributed to the *second* partner (according to the '*incremental cost test*'). Allocating on the basis of this difference, the *first* partner will not profit at all because there is no difference at all between the stand-alone option (alternative single-output process) and a coalition. Similar considerations may be made when the burdens of an alternative single-output process of the *second* coalition partner are subtracted from the burdens of the coalition." In this case, the *second* partner would not profit from the coalition. Frischknecht concludes by saying that this is why the 'system expansion' – or 'avoided burden' – approach will most probably be applied in a context where one *individual* decision-maker can determine his or her allocation factors independently.

[2] This attitude is also reflected in the EcoInvent Guidance [26] of which Frischknecht was a co-author.

5 Summary of Findings

The literature search revealed that a fairly extensive effort has been put forth by the LCA community in addressing the allocation issue. However, past efforts to examine the effects of different allocation schemes in a comparative system-wide perspective in a systematic way have been limited. Previous work has either been done in order to develop generic one-size-fits-all procedural guidelines or focused on specific process or facility operations.

The progress to develop generic guidelines has lead to *mixed recommendations* for allocation schemes (Table 1). The EPA guidance manual recommends using a mass basis, GREET

Table 1: Bases Used for allocation in selected general guidelines for life cycle assessment methodology

	Mass	Energy	Economic	Flexible
US EPA 1993 [19]	States that no allocation basis is always applicable however, the guide endorses *mass* basis.			
GREET [24]		Follows a process where co-product value is measured by *energy* units		
NREL 2004 [22]			Follows the ISO hierarchy but recommends *economic* basis	
EcoInvent 2004 [26]				Avoids using system expansion and allows for *choice* of basis
CML Guide [23]			*Economic* allocation is advised for all detailed LCAs	
eLCIe 2004 [30]				Recognizes the ISO standard and the need to allow *flexibility*

uses an energy basis, while NREL and CML prefer economic allocation. The creators of EcoInvent oppose the use of system expansion and opt to provide a flexible system that allows users to apply whatever allocation scheme they prefer.

Although not everyone agrees, there is a general agreement that a *system expansion* approach is a very attractive way to avoid the difficult problem of allocation altogether. Expanding system boundaries also allows the modeler to assign credits for avoided environmental burdens associated with product displacement. A drawback to using system expansion is that the model gets larger and more complicated, and more data are needed to complete the model. Data accessibility as well as time and effort to collect this data become a significant issue, bringing the *practicality* of applying system expansion into question.

Also, larger systems run the risk of being *less transparent* in that there is more detail on how data were arrived at than can be conveyed conveniently. This has been the case in some studies when subtracted systems were used and the avoided function was not been adequately described and justified. It is obvious that system expansion is currently used as a way to *claim a credit* for some perceived avoided environmental burden. Regardless, the majority of those who favor the system expansion approach recognize that some sort of allocation is still needed in order to collect the necessary background data.

The 12 published studies that were uncovered in the literature search that address the various ways to approach allocation are summarized in Table 2. The case studies on particular industries (ammonia, borax, ethanol, petroleum refining, combined heat and power production) focused on the need to model *specific sub-processes* within a facility. The emphasis is on the need to supplement foreground data (i.e. the process data) with background data (i.e. data on input materials)[3]. The preference here also seems to be the use of a physical basis, such as mass measured in kilograms or a mole basis. Economic or market based allocation is often viewed to be too volatile to be practical. It is recognized that a lot of effort may be needed to obtain the background data, but a good solution for *data availability* is not offered.

The notion of *determining, or intended, product* was raised. The idea is to recognize when a process was created with the intent of producing a single main product of interest. By-products and wastes that are created as a result of manufacturing this main product are considered to be incidental, including those that may have found a market over the years. Therefore, all of the environmental burdens are placed on the main product.

Recycling, specifically *open-loop recycling*, is viewed as a special condition of allocation and is given special attention in the literature. The concern is to capture the downstream benefits that post-consumer recycling may incur. Building materials and wood-based products have received most of the attention in this area. Economic allocation seems to be the preferred approach and is perceived to be the best avenue to capture the downstream recycling activities. A number of allocation methods for open loop recycling are based on arguments about fairness, or accountability, so that environmental burden is appropriately assigned to the offending activity. However, it is difficult to determine which procedure is most 'fair' since this is a subjective term and depends on the perspective on the person conducting the study. After studying eight different perspectives and allocation procedures, each of which can be considered to be fair in their approach, Ekvall and Tillman [10] conclude that different criteria (for selecting an allocation procedure) are important for different LCA applications. They add that different allocation procedures will probably be appropriate for different LCA applications.

The notion of *incentive* was also raised. The idea is to create allocation schemes that assign minimal or no environmental burdens to byproducts that may be used in other processes as input materials, thereby encouraging recycling. Creating incentive to *encourage recycling or meet some other environmental objective* should be clearly established in the goal definition stage.

All the case studies and generic guidelines appear to be in search of a single approach to applying allocation in all cases. Norris questions this notion by presenting the *ubiquity* versus *consistency-in-practice* models. Norris suggests that perhaps the goal is to find the appropriate allocation basis for a particular process or type of process and apply it consistently in practice rather than trying to decide on one allocation method to always be used. Furthermore, the effect of data variability on the final results and the potential impact on decision-making is mentioned but nothing further was revealed.

[3] Foreground versus background data considerations featured prominently in the literature, but the description of a process or industry as being in the foreground or background is dependent on the interest of the person conducting the study. A product, such as ethanol, may be of main interest in one study but considered an industrial raw material input, and hence, a background process, for another.

Table 2: Allocation methods investigated by various LCA researchers

System Expansion	Open Loop Recycling	Sub-Process Level	Intended Product
General Principle: Weidema [1]	General Principle: Kim, Hwang & Lee [34]	Ammonia Production: Kim & Overcash [41]	General Principle: Weidema [31]
General Principle: Ekvall [32]	Building Products and Materials: Vögtlander, Brezet & Hendricks [35,36]	Borax Production: Azapagic [42-44]	Aluminum Window Frames: Werner & Richter [38]
Corn Ethanol Production (Soybean Oil and Meal Alternatives): Kim & Dale [45]	Recycled Materials: Borg, Paulsen & Trinius [37]	Petroleum Refining: Wang, Lee & Molburg [46]	
		Combined Heat & Power: Frischknecht [47]	

6 Discussion

At this time, there is no single, scientifically-based method for input and co-product allocation. The only way to allocate burdens is in the application of assumptions drawn from sound, transparent logic. Therefore, any conclusions that are drawn and any decisions made as a result of how the data are partitioned are themselves subject to the same assumptions. The key issues that pertain to allocation methodology are discussed in the following sections.

6.1 Data availability

Limited data availability is a real-world factor that continues to impact life cycle assessment methodology. Applications such as linear programming require access to sub-process input and output data. For system expansion, data that are representative of generic background processes are needed. Life cycle inventory databases must become more accessible and readily available to users if the life cycle assessment tool is to become widely adopted.

6.2 Matching goal definition with methodology

In the pursuit of consistent allocation methodology, the debate needs to be defined, at a minimum, along two lines: one that pursues 'what if' scenarios (referred to as consequential life cycle assessment) and 'what was' scenarios (attributional life cycle assessments). Although a clearly defined goal statement can help identify the appropriate method to follow, the ISO standards do not clearly delineate the dependence of allocation method on goal definition. In general, goal definition remains a critical step in the life cycle assessment process and is need of further refinement.

6.3 Modelling 'reality'

While all those who strive to model life cycle inventory want to do so in an 'intuitively reasonable' way, there is a clear split between those who prefer approaches that attempt to model 'reality' versus those who aim is to provide a practical, or 'fair', solution to a technical problem. However, the definition of reality or fairness is in the eye of the modeler. Some practitioners use LCI as general representations of industrial processes, not exact modelling of reality. Others strive to capture precise sub-process emissions. The identification of a determining product is considered to reflect both fairness and real life. Using reality or fairness as a basis for allocation alone is too ambiguous for practical purposes.

6.4 Impact on decision making

Future studies should investigate the overall impact on the entire system, instead of only focusing on the effects around individual processes. The case studies presented in this paper demonstrate how variations in allocation approaches can alter the results in the inventory. This is not entirely unexpected and is somewhat predictable. The question should not be *if* the results vary but by *how much*. The important consideration is the impact that any given choice for allocation has on the decision making process. This aspect was not seen in the literature that was reviewed. Clearly, the results will only be as accurate as the input data that are used in the modelling. Wide ranges of variability exist in any data that represent an average technology or process mix. This variability may overshadow any differences caused by various allocation techniques.

6.5 Creating a theoretical balance

The allocation principle within ISO 14041 (Section 6.5.2) known as the '100% rule' states that the sum of allocated inputs and outputs of a unit process shall equal the unallocated inputs and outputs. That is, exchanges should not be allocated twice or not at all. This principle is intended to be applied to allocation around a process but could have relevance in a system-wide perspective. Application at a higher view to two or more systems would create a theoretical balance across systems so that instances where burdens are being assigned to more than one system, or to none at all, can be flagged.

7 Conclusions

Allocation methodology for creating life cycle inventories has been frequently addressed, discussed and debated in the public domain over the years, yet the methodology continues to be in a state of flux. While it is clear that there are many ways in which allocation is being accomplished, no single method stands out as a general solution to the problem. ISO 14041 puts perspective on the issues but its one-size fits all framework is being challenged. It is clear that although the ISO standard identifies a general methodological framework for allocation, it does not provide specific guidance on when and how to apply the steps that are outlined in14041. The ISO standard should be expanded to provide more precise guidance in how to approach allocation. The guidance should be goal dependent, which also reflects the users' needs.

The literature search revealed many publications on allocation that provided either generic guidance (e.g., EPA, GREET, NREL, CML, and EcoInvent) for conducting allocation or demonstrated how allocation can be done in specific case studies (ammonia, borax, ethanol, petroleum refining, and combined heat and power). The generic guidance documents included mass, energy, economic basis for allocation as well as flexibility for the user to select an allocation basis. A number of allocation approaches have been proposed based on the notion of modelling reality. Other allocation methods are based on the argument of fairness, or accountability, especially in open loop recycling systems. What continues to be lacking is a unifying theory that can explain what allocation key is justifiable in any given situation. Also, the unifying theory should help LCA practitioners determine where system expansion is appropriate.

In the LCA community, there is growing consensus around two perspectives of LCA application: attributional, or accounting, LCA and consequential, or change-oriented, LCA. Attributional LCA's provide a more historical view of environmental impacts associated with a system while consequential LCA's try to answer the 'what would happen if...?' Guidance on how to apply allocation to these very different views is needed.

A frequently recurring theme is the idea that methodological choices in LCA depend on the goal of the study. However, guidance that matches goal with approach is still lacking. It would be useful to develop the range of allocation approaches matched with for use in different applications. They should be tested and demonstrated in various case studies and further discussed within the LCA community.

Disclaimer. The information in this draft paper was generated by the author through independent research as an employee of the US EPA but should not be construed as Agency policy. It has not been subjected to peer and administrative review, and has not been approved for publication. Use of this methodology does not imply EPA approval of the conclusions of any specific life cycle inventory or assessment.

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4.5 Findings from the Literature Review

The foregoing literature review revealed that previous work either 1. provided generic procedural guidance for conducting allocation or 2. demonstrated how allocation affects the outcome in specific process operations. Mass, energy and market value were used most frequently as the bases for allocation in order to study the effect on creating life cycle inventories. What continues to be lacking is a unifying theory that can explain what allocation approach is justifiable in any given situation. From the literature, the following five additional points stood out as significant considerations in allocation methodology.

4.5.1 Matching Approach to the Goal of the Study

A frequently recurring theme addressed in the literature is the idea that methodological choices in LCA depend on the goal of the study. However, guidance that matches the goal with the approach is still lacking.

4.5.2 Identifying a Demand Product

It has been proposed that all of the environmental burdens should be placed on the main product, i.e. the *raison d'être* for the process. By-products and wastes that are created as a result of manufacturing this main product are considered to be incidental, including those for which a market has been found over the years. This approach also avoids the difficulty associated with by-products that are sold only when the market is favorable (but otherwise counted as waste). A methodology for identifying a clear demand product may be lacking in these cases.

4.5.3 Accounting for Recycling

Recycling, specifically open-loop recycling is viewed as a special condition of allocation and is given special attention in the literature. The concern is to capture the downstream costs and benefits that post-consumer recycling may incur. Economic allocation seems to be the preferred approach and is perceived to be the best avenue to capture the downstream recycling activities. A number of allocation methods for open loop recycling are based on arguments about fairness, or accountability, so that environmental burden is appropriately assigned to the offending activity. However, it is difficult to determine which procedure is most "fair" since this is a subjective term and depends on the perspective of the person conducting the study.

4.5.4 Creating Incentives to Recycle

Allocation may assign minimal or no environmental burdens to by-products that are used in other processes as input materials, thereby, encouraging recycling. That is, a manufacturer may see an advantage in using an input material that has no environmental burden assigned to it. Furthermore, when a waste becomes categorized as a by-product, the producer benefits from being able to record a smaller quantity of generated waste. Creating incentive to encourage recycling or meet some other environmental objective should be clearly established in the goal definition stage.

4.5.5 Applying Process-Specific Allocation

Authors of the case studies and generic guidelines appear to be in search of a single approach to applying allocation in all cases. An alternative view suggests that perhaps, the goal should be to find the appropriate allocation basis for a particular process or type of process and apply it consistently in practice rather than trying to decide on one allocation method to use in all cases. This is most likely the intent behind Step 2 of the ISO guidance on allocation, which states, "Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way which relect the underlying physical relationships between them."

4.6 Conclusions

Although allocation methodology has been frequently discussed, and debated, in recent years, the search continues for a single, preferred approach or an underlying theory that determines which approach is preferable in a given situation. While questions remain on the specifics of how to conduct some allocation techniques, such as how to identify a 'demand' product or classifying materials as wastes versus by-products, the most pressing issue appears to be the need for guidelines to match the most appropriate allocation approach to the goal of the study.

This literature review also makes it clear that while many authors have written about the allocation issue, no one has looked at the impact that allocation has on an entire product system. To that end, this thesis author conducted research to evaluate the effect that changing the basis of co-product allocation has on the results when comparing two products at a system level. The results of that study are presented in the Chapter 5.

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5 Co-Product Allocation

5.1 Chapter Overview

This chapter presents the research approach and findings from a test of two alternative product (fuel) systems that was conducted to study the system-wide effects of varying the method of co-product allocation. Sections 5.2 through 5.4 present the details of the methodology that was used in the case study. Section 5.5 presents the findings and results of the research; it consists of a paper, entitled "Studying the Effect on System Preference by Varying Co-Product Allocation in Creating Life Cycle Inventory," which was published in *Environmental Science & Technology 41(20).* A concluding discussion on the topic of allocation is presented in Section 5.6.

5.2 Testing the Effects of Allocation across Systems

The research was designed to investigate the overall impacts of various allocation schemes across an entire system, instead of only focusing on the effects around individual processes (Curran 2007). The important consideration was on the impact that choices for allocation have upon the total results of an LCA including the impact assessment phase. This research was performed to determine what impact the selection of a basis for co-product allocation has on the total LCA when comparing two or more systems.

5.3 The Research Approach

The study involved creating a life cycle inventory for a baseline system (using Excel spreadsheets) then running the inventory through LCIA models to generate impact indicators. These steps were repeated to create scores using different allocation schemes.

First, conventional gasoline was analyzed with a reference flow of 1,000 gallons (3,785 liters), then an alternative product, gasoline blended with 8.7% ethanol by volume, was analyzed based on an equivalent functional unit (this volume was calculated from the U.S. federally-mandated oxygen content required in automotive fuel). Ethanol has a lower energy value; hence, more gasoline-blended fuel is needed:

Conventional Gasoline with MTBE:	20.22 mpg	(8.59 kmpl)	1,000 gallons (3,785 l)
Gasoline with Ethanol:	19.95 mpg	(8.48 kmpl)	1,014 gallons (3,838 l)

These product systems were selected because inventory data were readily available from a research study conducted by the U.S. EPA (these data have not been peer-reviewed and should not be construed as representative for the industry). This in-house study was designed to compare the production and use of fuels to power a typical passenger vehicle for 12,000 miles (19,312 kilometers), the typical distance travelled in the U.S. in one year. Since this study on allocation modeling does not depend on typical use, the amount of fuel used as the reference flow for the LCI was changed to 1,000 gallons (3,785 liters).

Due to the similarities of the conventional gasoline and gasoline with ethanol (i.e. the major component of both products is gasoline), a second case study was conducted to compare conventional gasoline with E85 (85% ethanol). The spreadsheets that were used to model the data for the gasoline with ethanol additive were modified to reflect a fuel mix of 85% ethanol by volume, and the reference flow was adjusted to 1,380 gallons (5,224 liters) to equal the energy content of 1,000 gallons of conventional gasoline.

E85 Ethanol Fuel:	14.65 mpg	(6.23 kmpl)	1,380 gallons (5,224 l)

The fuel efficiencies were derived by the EPA MOBILE 6 model for generic passenger automobiles. Figures 5-1 and 5-2 depict the system boundaries and the processes that were included in the assessment.

Figure 5-1 Life cycle boundaries and flows of conventional gasoline

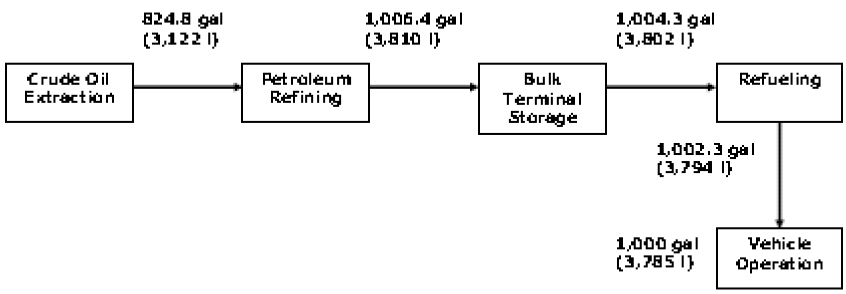
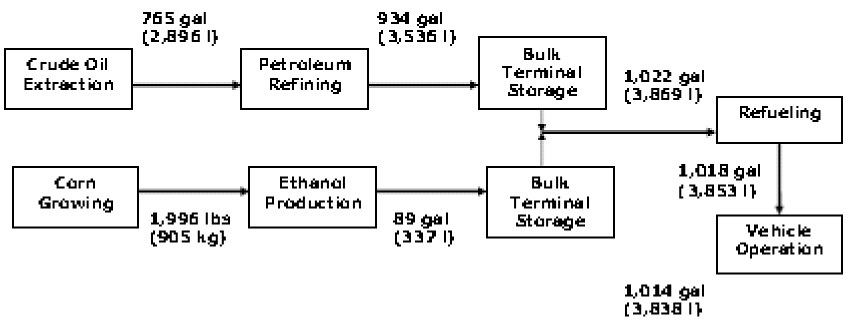


Figure 5-2 Life cycle boundaries and flows of gasoline with ethanol additive



The points in the fuel systems where allocation was applied include petroleum refining and corn milling (i.e. ethanol production). Corn was considered to be the only product from the field; the stover is considered to be a waste material but its use as a feedstock for ethanol may be addressed in future trials. Inventory spreadsheets were created using the following commonly-used bases for allocation: Weight, Volume, Market Value, and Energy. In addition, for corn milling, ethanol was identified as the main product for which mills are constructed. That is, the main purpose of a corn mill is to produce ethanol, considered the 'demand product,' and not to produce the other by-products (such as distillers dried grains, DDGS). Hence 100% of the air and water releases were assigned to ethanol as a fifth basis for allocation (see Table 5-1).

Table 5-1 Percentage of air and water releases assigned to gasoline from petroleum refining and to ethanol from corn milling (wet and dry) using weight, volume, market value, energy, and demand-based allocation

	Basis for Allocation				
	Weight	Volume	Market Value	Energy	Demand
Petroleum Refining: Gasoline	43.7%	46.5%	55.1%	44.3%	N/A
Corn Milling (Wet): Ethanol	48%	38%	70%	57%	100%
Corn Milling (Dry): Ethanol	49%	25%	76%	61%	100%

5.3.1 Data Reduction Procedures

The sources of the secondary data were publicly-available databases, such as the EPA's Toxics Release Inventory (TRI) and the Energy Information Administration (EIA). The inventory data spreadsheets that were compiled using these public databases were not externally reviewed nor was additional input from industry obtained. While an attempt was made to create *plausible* data, they should not be construed as *actual* release data. The assumption was made that the data themselves are inconsequential for the purpose of this study as long as the same data were used when comparing the different modeling schemes.

Much of the public data are presented at an aggregated level for an industry or at the facility level which does not allow modeling at the process level. Furthermore, the application is assumed to be at the national (or regional) level for which averaged data are appropriate. The data represent historical operations, i.e. what the releases were in a past period, and do not aim to optimize the system. The data that were used for the life cycle inventory were reported only in U.S. standard units, such as pounds (lbs).

This research required the creation of various spreadsheets containing life cycle inventory data in .xls and .csv format. The following procedure was followed to ensure that data were transcribed and recorded accurately.

5.3.2 Create Inventory Spreadsheets

The air and water emissions from the original summary inventory spreadsheets were cut and pasted into a new .xls file and labelled in a consistent format for easy identification (for example, CG_wb.xls for conventional gasoline using weight-based allocation). Both the original summary spreadsheet and the newly created spreadsheet with air and water emissions were viewed and compared side-by-side to ensure no data were omitted.

5.3.3 Format the Spreadsheets to be Compatible with TRACI

Since the EPA's Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) was used for calculating impact potentials (Bare, Norris et al. 2003), the .xls files with the air and water emissions data had to be in a format that is compatible for importing into the impact models. For example, specific column headings for Chemical, CAS Number, Media, Quantity (QTY), and Unit of Measure (UOM) are required. The files were saved in comma separated value (csv) file format and labelled consistently for easy identification (for example, CG_wb_air&water.csv).

5.3.4 Import Data into TRACI and Run Characterization

The .csv files were imported into TRACI which then calculated the scores for nine impact categories (acidification, ecotoxicity, eutrophication, global warming, human health cancer, human health criteria, human health noncancer, ozone depletion, and photochemical smog). The scores, along with the inventory used to calculate the scores, are exported by TRACI and saved as .xls files. The files were labelled consistently for easy identification (for example, CG_wb_chara.xls).

5.3.5 Data Validation Procedures

The importing of the LCI results into TRACI required the development of intermediate spreadsheets within the LCI workbook files (a workbook consists of a series of related spreadsheets) and adding direct spreadsheets to database import capabilities within TRACI. Data were entered at the system level to assess total impact of varying allocation methods on the product system.

Quality checks were accomplished manually by verifying that all processes, identified emissions and resources were present and consistent with the original LCI data set for each scenario. The manual quality check helped to verify that all processes, identified emissions and resource use elements had been accurately imported to TRACI.

Tracking sheets were used to guide the generation of the spreadsheets as well as to track the process of validating the entries in the converted spreadsheets.

5.4 Impact Assessment

Impact assessment of air emissions and water effluents was conducted across nine categories: acidification, ecotoxicity, eutrophication, human health (including cancer, non-cancer, and criteria pollutants), global warming, ozone depletion, and smog formation using TRACI (Bare, Norris et al. 2003). While the impact data included in TRACI is limited and cannot model all the inventory data that are imported, the assumption was made for this study that the consistent use of the same impact modeling method is adequate for identifying system preferability. Following are brief descriptions of each impact category, and how the indicators are calculated.

5.4.1 Acidification

Acidification can change the environment for terrestrial and freshwater systems and can damage plant and animal populations, buildings, and other structures. Acid deposition can occur through wet (e.g., rain, snow, sleet), dry, or cloud water deposition (e.g., fog). The acidification model uses the results of an empirically calibrated atmospheric chemistry and transport model to estimate total North American terrestrial deposition as a function of the emissions location. The potential for emissions to cause acidification is measured in H+ mole equivalents per kilogram of emissions. Because of the lack of a regional database of receiving environment sensitivities, modeling stops at acid deposition and does not include the buffering capacity or the sensitivity of the receiving environment. Factors are available for each State, but only a national average characterization factor is used.

5.4.2 Ecotoxicity

Ecotoxicity characterization provides a relative prediction of harm to plant and animal life following individual chemical releases to the environment. The CalTOX model provides multimedia transport and transformation of chemicals, and then couples these results with a toxicity factor (based on species sensitivity distributions of no observable effect concentration values) to account for harmful effects. Emissions to air and surface water are considered separately for terrestrial and aquatic ecosystems and the final ecotoxicity score is a combination of the air and water emission scores. The reference chemical selected for comparison is 2,4-Dichlorophenoxyacetic acid (2,4-D), and thus the units of the ecotoxicity potentials are expressed in kilograms of 2,4-D equivalents per kilogram of emissions. Obviously, this model is a very simplified version of what might be done with a smaller number of chemicals, a more specific location of study, and specific toxicity data on the species of highest concern for the location.

5.4.3 Eutrophication

Eutrophication is the fertilization of surface waters which leads to a change in the aquatic environment. When a previously limiting nutrient becomes more abundantly available this will often lead to an increase in the aquatic plant growth followed by a chain of other events including fish death, decreased biodiversity, and foul odor and taste. Further effects could include the production of chemical compounds in quantities toxic to humans, marine animals or livestock. The limiting nutrient is often phosphorus for freshwater systems and nitrogen for estuaries and coastal waters, and thus the location of the release often makes a significant impact on the relative potential for damage. The calculation of the eutrophication potential considers: 1. the relative strength or potency of the nutrient when it is released to an environment, 2. the availability of the nutrient, and 3. the probability that the nutrient is released to an area where it is the limiting nutrient. The units of the eutrophication potential are given in kilogram nitrogen (N) equivalents.

5.4.4 Global Warming

The potential effects that may occur with increasing concentrations of "greenhouse gases" which trap heat in the earth's atmosphere include increased droughts, floods, loss of polar ice caps, sea-level rise, soil moisture loss, forest loss, change in wind and ocean patterns, and changes in agricultural production. The intent of global warming characterization is to estimate the relative contribution that various greenhouse gases may have on climate change. The model uses the global warming potentials developed by the International Panel on Climate Change (IPCC) for the calculation of potency relative to carbon dioxide (CO_2). The 100-year time frame is consistent with the time horizons used by the U.S. for reporting.

5.4.5 Human Health Cancer

The intent of the human health cancer characterization is not to provide an absolute value of risk, but to provide relative comparisons of a large number of chemicals which may have the potential to contribute to cancer. The focus of this category is not on the localized use of chemicals within a work environment, (e.g., industrial hygiene), but the long-term exposures to chemicals in the regional and global environment. CalTOX is the base model for calculations within human health cancer. Human toxicity potentials for cancer are calculated using multimedia modeling, twenty-three human exposure pathways to estimate the dose per unit release rate, and a carcinogenic risk potency factor to determine relative potency. Benzene (C_6H_6) is used as the reference chemical.

5.4.6 Human Health Criteria

Ambient concentrations of criteria pollutants have been strongly associated with increased rates of chronic and acute respiratory symptoms and increased mortality rates. Pollutants within this impact category include sulphur dioxide (which leads to secondary particulate sulphate), nitrogen oxides (which leads to secondary particulate nitrate), particulate matter, PM less than 2.5 micrometers in diameter, and PM less than 10 micrometers in diameter. The model uses the output of atmospheric transport models to estimate the change in exposure to particulates coupled with concentration-response functions based on epidemiological studies to estimate mortality rates and a variety of morbidity effects. These effects are then combined into a unit of disability-adjusted-life-years (DALYs).

5.4.7 Human Health Noncancer

The intent of the human health noncancer characterization is not to provide an absolute value of risk, but to provide relative comparisons of a large number of chemicals which may have the potential to contribute to noncancerous effects. (Pollutants which were within the human health criteria category are not included within this category.) The focus of this category is not on the localized use of chemicals within a work environment, (e.g., industrial hygiene), but the long-term exposures to chemicals in the regional and global environment. CalTOX is the base model for calculations within human health noncancer. Human toxicity potentials for noncancer are calculated using multimedia modeling, twenty-three human exposure pathways to estimate the dose per unit release rate, and a reference dose or reference concentration to determine relative potency. Toluene (C_7H_8) is used as the reference chemical.

5.4.8 Ozone Depletion

Stratospheric ozone depletion is the reduction of the protective ozone layer within the stratosphere caused by the emissions of ozone-depleting substances. Secondary effects can include skin cancer, cataracts, and negative impacts on crops, materials, and marine life. The model adopts the ozone depletion potentials published in the Handbook for the International Treaties for the Protection of the Ozone Layer where chemical scores are based on trichlorofluoromethane (CFC-11) as the reference compound. The model takes into account the expected lifetime, transport, and potency of the compounds.

5.4.9 Photochemical Smog Formation

Ozone is formed within the troposphere from a variety of chemicals including nitrogen oxides, carbon monoxide, methane, and other volatile organic compounds in the presence of high temperatures and sunlight. High concentrations of ozone led to negative impacts on human health and the environment. The model characterizes ozone formation rather than modeling human and environmental endpoints, and the photochemical smog formation potential is expressed in nitrogen oxide (NOx) equivalents. The model utilizes the maximum incremental reactivity calculations (when VOC influence is at its maximum) of` West Carter at the University of California Riverside to provide state specific models which estimate the likelihood of ozone formation.

5.5 Paper VII - Studying the Effect on System Preference by Varying Co-Product Allocation in Creating Life Cycle Inventory

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Environ. Sci. Technol. **2007**, *41*, 7145–7151

Studying the Effect on System Preference by Varying Coproduct Allocation in Creating Life-Cycle Inventory

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How one models the input and output data for a life-cycle assessment (LCA) can greatly affect the results. Although much attention has been paid to allocation methodology by researchers in the field, specific guidance is still lacking. Earlier research focused on the effects of applying various allocation schemes to industrial processes when creating life-cycle inventories. To determine the impact of different allocation approaches upon product choice, this study evaluated the gas- and water-phase emissions during the production, distribution, and use of three hypothetical fuel systems (data that represent conventional gasoline and gasoline with 8.7 and 85% ethanol were used as the basis for modeling). This paper presents an explanation of the allocation issue and the results from testing various allocation schemes (weight, volume, market value, energy, and demand-based) when viewed across the entire system. Impact indicators for global warming, ozone depletion, and human health noncancer (water impact) were lower for the ethanol-containing fuels, while impact indicators for acidification, ecotoxicity, eutrophication, human health criteria, and photochemical smog were lower for conventional gasoline (impacts for the water-related human health cancer category showed mixed results). The relative ranking of conventional gasoline in relation to the ethanol-containing fuels was consistent in all instances, suggesting that, in this case study, the choice of allocation methodology had no impact on indicating which fuel has lower environmental impacts.

Introduction

Conducting a life cycle assessment (LCA) begins with the collection and creation of a life cycle inventory (i.e., the input and output data for all the industrial processes that comprise a product system). To do this, the resource inputs and environmental outputs for each industrial process and technology must be modeled. How this modeling is done can vary widely depending on the underlying assumptions that are used to calculate the inputs and outputs. In particular, the choice of an allocation approach for processes that produce more than one coproduct can have a profound effect on the results of the data that are entered into the inventory.

While a mass basis seems to be the preferred approach, other methods, such as market-value, volume, or energy, can also be used as a basis for allocation. While it is clear that there are many ways in which allocation can be done, no single method stands out as a general solution for how it should be done. The diversity of allocation approaches has caused much debate among LCA practitioners and researchers.

This researcher investigated the overall impacts of various allocation schemes across an entire system. The important consideration is the impact that any choice for allocation has on the decision making process. Wide ranges of variability exist in any data that represent an average technology or process mix. This variability may overshadow differences caused by various allocation techniques.

This paper begins with an explanation of what allocation is and why it is an important issue in life cycle assessment methodology. The initial results from testing various allocation schemes and evaluating their impacts across an entire system follow.

Background on Allocation Methodology. Allocation is the partitioning or assignment of material inputs and environmental releases when more than one activity is involved in the operation being studied. This paper focuses on the issue of allocating aggregated emissions data over a process. Historically, a simple mass-based approach has been the most widely used solution to coproduct allocation. Figure 1 shows an example of using a mass-based approach to calculate the portion of input materials and emissions inventory that would be assigned to the production of 1000 pounds of product A or of 500 pounds of product B (*1*).

A simple mass allocation method frequently gives reasonable results, but not always. A mass-based approach may seem impractical in cases where one coproduct far outweighs another. For example, mining produces a small amount of ore or mineral along with a much larger quantity of mined waste, which can be used as roadbed material. It might seem unreasonable to assign the majority of the environmental impact to the material that is used for roadbeds and not to the desired mined product. These kinds of results have led several researchers to delve further into understanding how much impact the choice of allocation methodology has on the resulting inventory.

A literature review (*2*) revealed many publications that address allocation and provide either generic guidance for conducting allocation or demonstrate how allocation can be done in specific case studies. The generic guidance documents promote the use of different approaches. The U.S. EPA guidance (*3*) follows a mass-based approach, while the GREET model (*4*) employs an energy-based approach. An economic basis for allocation is recommended in the CML guide (*5*). The EcoInvent database provides flexibility for the user to select an allocation basis (*6*). These choices appear to be based on each modeler's preference, presumably based on their experience and background, rather than on some type of scientific proof.

Published case studies have often focused on specific facilities or industrial operations to which different allocation schemes were applied in side-by-side comparisons. Production of ammonia (*7*), borax (*8*), ethanol (*9*), petroleum refining (*10*), and combined heat and power (*11*) are examples of case studies that were reported. A typical example, seen in Figure 2, shows how the calculated amount of apparent net energy needed to produce ethanol from corn can vary.

Other discussion on allocation methodology uncovered by the literature survey is based on the argument of fairness, or accountability, especially in open-loop recycling systems (*12*). Although the body of literature on allocation methodology is significant, most efforts focused primarily on the examination of the effects of different allocation schemes

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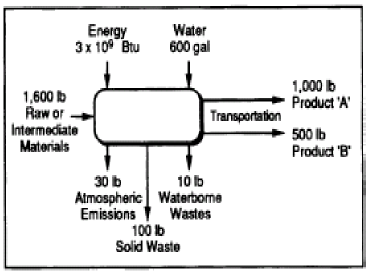
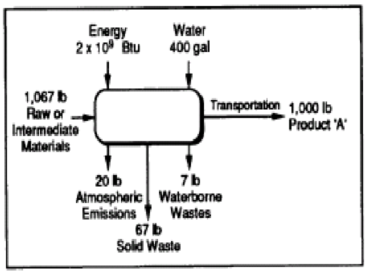
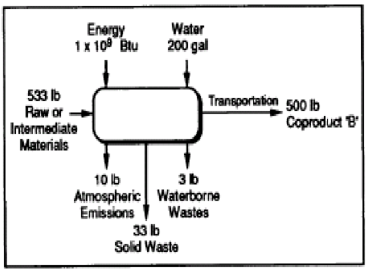


Co-Production of Product "A" and Product "B"



Allocation of Inputs and Emissions for Product "A"



Allocation of Inputs and Emissions for Product "B"

FIGURE 1. Allocating inputs and emissions on a mass basis for two coproducts (*1*).

and the resultant impact at the process level. These studies fail to identify a single, logically defensible approach for allocation. What continues to be lacking is a unifying theory that can explain what allocation key is justifiable in any given situation.

Purpose of the Current Research

This research was based on the hypothesis that the selection of a basis for coproduct allocation has no impact on the total LCA when comparing two or more systems as long as the chosen basis is applied consistently. That is, the relative ranking of two products, for example, product A is preferable to product B, will be the same regardless of the allocation methodology that is chosen.

Approach

The functional unit for the study was set at 1000 gallons (3785 liters) of conventional gasoline. The research involved creating a life-cycle inventory for the conventional gasoline system (using Excel spreadsheets) and then running the inventory through life-cycle impact assessment models to generate impact indicators. These steps were repeated to create scores using different allocation schemes.

These steps were then repeated for an alternative product, gasoline blended with 8.7% ethanol by volume, based on an equivalent functional unit (this volume was calculated from the mandated oxygen content required in fuel). Ethanol has a lower energy value; hence, data are first needed for the "normal" gasoline and then for the fuel with ethanol added which has a lower fuel economy

conventional gasoline with MTBE =
20.22 mpg (8.59 km/L) 1000 gallons (3785 L)

gasoline with ethanol =
19.95 mpg (8.48 km/L) 1014 gallons (3838 L)

Because of the similarities of the conventional gasoline and gasoline with ethanol, a second case study was conducted to compare conventional gasoline with E85 (85% ethanol). The spreadsheets that were used to model the data for the gasoline with ethanol additive were modified to reflect a fuel mix of 85% ethanol by volume, and the reference flow was adjusted to 1380 gallons (5224 liters) to equal 1000 gallons of conventional gasoline.

E85 ethanol fuel =
14.65 mpg (6.23 km/L) 1380 gallons (5224 L)

The fuel efficiencies were derived by the EPA MOBILE 6 model for generic passenger automobiles. Figures 3 and 4 depict the system boundaries and the processes that were included in the assessment.

The points in the fuel systems where allocation was applied include petroleum refining and corn milling (i.e., ethanol production). Corn was considered to be the only product from the field; the stover, that is, the leaves and stalks of corn, is considered to be a waste material, but its use as a feedstock for ethanol may be addressed in future trials. Five sets of inventory spreadsheets were created using the following commonly used bases for allocation: mass, volume, energy, and market value. In addition, for corn milling, ethanol was identified as the main product for which mills are constructed. That is, the main purpose of a corn mill is to produce ethanol, considered the "demand product", and not to produce the other byproducts. Hence 100% of the gas- and water-phase emissions were assigned to ethanol.

The sources of the secondary data were publicly available databases, such as the EPA's Toxics Release Inventory (TRI). Such data are aggregated at the facility-level data and do not allow for modeling at the process level. Furthermore, the application is assumed to be at the national (or regional) level for which averaged data are appropriate. The data represent historical operations, that is, what the releases were in a past period and do not aim to optimize the system.

The data that were used for the life-cycle inventory were reported only in U.S. standard units, such as pounds (lbs). The inventory data have not been externally reviewed nor was input from industry obtained. While an attempt was made to create plausible data, they should not be construed as actual release data. The assumption was made that the data themselves are inconsequential for the purpose of this study as long as the same data are used as the basis for comparing the different modeling schemes.

The EPA's Tool for the Reduction and Assessment of Chemical and other Impact (TRACI) was used for calculating impact potentials. The inventory data were converted into .csv files which were imported into TRACI to calculate the scores for nine impact categories (acidification, ecotoxicity, eutrophication, global warming, human health cancer, human health criteria, human health noncancer, ozone depletion, and photochemical smog).

Methodology for Five Allocation Schemes

The following section describes the calculations and data sources that were used to model petroleum refining and corn milling using the five allocation schemes (weight-based,

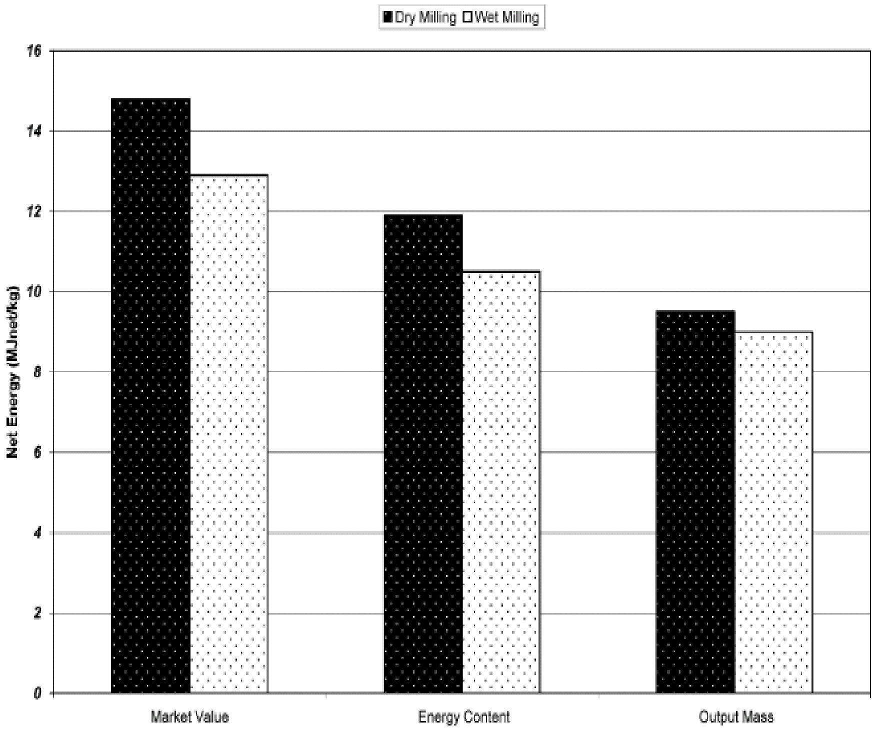


FIGURE 2. Effect of the choice of allocation procedures on the net energy for ethanol produced from corn (*9*).

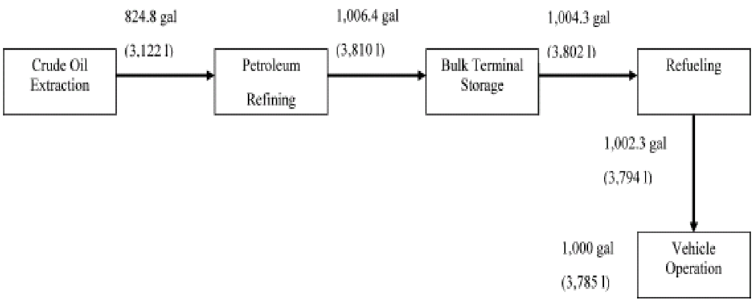


FIGURE 3. Life cycle boundaries and flows of conventional gasoline.

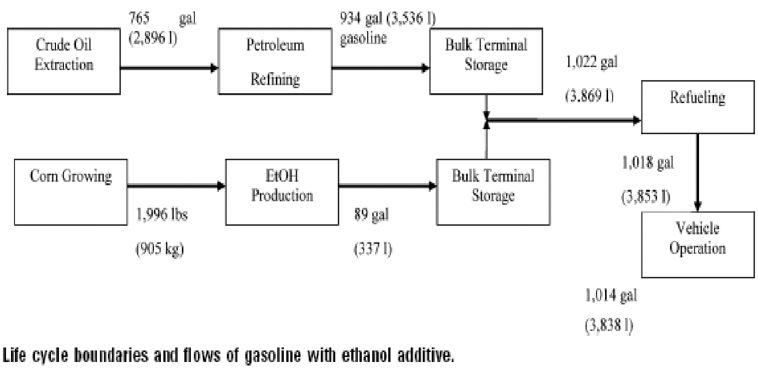


FIGURE 4. Life cycle boundaries and flows of gasoline with ethanol additive.

volume-based, market value-based, energy-based, and demand-based). These models are also shown in equation form in Tables 1(for petroleum refining) and 2 (for corn milling). Table 3 indicates the percentage of the gas- and water-phase emissions that was assigned to gasoline from petroleum refining and to ethanol from wet and dry corn milling. Ethanol produced by corn milling was modeled as 60% via wet milling and 40% via dry milling.

Methodology for Weight-Based Allocation. Petroleum Refining. Gas- and water-phase emissions from petroleum

TABLE 1. Calculating Allocation for Petroleum Refinery Products

allocation method	calculation procedure
weight	$R_W = E/T\eta Q_{ref}\rho$ R_W = release to air or water (lb) E = emission or effluent from the Petroleum Refining Industry in 1999 (lb) T = total petroleum refinery products in 1999 (bbl) η = conversion (lb/bbl) Q_{ref} = reference flow (gal) ρ = density of gasoline (lb/gal)
volume	$R_V = E/T\eta Q_{ref}$ R_V = release to air or water (lb) E = emission or effluent from the Petroleum Refining Industry in 1999 (lb) T = total petroleum refinery products in 1999 (bbl) η = conversion (lb/bbl) Q_{ref} = reference flow (gal)
market value	$R_M = E/TQ_{ref}s$ R_M = release to air or water (lb) E = emission or effluent from the Petroleum Refining Industry in 1999 (lb) T = total petroleum refinery products sales in 1999 (bbl) Q_{ref} = reference flow (gal) s = sale price of gasoline from refineries ($/gal)
energy	$R_E = E/T\eta Q_{ref}e$ R_E = release to air or water (lb) E = emission or effluent from the Petroleum Refining Industry in 1999 (lb) T = total petroleum refinery products in 1999 (bbl) η = conversion (lb/bbl) Q_{ref} = reference flow (gal) e = energy value of produced gasoline (Btu/gal)
demand	not applicable

refining were modeled as pounds of emission per pound of product from petroleum refining, SIC 2911. Emissions from the petroleum refining sector, using EPA sector notebook (*13*) and TRI data, were divided by the total mass of refinery products produced in 1999 (*14*), then multiplied by the reference flow, gallons of conventional gasoline, which was converted to weight by using an average density of gasoline, 6.15 lb/gal.

Corn Milling (Wet). Wet milling yields 31.5 pounds of starch with corn being processed into 33 pounds of sweetener or 2.5 gallons of ethanol. Also, gluten feed, gluten meal, and corn oil are produced. Ethanol is 48% of the output by weight. Gas- and water-phase emissions from wet corn milling were modeled as pounds of emission per pound of milled product. Total production in 1999, reported for 11 ethanol plants, was estimated by taking the total capacity for all wet mills (TRI) and adjusting for an average utilization rate of 77%. Emissions from wet mill ethanol production (TRI) were divided by the adjusted annual production volume and then multiplied by the reference flow for wet milling, in gallons, and by 48% to allocate to the weight of the ethanol (*15*).

Dry Milling. Dry milling yields 17.5 lb/bushel distillers dried grains (DDGS), as well as ethanol. Ethanol is 49% of the output by weight. Gas- and water-phase emissions from corn milling were modeled as pounds of emission per pound of milled product. Total production in 1999, reported for 21 ethanol plants, was estimated by taking the total capacity for all dry mills (TRI) and adjusting for an average utilization rate of 77%. Emissions from dry mill ethanol plants (TRI) were divided by the total adjusted annual production volume and then multiplied by the reference flow for dry milling, in gallons, and by 49% to allocate to the weight of the ethanol (*15*).

Methodology for Volume-Based Allocation. Petroleum Refining. Gas- and water-phase emissions from petroleum

TABLE 2. Calculating Allocation for Corn Mill Products

allocation method	calculation procedure
weight	$R_W = E/C\mu Q_{ref}w$ R_W = release to air or water (lb) E = emission or effluent from the Corn Milling Industry in 1999 (lb) C = total corn milling capacity in 1999 (gal) μ = utilization U.S. average (%) Q_{ref} = reference flow (gal) w = weight proportion of produced ethanol (%)
volume	$R_V = E/C\mu Q_{ref}v$ R_V = release to air or water (lb) E = emission or effluent from the Corn Milling Industry in 1999 (lb) C = total corn milling capacity in 1999 (gal) μ = utilization U.S. average (%) Q_{ref} = reference flow (gal) v = volume proportion of produced ethanol (%)
market value	$R_M = E/C\mu Q_{ref}m$ R_M = release to air or water (lb) E = emission or effluent from the Corn Milling Industry in 1999 (lb) C = total corn milling capacity in 1999 (gal) μ = utilization U.S. average (%) Q_{ref} = reference flow (gal) m = market value proportion of produced ethanol (%)
energy	$R_E = E/C\mu Q_{ref}e$ R_E = release to air or water (lb) E = emission or effluent from the Corn Milling Industry in 1999 (lb) C = total corn milling capacity in 1999 (gal) μ = utilization U.S. average (%) Q_{ref} = reference flow (gal) e = energy value proportion of produced ethanol (%)
demand	$R_D = E/C\mu Q_{ref}$ R_D = release to air or water (lb) E = emission or effluent from the Corn Milling Industry in 1999 (lb) C = total corn milling capacity in 1999 (gal) μ = utilization U.S. average (%) Q_{ref} = reference flow (gal)

TABLE 3. Percentage of Gas- and Water-Phase Emissions Assigned to Gasoline from Petroleum Refining and to Ethanol from Corn Milling (Wet and Dry) Using Weight, Volume, Market Value, Energy, and Demand-Based Allocation

	basis for allocation				
	weight	volume	market value	energy	demand
petroleum refining: gasoline	43.7%	46.5%	55.1%	44.3%	N/A
corn milling (wet): ethanol	48%	38%	70%	57%	100%
corn milling (dry): ethanol	49%	25%	76%	61%	100%

refining were modeled as pounds of emission per gallon of product from petroleum refining, SIC 2911. Emissions from the petroleum refining sector, using EPA sector notebook (*13*) and TRI data, were divided by the total volume of refinery products produced in 1999 (*14*) and then multiplied by the reference flow, gallons of conventional gasoline.

Corn Milling (Wet). Ethanol produced via wet milling is 38% of the output by volume. The typical yield from wet mills is 2.55 gallons of ethanol per bushel of corn. Gas- and water -phase emissions from wet milling were modeled as pounds of emission per gallon of milled product. Total production in 1999 was estimated by taking the total capacity

TABLE 4. Impact Indicators for Conventional Gasoline (CG) and Gasoline Oxygenated with Ethanol (EtOH) Using Allocation Methodologies Based on Weight, Volume, Market Value, Energy, and Demand[a]

	weight		volume		market value		energy		demand[b]
	CG	EtOH	CG	EtOH	CG	EtOH	CG	EtOH	EtOH
air impacts									
acidification	1495	1749	1506	1737	1518	1771	1485	1741	1742
ecotoxicity	1.42	17.80	1.43	17.80	1.44	17.85	1.42	17.81	17.87
eutrophication	1.39	1.64	1.39	1.64	1.40	1.65	1.39	1.64	1.64
global warming	10860	9776	10860	9713	10860	9776	10860	9776	9776
human health cancer	1.095	1.104	1.095	1.104	1.096	1.105	1.0948	1.104	1.104
human health criteria	6477	7049	6502	7034	6529	7098	6454	7031	7031
human health noncancer	117.8	126.3	118.2	126.4	118.6	127.6	117.6	126.3	127.2
ozone depletion	2.42×10^{-5}	2.26×10^{-5}	2.55×10^{-5}	2.38×10^{-5}	2.69×10^{-5}	2.51×10^{-5}	2.32×10^{-5}	2.17×10^{-5}	2.17×10^{-5}
photochemical smog	41.37	45.29	41.46	45.25	41.56	45.47	41.28	45.22	45.23
water impacts									
ecotoxicity	3.72	5.89	3.72	5.89	3.72	5.89	3.72	5.89	5.89
eutrophication	0.008	0.237	0.008	0.237	0.009	0.237	0.007	0.237	0.237
human health cancer	0.0669	0.0728	0.0669	0.0729	0.0669	0.0729	0.0669	0.0728	0.0728
human health noncancer	3760	3540	3760	3536	3760	3540	3760	3540	3540

[a] Results in italics indicate categories with lower impacts for EtOH. [b] Demand-based allocation is not applicable to petroleum refining of gasoline.

for wet mills (TRI) and adjusting for 76% utilization (*16*). Emissions from wet mill ethanol production (TRI) were divided by the total adjusted production to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow, in gallons, for wet milling.

Corn Milling (Dry). Ethanol produced via dry milling is 25% of the output by volume. The typical yield from dry mills is 2.7 gallons of ethanol per bushel of corn. Gas- and water-phase emissions from dry milling were also modeled as pounds of emission per gallon of milled product. Emissions from dry mill ethanol production (TRI) were divided by the total adjusted production to calculate pounds of emissions per gallon of milled product, and then multiplied by the reference flow, in gallons, for dry milling.

Methodology for Market Value-Based Allocation. Petroleum Refining. Gas- and water-phase emissions from petroleum refining were modeled as pound of emission per dollar of product sales from petroleum refining, SIC 2911. Emissions from the petroleum refining sector (*13*) were divided by the total refined product sales in 1999 (*17*) and then multiplied by the reference flow, gallons of conventional gasoline, and by the average sale price of gasoline (*18*).

Corn Milling (Wet). Gas- and water-phase emissions from corn milling were modeled as pound of emission per dollar of milled product. Emissions from wet mill ethanol production (TRI) were divided by the estimated total annual production volume to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow for wet milling, in gallons, and by 70% to allocate to the market value of the ethanol (*15*).

Corn Milling (Dry). Gas- and water-phase emissions from corn milling were also modeled as pound of emission per dollar of milled product. Emissions from dry mill ethanol production (TRI) were divided by the estimated total production volume to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow for wet milling, in gallons, and by 76% to allocate to the market value of the ethanol (*15*).

Methodology for Energy-Based Allocation. Petroleum Refining. Gas- and water-phase emissions from petroleum refining were modeled as pound of emission per Btu of refinery product. Emissions from the petroleum refining sector (*13*) were divided by the total energy content (Btu) of refinery products produced in 1999 (TRI) and then multiplied by the reference flow, gallons of conventional gasoline, which was converted to Btu by using an average energy content of gasoline, 115 500 Btu/gal.

Corn Milling (Wet). Gas- and water-phase emissions from wet corn milling were modeled as pound of emission per Btu of milled product. Emissions from wet mill ethanol production (TRI) were divided by the estimated annual production volume to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow, in gallons, for wet milling, and by 57% to allocate to the energy content of the ethanol (*15*).

Corn Milling (Dry). Gas- and water-phase emissions from dry corn milling were also modeled as pound of emission per Btu of milled product. Emissions from dry mill ethanol production (TRI) were divided by the estimated total annual production to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow, in gallons, for dry milling, and by 61% to allocate to the energy content of the ethanol (*15*).

Methodology for Demand-Based Allocation. Petroleum Refining. Demand-based allocation is not applicable to petroleum refining which produces multiple products, none of which can be identified as a main, intended product. If gasoline were no longer produced, the many other petroleum products would continue to be produced. To create data for the ethanol fuel systems, petroleum refining was modeled using energy-based allocation, since demand is driven by energy content for refinery products.

Corn Milling (Wet). One hundred percent of the emissions were assigned to ethanol. Emissions from wet mill ethanol production (TRI) were divided by the estimated annual production volume to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow for wet milling, in gallons.

Corn Milling (Dry). One hundred percent of the emissions were assigned to ethanol. Emissions from dry mill ethanol production (TRI) were divided by the estimated total annual production to calculate pounds of emissions per gallon of milled product and then multiplied by the reference flow for dry milling, in gallons.

TABLE 5. Impact Indicators for Conventional Gasoline (CG) and E85 Fuel Using Allocation Methodologies Based on Weight, Volume, Market Value, Energy, and Demand[a]

	weight		volume		market value		energy		demand[b]
	CG	E85	CG	E85	CG	E85	CG	E85	E85
					air impacts				
acidification	1495	5060	1506	5060	1518	5071	1485	5061	5073
ecotoxicity	1.42	219.39	1.43	219.21	1.44	219.81	1.42	219.56	220.37
eutrophication	1.39	4.86	1.39	4.86	1.40	4.86	1.39	4.86	4.86
global warming	10860	−3178	10860	−3178	10860	−3178	10860	−3178	−3178
human health cancer	1.095	1.158	1.095	1.157	1.096	1.160	1.095	1.159	1.163
human health criteria	6477	14892	6502	14898	6529	14903	6454	14888	14888
human health noncancer	117.8	241.6	118.2	238.6	118.6	248.4	117.6	244.3	257.3
ozone depletion	2.42×10^{-5}	7.73×10^{-6}	2.55×10^{-5}	8.00×10^{-6}	2.69×10^{-5}	8.29×10^{-6}	2.32×10^{-5}	7.52×10^{-6}	7.52×10^{-6}
photochemical smog	41.37	88.63	41.46	88.63	41.56	88.71	41.28	88.63	88.71
					water impacts				
ecotoxicity	3.72	33.63	3.72	33.63	3.72	33.63	3.72	33.63	33.63
eutrophication	0.008	3.133	0.008	3.133	0.009	3.133	0.007	3.133	3.133
human health cancer	0.0669	0.0619	0.0669	0.0619	0.0669	0.0619	0.0669	0.0619	0.0619
human health noncancer	3760	1428	3760	1428	3760	1428	3760	1428	1428

[a] Results in italics indicate categories with lower impacts for E85. [b] Demand-based allocation is not applicable to petroleum refining of gasoline.

Results

Tables 4 and 5 present the results of the impact modeling of gas and water phase emissions for the three fuel systems using allocation methodologies based on weight, volume, market value, energy, and demand. Table 4 compares conventional gasoline (CG) to gasoline with ethanol (EtOH), while Table 5 compares conventional gasoline to E85.

As can be seen in the tables, the relative ranking of conventional fuel to the alternate fuels is consistent in all cases. Global warming, ozone depletion, and human health-noncancer (for water) are the only impact categories that have lower results for gasoline with ethanol than for conventional gasoline. E85 also shows lower results for these three impact categories, as well as for human health cancer.

The results in Tables 4 and 5 are presented as precise numbers, that is, no ranges or error bars are provided. As noted earlier, while an attempt was made to acquire plausible data for the study, the data have not been verified for accuracy. Actual inventory data would reflect some variation which, in turn, would affect the impact results. This variation should be taken into account when identifying which product is preferable. However, the purpose of this study was to compare allocation methodologies, rather than to assess the environmental profiles of the selected products. Therefore, the data were taken as single numbers without ranges.

The results of the study show that, for this particular case study, allocation methodology does not alter the final LCA result when comparing systems. In the first case (conventional gas versus gasoline with ethanol), it should be recognized that the alternative products are very similar in composition: one product is 100% gasoline, while the alternative is approximately 92% gasoline. Therefore, there may not be enough of a difference in the inventory between the two products to affect the final results in the impact modeling. However, in the comparison with E85, variation of the allocation showed the same result.

Acknowledgments

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5.6 Discussion

5.6.1 General Observations

As can be seen in Tables 4 and 5 of the preceding paper, the results of the allocation scheme showed remarkable consistency when comparing the impacts of conventional gasoline to the ethanol alternatives. In each scenario, conventional gasoline resulted in lower impact scores in the global warming, ozone depletion, and human health-noncancer (water) categories; E85 resulted in a lower impact score for human health cancer.

As expected, the individual scores within each impact category varied depending on the allocation method (weight, volume, market value energy, or demand-based allocation) that was used. Conventional gasoline showed the largest variation (approximately 11%) in the ozone depletion category; the other variations were around 6% or less. However, the relative ranking of conventional gas to the alternative ethanol fuels is consistent in all cases.

The results of the study showed that for this particular case study allocation methodology does not alter the final LCA results when comparing systems. In the first case (conventional gasoline versus gasoline with ethanol), it should be recognized that the alternative products are very similar in composition; one product is 100% gasoline while the alternative is approximately 92% gasoline. Therefore, there may not be enough of a difference in the inventory between the two products to affect the final results in the impact modeling. However, in the comparison with E85, varying allocation showed the same result.

These results should not be interpreted as a reflection of the actual environmental impacts related to the three products. The inventory data that were collected on the input and outputs to the systems have not been peer-reviewed. Furthermore, the results of the TRACI model have not been validated nor were any gaps in the modeling ascertained, i.e. it was not verified that all the inventory data were modeled by TRACI. This should not be equated to a weakness in the research since the 'correctness' of the results in assessing harm to human health and the environment is not the issue here. The goal of the research study was to ascertain the modeling variability which resulted as long as the same impact models were applied to the same starting data.

This supports the hypothesis that the selection of a basis for co-product allocation has no impact on the results of the total LCA when comparing two or more systems as long as the chosen basis is applied consistently. That is, the relative ranking of two products, i.e. Product A is preferable to Product B, will be the same regardless of the allocation methodology that is chosen. This suggests that in creating an LCA it is appropriate to select an allocation rule that is most 'meaningful' to the practitioner, or the rule that the practitioner deems is most appropriate, as long as the rule is applied consistently to all the systems being modeled.

5.6.2 Limits of the Study

The results outlined here only apply to the current case study of automobile fuels. The applicability of allocation rules is somewhat limited in that allocation is only needed at the fuel production processes of crude oil refining and ethanol production; the other processes, i.e., fuel use and distribution and corn growing, as well as any associated transportation, did not require allocation.

Although a 'process-specific' approach, proposed in Chapter 4, was not conducted explicitly, the results of such a comparison can be seen in the results tables (Tables 4 and 5). Since the variability within each impact category is small, the overall comparison of the alternative systems is the same, regardless of the allocation method used.

5.6.3 Incentivising Recycling

In processes that produce a recycled stream that is sent to another process, the material is still considered to be a waste in the present system even though it becomes a by-product in another product system; therefore, it should no longer be counted as a waste in the LCI. Since inventory allocation only applies to co-products and not to waste materials (wastes are counted as part of the LCI), the inventory emissions for the process are not allocated to it. The emissions are allocated to the main product or to other by-products in System 1, so the recycled material does not carry the environmental releases that were associated with its creation; however, any transportation and reprocessing of the recycled waste material should be accounted for in the LCA of the second product system. System 1 benefits by being able to report a lower inventory for waste created by the process. System 2 benefits from a lowered inventory due to a decrease in the amount of virgin material used as well as due to the use of recovered material that carries no associated environmental burden.

5.7 Recent Research by Others

In a recent study that was made available during the performance of this research, Guinee and Heijungs conducted a "quick scan" of different allocation scenarios for fossil fuel chains, using EcoInvent data (Guinee and Heijungs 2006). They sought to gain insight into allocations made in fossil fuel chains. Their conclusion is consistent with this thesis author's findings.

The investigators compared an average Dutch passenger car running on gasoline (petrol) with a similar car running on bio-ethanol, using economic allocation, physical allocation, and the EcoInvent default allocation. The authors point out that while the purpose of the scan was limited to being an illustration of the possible influences of different allocation scenarios, they conclude that different allocation methods generate large differences in allocation factors and thus, also vary the results at the level of single processes (up to almost 250). However, the aggregated results for the present case study only differ modestly (1 – 1.5). Therefore, these preliminary results are in agreement with the findings of this thesis author's research.

5.8 Recommendations for Further Allocation Research

• Develop a method to identify a 'demand product' in processes that produce co-products that may have been considered to be a waste material at one time but later were found to be marketable, such as manure used as a fertilizer.

• Study a variety of industries, including one that has a larger number of allocation points than was applied in the research conducted here.

• Further evaluate the effects of varying allocation on input, as well as output, materials.

• Develop general guidance designed for synchronizing the stated goal of the study with the most appropriate input materials and co-product allocation methodologies.

5.9 Conclusions

If one were to go solely on the publications on co-product allocation, the findings of the case study presented here on fuel systems would be unexpected. Many researchers who have written on allocation have been very keen on demonstrating how different allocation approaches can drastically alter the outcome of the inventory. However, all of this prior work was done at the process level.

The current research took this research one step further and investigated the impact of allocation at the product system level. For this particular case study which used hypothetical data that are loosely based on alternative fuel systems, it was shown that altering allocation based on weight, volume, energy, market value, or demand does not change the relative ranking of the alternative products.

However, these findings should not be extended to other products without further investigation. Additional case studies are needed to support, or not support, this thesis author's findings.

6 Concluding Discussion and Recommendations

"For a successful technology, reality must take precedence over public relations, for Nature cannot be fooled."

Richard Feynman (1918-1988)

6.1 What is the Future of LCA?

In this thesis, the author covered the many facets of LCA, which is both an emerging tool for environmental management as well as a concept which requires the user to look holistically at environmental issues. The chapters in this thesis have described LCA methodology, with a focus on allocation methodology, and explored the outlying issues that remain unresolved. In addition, the thesis author proposes that life cycle thinking is an important component in achieving sustainability. However, the tenor of the thesis has been largely on the origins and development of LCA up to this point in time. But what is the future of LCA/LCM as we journey towards sustainability?

There has been a dramatic increase of interest in LCA-based tools during 2007. This was brought to the forefront by the on-going debate of switching from fossil fuels to bio-fuels. Despite the rush to champion bio-fuels as an alternative energy source that *can* increase national security, reduce vehicle emissions, and provide increased revenue for the farming community; these claims are being contested by those who are taking a broader view, often referred to as "systems thinking," of the potential consequences. One prominent publication among the many newspaper and journal articles and reports that have been written to address these far-reaching concerns was produced by the OECD, entitled "Biofuels: Is the Cure Worse than the Disease?" (Doornbosch and Steenblik 2007). The authors present the following two fundamental questions:

1. Do the technical means exist to produce bio-fuels in ways that enable the world to meet demand for transportation energy in more secure and less harmful ways, on a meaningful scale and without compromising the ability to feed a growing human and animal population?

2. Do current national and international policies that promote the production of bio-fuels represent the most cost-effective means of using biomass and the best way forward for the transport sector?

The authors conclude that the rush to produce more energy crops threatens to cause food shortages and to damage biodiversity with limited benefits.

LCA practitioners around the world are also reporting an increase in interest as seen in the growing demand for their services by companies who wish to have LCAs conducted on their products and materials. In the U.S., Dr. James Fava attributes this surge in interest to a combination of the introduction of the Wal-Mart Scorecard , Al Gore, and the US Green Building Council (USGBC) (Fava 2007).

Wal-Mart Stores, Inc. developed criteria for use in a scorecard that its stores will make available to buyers to evaluate environmental sustainability of their products. Starting in 2008, Wal-Mart will ask suppliers to fill out the scorecard and buyers will have the option to use the scorecard results to influence purchasing decisions. The scorecard will evaluate electronics on energy efficiency, durability, upgradability, end-of-life solutions, and the size of the package containing the product (http://www.walmartfacts.com/articles/4861.aspx).

The Wal-Mart scorecard is a prime indication of how the private sector is ahead of the curve on taking action to meet sustainability goals. At the same time, policy making in the public sector is lagging in that regard. Guidance from local, State, and national governments continue to be mainly media-focused without a cross-media view, and often without assessing the entire product system. There are a few examples which are exceptions to this narrow focus, such as the U.S. EPA's Environmentally Preferable Purchasing Program (http://www.epa.gov/epp/). The EPP Program promotes the use of a life cycle perspective but its efforts are organized in a 'silo' mode with its activities mainly directed by media-driven regulations. In EPP, environmental preferability reflects specific environmental attributes such as increased energy efficiency, reduced toxicity, or reduced impacts on fragile ecosystems.

The European Union member states continue to witness an increase in interest in life cycle thinking which surged with the passage in 1998 of the Integrated Product Policy. IPP was created to address the whole life-cycle of a product, thus avoiding shifting environmental problems from one medium to another. In a communication on IPP (COM (2003)302), the European Commission concluded that LCAs provide the best framework for assessing the potential environmental impacts of products currently available. This document, which identified the need for more consistent data and consensus LCA methodologies, led to the formation of the EC Platform on Life Cycle Assessment (http://lca.jrc.ec.europa.eu/). The Platform's deliverables have two main targeted applications: Support life cycle thinking in the development of goods and services in European Private Sector and to support life cycle thinking in a broad range and variety of European policy activities. The first phase of the project started in September 2005, and it is expected to conclude at the end of 2008.

LCA has also received much attention in Asia. Recognizing that LCA is utilized widely by many companies and organizations as a method for assessing environmental-friendly products, the Japanese Research Center of Life Cycle Assessment (LCA-Center) was established in 2001 and granted a period of seven years for its research activities. The goal was to encourage the adoption of LCA for towns, local communities and societies in order to reduce environmental load. Progress in LCA was significantly catalysed by the National LCA Project. The project developed a publicly available database and developed the LCA software "AIST-LCA Ver.4." Furthermore, the Japanese have held the bi-annual "Ecobalance" conference in Tsukuba, Japan, since 1994. Beginning in 2008, LCA research will continue in the research institute of AIST (Advanced Industrial Science and Technology)

Table 6-1 Societies and organizations have been formed around the world to meet the increasing demand for LCA information and services

Africa	African LCA Network (ALCANET)	http://ciclo-cycle.obiki.org/net/ALCAN.html
America	American Center for LCA (ACLCA)	www.lcacenter.org
APEC	LCA Researcher's Network for APEC Member Economies (APLCANET)	http://unit.aist.go.jp/lca-center/asianetwork/top.htm *or* http://aplcanet.rmit.edu.au/
Australia	Australian Life Cycle Assessment Society (ALCAS)	www.alcas.asn.au
Canada	CIRAIG	http://www.polymtl.ca/ciraig/
Denmark	LCA Center Denmark	www.lca-center.dk
India	Indian Society of LCA (ISLCA)	members.tripod.com/neef.in/islca.html
Japan	LCA Society of Japan (JLCA)	www.jemai.or.jp/lcaforum/
	Research Center for LCA	http://unit.aist.go.jp/lca-center/english/top.htm
	The Institute of LCA, Japan	http://ilcaj.sntt.or.jp/ (in Japanese)
Korea	Korean Society for LCA (KSLCA)	kslca.com (in Korean)
	LCA Research Center (LCARC)	http://www.lcarc.re.kr/English/
Latin America	Association of LCA in Latin America (ALCALA)	http://www.scientificjournals.com/sj/lca/Pdf/aId/7637
Mexico	Mexican Center for LCA and Sustainable Design	http://www.lcamexico.com
Sweden	Center of Environmental Assessment of Product and Material Systems (CPM)	http://www.cpm.chalmers.se
Philippines	Philippines LCA Clearinghouse (PhiLCA)	http://www.dlsu.edu.ph/research/centers/cesdr
Thailand	Thai LCA Network	http://www.thailca.net

(Curran 2006)

So, given the rising interest in LCA, what is its likely future? While LCA is simple in concept, the details of its practice are complex and still evolving. Researchers are continually coming up with new ways to enhance accuracy and applicability. In order for LCA to continue on the path of increased awareness and broadened applicability to serve as a decision-support tool, several key factors must be met in the near future:

- Life cycle thinking must be implemented in the earliest stage of the decision-making process.

- The use of life cycle thinking should be encouraged within all industrial and governmental sectors; however, this approach must maintain the essential characteristics of LCA and not overly simplify the tool in such a way that the system view is lost.

- The implementation of life cycle tools and techniques should be implemented at the simplest level of data collection without sacrificing rigor.

- LCA-based tools and data must be readily accessible by the public at little or no cost.

LCA was initially developed as a way to evaluate solid waste management in order to address the anticipated growing shortage of landfill space (referred to in the 1980s as the 'landfill crisis'). With the rise in interest in "product policy" and "extended producer responsibility (EPR)" in the 1990s, interest in LCA shifted from being a little-known cottage industry to an internationally-recognized analytical tool in support of environmental management decision-making. Today, the emphasis of environmentalism in regard to global climate change further demonstrates a need that LCA can/may help to fulfil.

For many people sustainability translates into environmentally friendly processes, products, and services; however, its scope is much broader than that. LCA tests our assumptions about what is 'green.' Understanding and adopting sustainable practices requires a new awareness of the world as a whole. LCA can help to provide a composite measure of sustainability practices. As a result, an increasing number of governments and corporations around the world have turned to LCA as an important environmental management, decision-support tool.

6.2 Closing the Gap between Methodology and Implementation

As Donella Meadows put it, we can never fully understand our world. Self-organizing, nonlinear, feedback systems are inherently unpredictable. And they are not completely controllable. The best we can do is to develop an understanding of them in the most general way (Meadows 2001).

LCA can be a useful tool for helping us with developing a more holistic understanding. LCA can help us move into the future; because key messages need to be conveyed to users and decision-makers, at all levels for both the short and for the long-term.

The following 'take away' messages have been gleaned from the preceding chapters of this thesis and should be kept in mind when embarking on conducting an LCA of a product, process, or activity:

- Understand and appreciate the full meaning of systems thinking and avoid a single media and single pollutant focus. An LCA can identify when an alternate solution has the potential to create larger, or different, environmental impacts when viewed across all thee media (air, water, and land).

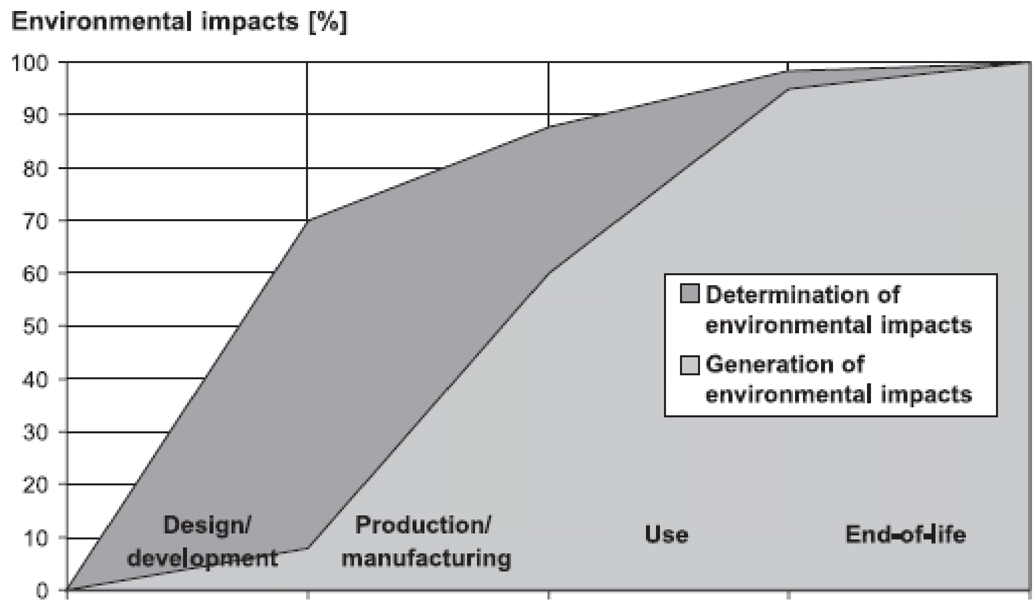
- Optimize the entire system and avoid optimizing only parts of the system. Place as much emphasis on modeling 'upstream' and 'downstream' activities as on production and manufacturing. For example, initiatives such as 'green chemistry,' 'greening the supply chain,' 'extended producer responsibility,' and 'end-of-life management' include only part of the system. LCA allows decision-makers to study an entire product system and avoid the sub-optimization that could otherwise result.

- Avoid the "if you can't define it and measure it, then ignore it" mindset. No one can precisely define or measure justice, democracy, security, freedom, truth, or love. No one can precisely define or measure any value. But account for them in a qualitative way and retain their significance in the analysis.

- Not all impacts can or should be translated to a monetary value. Other tools, such as Benefit-Cost Analysis (BCA) and Life Cycle Costing (LCC) are more appropriate tools for assessing environmental actions and reporting them in financial terms.

Furthermore, the product design and development phase is often overlooked in discussions on LCA methodology and application. Yet, a product's design can highly influence the environmental impacts of the life cycle stages (Rebitzer, Ekvall et al. 2004). The design/development stage has been depicted as separate from the other life cycle stages but as an inter-related process (see Figure 6-1). Additional research is needed to develop an integrated approach which companies can use to fully address the concept of product design from a life cycle perspective while addressing regulatory and consumer concerns.

Figure 6-1 The design/development stage is inter-related with the life cycle stages



(Rebitzer 2002)

6.3 Life is Complicated; Lists are Simple

This dynamic development of life cycle approaches demonstrates their value but only if they can be developed so as to be practical, relevant, and scientifically sound. Only then can the application of life cycle thinking help to contribute to improved environmental improvement. The recent interest in life cycle tools provides the opportunity to 'mainstream' life cycle approaches and to make them relevant on a broader scale. At the same time, this acceleration poses a risk of 'quick and dirty fixes' which are well meant but may compromise scientific validity and transparency (Rebitzer, Hellweg et al. 2007) and thus, decrease the trust and confidence decision-makers may have in the usefulness of the insights that may be gained from LCAs.

LCAs and the wicked problems that they can help us to address are often seen as overly complicated and impractical because of problems related to data needs, time, expense, and reproducibility. In order to get to something more practical, some people have turned to the use of lists and metrics. Environmental metrics have been evolving for several decades generally paralleling environmental regulations. Today, a variety of metrics for use in environmental management is available. The decision-maker must decide if the metrics and tools that are being used are helpful for making progress towards environmental sustainability, or if they are only effective for meeting individual or corporate needs (Bridges and Curran 2007)?

For example, the Leadership in Energy and Environmental Design (LEED) Green Building Rating System™ is the nationally accepted benchmark for the design, construction, and operation of high performance green buildings. The LEED rating system was developed by the US Green Building Council (USGBC) to promote a whole-building, whole life cycle, approach to sustainability. LEED recognizes performance in the following five key areas of human and environmental health:

1. Sustainable sites
2. Water efficiency
3. Energy and atmosphere
4. Materials and resources
5. Indoor environmental quality
6. Innovation and design process

The acceptance of LEED as a green building rating system has generated substantial excitement among the green building and environmental communities. It appears, in fact, that the LEED rating system has become a key driving force for uniting U.S. environmental and architectural movements. As the USGBC grows in membership and influence it is important to explicitly state this objective, since a broader membership concomitantly means that broader interests are at stake. The overarching objective, simply put, is to create buildings that are designed to minimize impacts on human and ecosystem health during their entire life as buildings.

In their paper entitled "Integrating LCA Tools in Green Building Rating Systems," Trusty and Horst focus on the disconnect between the complexity of the building/environment relationship and the specifics of rating system credits and requirements (Trusty and Horst 2002). They argue that achieving the ultimate objective means working to minimize flows from and to nature: the use of natural resources of all kinds, and the emissions to air, land and water throughout a building's complete life cycle. Through several examples, they show how certain credit/requirement combinations do not necessarily lead to improved environmental performance in the sense of this objective function. They are based on assumptions that may or may not prove to be correct in certain situations. They also discuss how LCA, while not a panacea, has the ability to move the building community in the right direction (Horst and Trusty 2003).

In 2004, the leadership at the USGBC began discussing the need to integrate LCA into the LEED rating system. Work on this integration continues (Trusty 2006).

6.4 Recommendations for Future Research on LCA Methodology and Application

Based on the foregoing chapters of this thesis, a number of topics for further research on LCA methodology and application have been identified. They are grouped under the following four headings: LCA Goal and Scope Definition, Life Cycle Inventory, Life Cycle Impact Assessment, and Life Cycle Management.

6.4.1 Specific Research Recommendations Related to LCA Goal and Scope Definition

The first step in an LCA is to clearly establish the goal of the study. From there, the scope of the analysis can be defined, the boundaries for the study can be drawn, the functional unit can be established, and the impacts to be modelled can be identified. From that point on, conducting the LCA should be consistent with the stated goal. For example, if the goal is to look at the national impact of a process, such as transportation, then data should accurately reflect the national, versus a regional, average. However, currently there is no guidance on how to match the goal with the study and how it is carried out. This disconnect can be especially problematic in studies that are intended to be at the screening level, sometimes called simplified LCA, where it is not clear what level of effort should be expended.

The level of specificity that is required in inventory data depends on the goal of the study. For some studies, especially those conducted for internal use within a company, company specific data may be used. But in other cases, the use of average or generic data is needed and is appropriate. How these data are averaged can have a big impact on the output results and tentative conclusions. For example, data can be collected to represent average operations. Or, the worst-case or best-case might be modeled. In addition, processes that make similar products may vary greatly in age or have different permitting requirements depending on their location. These variations result in different types or amounts of environmental releases.

Data collected for an inventory should always be associated with a quality measure using data quality goals and data quality indicators. No pre-defined list of data quality goals exists for all LCA projects. The number and nature of data quality goals that are necessary depend on the level of accuracy required to inform the decision-makers involved in the process. Data quality indicators are benchmarks to which the collected data can be measured to determine if data quality requirements have been met. Similar to data quality goals, there is no pre-defined list of data quality indicators for all LCIs. The selection of data quality indicators depends upon which ones are most appropriate and applicable to the specific data sources being evaluated. Examples of data quality indicators are accuracy, precision, completeness, representativeness, consistency, and reproducibility. Although formal data quality indicators (DQIs) are strongly preferred, a description of how the data were generated can also be useful in judging quality.

The validity of the system expansion approach has been questioned in this thesis. In a system expansion approach, the boundaries are expanded to include the alternative production of exported functions. To do this, a necessary requirement of system expansion is the existence of an alternative way to produce a by-product. While this concept seems reasonable on the surface, it can be problematic. It is often used to 'credit' the system with avoided burdens that are offset by the alternative process. For example, byproducts from corn mills, such as corn meal and corn oil, are assumed to be reasonable substitutes for soybean products, such as soybean meal and soybean oil. But, guidance for identifying displaced products does not exist.

Furthermore, system expansion runs the risk of presenting a one-sided view. Even though petroleum is the staus quo, is it not reasonable to apply a balanced view and also look at the offset potential of petroleum-based products? That is, should we also account for the avoided emissions because corn is *not* being grown and ethanol is *not* being produced? In the absence of clear guidance, how system expansion is done is up to the whim of the person conducting the LCA.

Specific Research Recommendations Related to LCA Goal and Scope Definition

- Develop general guidance for defining research goals and for establishing appropriate study scope and boundaries for LCAs based on the goals.

- Develop general guidance for synchronizing the goal of the study with inventory data quality objectives, thereby aiding the determination of the level of effort needed to collect data, the use of surrogate data, the age of data, etc.

- Develop general guidance for synchronizing the stated goal of the study to the impact assessment methodology, thereby aiding in the selection of impact categories.

- Explore the validity of assigning 'credits' in system expansion and ascertain the appropriateness of this approach.

6.4.2 Specific Research Recommendations Related to Life Cycle Inventory

The lack of readily available inventory data continues to be a major hurdle for LCA practice. Inventory data can be created by collecting primary data directly from the sources, such as material and product manufacturers. More often data are collected from secondary sources such as reports, publications and databases. Data are held either privately, such as in LCA practitioners' software, or in the public domain, such as government sources. Commercial tools are usually fairly simple to use, although some training may be needed before the user is adept at using them. There is usually a subscription or purchase fee associated with these products.

While the use of readily-available software tool makes it easier to conduct an LCA, it is not always completely clear how the data were modeled in order to create the data found within them. The numerous, underlying assumptions, such as exclusions, which were applied during data collection are not typically revealed in most pre-packaged data programs. Ultimately, the user must rely on the reputation of the vendor for assurance on the quality of the data and the methods used to collect them.

Another option for creating life cycle inventories is the use of publicly-available databases. These databases are often government-sponsored, such as the US EPA's Toxic Release Inventory (TRI) and Australia's National Pollutant Inventory (NPI). They are easily accessible and available at no cost. But these sources do not lend themselves easily to use in most life cycle studies because the data are reported for individual sites or facilities and not as industry averages for a country or a region. Often assumptions have to be made about the data in order to aggregate them to represent an industry sector. Also, data are not allocated by production; therefore, additional information is needed in order to determine releases per product.

To achieve this, the most effective way to simplify the LCA process is to increase the collection, publication, and standardization of LCI data. The Europeans have been successful in creating publicly-available databases through efforts such as the EcoInvent database and more recently the European Commission's Platform on Life Cycle Assessment. The US has seen limited success in creating a national inventory database. The National Renewable Energy Laboratory (NREL) created the Life Cycle Inventory Database which now contains approximately 80 data modules. Additional funding is needed to support expansion and updating of the database.

Uncertainty analysis is the process of determining the variability of the data and the impact on the final results. This variability can be attributed to either errors or fluctuations in the data. Uncertainty applies to both the inventory data and the impact assessment indicators and can have a great impact on how the results are used in decision-making. However, the actual influence of uncertainty on decision-making has not been adequately studied thus far.

An alternate approach to streamlining the data collection effort could be accomplished through studying specific industry sectors and creating profiles to identify dominant stages that require closer scrutiny, or identify common inputs that may be negligible (however, negligibility cannot be based solely on mass). Such determinations could be made for the industry as a whole, decreasing the effort required in performance of subsequent LCAs. These streamlined methods would apply only specific systems, so that any "rules of thumb" would apply only to that sector.

Specific Research Recommendations Related to Life Cycle Inventory

- Establish a publicly-accessible, peer-reviewed database of life cycle inventory data presented in LCA-compatible format. Emphasis should be given to the use stage of products, which has not been studied as often as manufacturing processes.

- Develop general guidance for applying rules of exclusion, e.g. small amounts, capital equipment, infrastructure, workers, etc.

- Develop general guidance for conducting uncertainty analysis for life cycle inventory studies.

- Investigate the development of industry-specific guidelines that identify dominant stages, applicable rules of exclusion, etc. to help direct and minimize data collection.

6.4.3 Specific Research Recommendations Related to Life Cycle Impact Assessment

The selection of impact categories, or 'areas of protection,' is a critical step in impact assessment. However, determining which impact categories are the most relevant through a pre-defined list has not been done. Therefore, no grand taxonomy of impact categories is available from which to choose. However, it is recognized that different goals and scopes require different categories, data sets, and methodologies. There is general agreement that impacts to natural resources, the natural environment, the manmade environment, and human health are the ultimate goal, but the appropriate subdivision of these higher level categories has not been established.

While there is general consensus on how to model the global level impacts for global climate change and ozone depletion, impact models that occur at the regional level (acidification, eutrophication, etc.) or at the local level (human toxicity, ecotoxicity, etc.) vary greatly. In addition reliable models for nuisance impacts (noise and smell) and for radiation are yet to be developed.

Midpoint impact assessment models reflect the relative potency of the stressors at a common midpoint within the stressor-effect chain. Analysis at a midpoint minimizes the amount of forecasting and effect modeling incorporated into the LCIA, thereby reducing the complexity and cost of the modeling and often simplifying communication. Midpoint modeling can minimize assumptions and value choices, reflect a higher level of societal consensus, and be more comprehensive than model coverage for endpoint estimation. On the other hand, endpoint modeling leads to discrete, identifiable metrics which may have more relevance and meaning to the decision-maker. That is, it is easier to act on information regarding the potential new cases of cancer caused by ozone depletion than on the increase in the impact indicator for ozone depletion which is given in CFC-11 equivalents.

It has already been acknowledged that LCIAs do not attempt to directly assess the potential impact of releases, as a traditional risk assessment would do. To conduct the risk assessment of a release, environmental loading information must be subjected to fate and transport analysis to determine how the releases will be transferred to various environmental compartments (air, water, and soil). Additional temporal (time) and spatial (location) meta data are needed in order to conduct more rigorous impact modelling.

The life cycle inventory provides information on environmental releases to all media; however, to model potential impacts, it is necessary to know if the release has an affinity to the air, to water, or to the ground. That is, if released to the ground, does it stay in the ground? Or, does the release move to nearby water sources, or evaporate to the air? Furthermore, the exposure pathways to human and ecosystems must be clearly delineated. Actual monitoring or modeled data may be needed in these cases. For many chemicals, the availability of data for evaluating fate and exposure remains limited. Better models and monitoring methods are necessary to develop accurate release estimates.

The assignment of inventory data to different impact categories is referred as classification. In some cases, an environmental release contributes to only one impact category; however, other environmental releases contribute to more than one impact category. For example, nitrogen oxides (NOx) can be assigned to acidification, global warming, and stratospheric ozone depletion. In the real world, a release would result in contributing to a single impact, or possibly splitting across impacts. However, current LCIA models do not account for this partitioning and can assign the full amount of a release multiple times to more than one category, resulting in an overstatement of effects.

Specific Research Recommendations Related to Life Cycle Impact Assessment

- Develop general guidance and international consensus on life cycle impact assessment models to bring the practice to a consistent level.
- Generate temporal and spatial meta-data to accompany chemical release and resource use data in order to enhance impact assessment results.
- Generate impact data for chemicals and resources that have not yet been studied and catalogued.
- Develop general guidance for partitioning releases across media (air, water, and soil) and across impact caegories.

6.4.4 Specific Research Recommendations Related to Life Cycle Management

While the driving force for LCA studies has been the desire to reduce burdens on the environment by altering parts of a product system, benchmarking a product against a competitor or proving that one product is environmentally preferable to another has also been a motivator. However, moving from the results of the impact assessment to a final decision requires three additional considerations: normalization, valuation, and uncertainty management.

Normalization is applied in order to indicate the relative contribution of an environmental impact. Normalization calculates the magnitude of each impact compared to a reference (or normal) value by dividing the impact category by the reference, such as the total annual environmental load in a country, or the number of inhabitants.

While normalization can be described as a science-based approach, the underlying assumptions in normalization methodology, such as the choice to use a per capita basis, is a very subjective process, blurring the line between sound science and modeling assumptions (Bare, Pennington et al. 1999). From a decision analysis point of view, the preferable outcome of the normalization step is to convert the different scales of the impact indicators into the same range. There are many different ways that normalization is being conducted worldwide, but no standardized method exists.

It is not immediately obvious how to compare impact assessment results. For example, how do global warming impacts compare to acid rain results? Studies that aim to identify a "winning" option require the application of value judgments to the LCA results. Before impacts scores can be summed into a single indicator, a weighting, or valuation, step is needed. In order to address this challenge, various weighting methods and approaches have been proposed. However, no single weighting method can meet all decision-makers' needs. All methods contain a high degree of subjectivity. It is up to the individual decision-maker to reflect his or her preference in the interpretation of results. Clear guidance is needed for helping decision-makers determine such weighting schemes.

Uncertainty analysis is the process of determining the variability of the data and the impact on the final results. This variability can be attributed to either errors or fluctuations in the data. Uncertainty applies to both the inventory data and the impact assessment indicators and can have a great impact on how the results are used in decision-making. However, the actual influence of uncertainty on decision-making has not been adequately studied thus far.

Sustainable development requires the consideration of the economic, environmental and social aspects of product systems. Therefore, responsible decision-making in public policy, industry and related fields should consider those issues for present and future relevance. Currently there is no single technique to deliver an overall answer with regard to environmental decision-making. Results of an LCA will not determine which product or process is the most cost effective or works the best. Therefore, the information developed in an LCA study should be used as one component of a more comprehensive decision-making process for assessing the trade-offs. Research is needed to develop a framework which integrates results of LCAs into the decision-making process along with other pertinent factors, especially cost and societal needs.

Specific Research Recommendations for Life Cycle Management

- Develop a framework which integrates LCA results into the decision-making process, including development of normalization factors and weighting schemes.
- Develop general guidance for assessing the social aspects of products, processes and services that can be integrated with the results of an LCA in the decision-making process.
- Conduct research to develop guidance on how to integrate environmental, economic and social impact factors and to assist in determining whether this should be or could be done by integrating the tools, or by integrating the information generated by the tools in the decision-making framework.
- Conduct application–focused research to determine how businesses and governments can apply LCA as a support tool for decision-making.

6.5 Epilogue

The business climate of the 21st century continues to see increases in globalization, advancements in information technology, rapid process and product innovations and chaotic marketplace demands. All this is taking place on a planet that is quickly approaching its ecological limits. Signs are already being seen in critical areas, such as ozone layer depletion, global climate warming and various types of environmentally induced toxic pollution.

Increasing demands for change are causing significant adjustments in how companies design, produce and deliver products and services to their customers. It can be anticipated that the economy of this evolving century will be characterized by a decrease in the material and energy content of products and an increase in their knowledge content. The challenge for businesses and governments will be to ensure that continued economic development is ecologically and socially sustainable. With the need and desire for equal development opportunities for all, the issue of sustainability continues to be a key issue as we move forward. We are beginning to recognize that the path towards sustainability requires life cycle thinking and the cooperation among the various stakeholders throughout the life cycles of products and services.

If we are to achieve a truly sustainable society, the application of methods and tools to measure and compare the environmental impacts of human activities for the provision of products and services is a requirement. Every product, as well as service, has a life with associated environmental impacts from emissions to the environment and the consumption of resources that occur when extracting resources, producing materials, manufacturing the product, during consumption/use, and at the product's end-of-life. These emissions and use of resources contribute to a wide range of impacts, such as climate change, stratospheric ozone depletion, tropospheric ozone (smog) creation, eutrophication, acidification, toxicological stress on human health and ecosystems, the depletion of resources, including water use and land use. A clear need exists to take a proactive stand and to provide insights, apart from current regulatory practices, to help to reduce such impacts.

The paradigm of sustainability, with its three aspects of social responsibility, economic performance, and environmental stewardship, has become accepted as the goal of public policy. However, as in the past, approaches to environmental protection continue to be based on 'end-of-pipe' solutions which are focused on single media (e.g., air, water, soil) and within a single stage in the life-cycle stage of a product. These do not always lead to an overall reduction in environmental impacts. Pollution control resources are spent on activities that are required by laws and regulations, and that do not always provide the most efficient use of those resources in terms of reducing impacts. This has often led to unintended consequences where one environmental problem is solved while another is generated. Because they are not designed to address a full understanding of the trade-offs and their implications in a systematic fashion, end-of-pipe approaches often diminish opportunities for achieving net environmental improvements.

Interest in life cycle management continues to grow. Our knowledge and understanding of life cycle-based methods and applications have developed significantly over the past three decades. Further refinement of these methodologies is needed but care should be taken to ensure that the pursuit does not end up being one of getting to more and more precise numbers. The life cycle thinking perspective has intrinsic value. As seen in the bio-fuels examples, different results have been presented but the overall message is the same – we must take care in how we acquire bio-feedstocks and should be cautious in jumping to conclude that everything natural is automatically good. The future offers both significant challenges and opportunities for LCA and LCA-related approaches to play important decision-making support roles.

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8 Glossary

Allocation – the apportionment of environmental burdens from an industrial process across multiple inputs and/or multiple products.

Attributional LCA – an accounting of input and output flows and assessment of the resulting potential impacts related to a defined system at a given point of time.

Characterization Factor – an estimate of potential contribution towards an impact category, such as global warming potential.

Cleaner Production – the continual effort to prevent pollution; the reduction of the use of energy, water and material resources, and minimization of waste in the production process. It involves re-thinking products, product components and production processes to achieve sustainable production (see also Pollution Prevention)

Consequential LCA – an estimation of how flows to and from the environment, and assessment of the resulting potential impacts, will be affected as a result of a change to a system.

Ecobalance – life cycle assessment

Exclusions – see Rules of Exclusion

Life Cycle Assessment - a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impact of those energy and materials used and releases to the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing, extracting and processing raw materials; manufacturing, transportation and distribution; use, reuse, maintenance; recycling, and final disposal.

Life Cycle Cost - A product's initial costs plus all future costs (operating, maintenance, repair and replacement costs, and functional-use costs) minus the product's salvage value (i.e., value of an asset at the end of economic life or study period). All costs are discounted to adjust for the time value of money.

Life Cycle Inventory- the accounting of the natural resource inputs and outputs (releases to the environment) across a defined product or process life cycle.

Life Cycle Impact Assessment – evaluation of the magnitude and significance of the potential environmental impacts of product or process system.

Life Cycle Design - an engineering approach that takes the complete life cycle of products into account in the design/development phase.

Life Cycle Management – the integration of concepts and techniques to address environmental, economic, technological, and social aspects of products, services, and organizations on a life cycle basis.

Life Cycle Stages – the processes in a product system including raw material acquisition, production use/reuse/maintenance, and final disposal.

Life Cycle Thinking - a qualitative discussion to identify the stages of a product or process life cycle and the potential environmental impacts of greatest significance.

Primary Data – data collected directly from the source of use or release.

Pollution Prevention – the reduction or elimination of waste at the source by modifying production processes, promoting the use of non-toxic or less-toxic substances, implementing conservation techniques, and re-using materials rather than putting them into the waste stream.

Rules of Exclusion – also called cut-off rules, limiting the study by including only processes that make relevant contributions to some environmental input or output.

Screening LCA - an initial, first-run study that may be conducted to determine if additional study is needed and, if so, where the focus of additional study should be placed

Secondary Data – previously collected and reported resource use and release data.

Simplified LCA – an application of the LCA methodology which minimizes the data collection effort by using generic data (qualitative and/or quantitative) or standard data modules, such as transportation or energy production, or by focusing on the most important environmental aspects and/or potential environmental impacts and/or stages of the life cycle and/or phases of the LCA.

Streamlined LCA – see Simplified LCA

Sustainability - the concept of meeting the needs of the present without compromising the ability and quality of life of future generations.

System – interconnected industrial processes that encompass the cradle-to-grave operations which meet a defined function.

9 Executive Summary

Introduction

The idea of a comprehensive environmental management scheme that identifies unintended consequences before they occur is very alluring. Over the years, the instances in which one problem was solved but caused another are numerous. Tools are needed that can help us to evaluate the comparative potential cradle-to-grave impacts of our actions in order to help us to prevent such wide-ranging effects. Life Cycle Assessment, or LCA, is one tool that can provide assistance in the decision-making process.

After a brief look at general LCA methodology, this thesis author presents the current limitations that currently prevent LCA from being a widely-used tool that generates replicable, defensible results. This thesis author then introduces the subject of integrating LCA results into the decision-making process, suggesting that the potentially, holistic view offered by LCA can be valuable as one tool for helping to achieve sustainable societal development. Bio-fuels are presented as the subject of a "wicked problem" for which LCA can be used to support the decision-making process. This thesis author then focuses on the co-product allocation issue and presents the results of testing various allocation schemes across an industrial system. The final chapter presents overall conclusions and recommendations for further, urgently needed, research.

LCA Methodology

LCA is a tool for evaluating possible outcomes and consequences of different actions. The specific application under review is a "cradle-to-grave" approach for assessing industrial systems. It considers aspects of resource use and environmental releases associated with a system, as defined by the function provided by a product or good. This holistic approach considers relevant impacts up-stream and down-stream of the producer or consumer. Specifically, LCA helps researchers to evaluate potential environmental interactions that cover a range of activities, from the extraction of raw materials from the Earth and the production and distribution of energy, through the manufacture, use, and reuse, and final disposal of a product. LCA can also be applied to a process, an activity, or a service. By including the impacts throughout the life cycle, LCA provides a comprehensive view of the related environmental aspects, thus, providing a more accurate assessment of potential environmental issues when selecting between products and processes.

This ability to track and document shifts in environmental impacts can help decision-makers and managers more fully characterize the environmental trade-offs associated with product or process alternatives. By performing LCAs analysts can, for example:

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

Integrating LCA and Sustainability

There has been a dramatic increase of interest in LCA-based tools during 2007. This was brought to the forefront by the on-going debates of switching from fossil fuels to bio-fuels. Despite the rush to champion bio-fuels as an alternative energy source that ***can*** increase national security, reduce vehicle emissions, and provide increased revenue for the farming community; some are contesting these claims by taking a broader view, often referred to as "systems thinking," of the potential consequences. One prominent publication among the many newspaper and journal articles and reports that have been written to address these far-reaching concerns was produced by the Organisation for Economic Cooperation and Development (OECD), entitled "Biofuels: Is the Cure Worse than the Disease?" The authors present two fundamental questions: 1. Do the technical means exist to produce bio-fuels in ways that enable the world to meet demand for transportation energy in more secure and less harmful ways, on a meaningful scale and without compromising the ability to feed a growing human and animal population? and 2. Do current national and international policies that promote the production of bio-fuels represent the most cost-effective means of using biomass and the best way forward for the transport sector? The authors conclude that the rush to energy crops threatens to cause food shortages and to damage biodiversity with limited benefits.

The issue of environmental impacts related to bio-based materials, including bio-fuels, is a complicated one. There is no simple answer to the question "are materials from bio-based feedstocks environmentally preferable?" Bio-fuels, for example, appear to be effective in reducing some aspects (such as fossil fuel use) while increasing others (such as water quality impacts). It brings into question how we define and measure 'sustainability.' Whichever metrics are chosen to measure sustainability, the analysis must be on a life cycle basis.

Allocation Methodology in LCA

As part of LCA methodology development, allocation has been the topic of much debate in recent years. The search continues for a single, preferred approach or an underlying theory that determines which allocation approach is preferable in a given situation. While questions remain on the specifics for how to conduct some allocation techniques, such as how to identify a 'demand' product or classifying materials as wastes versus by-products, the most pressing issue appears to be the need for guidelines to match the most appropriate allocation approach to the goal of the study.

A literature review conducted by this thesis author shows that while many authors have written about the allocation issue, no one had looked at the impact that allocation has on an entire product system. To that end, this thesis author conducted research to evaluate the effect that changing the basis of co-product allocation has on the results when comparing two products at a system level. The research was designed to investigate the overall impacts of various allocation schemes across an entire system, instead of only focusing on the effects around individual processes. The important consideration was on the impact that choices for allocation have upon the total results of an LCA including the impact assessment phase. This research was performed to determine what impact the selection of a basis for co-product allocation has on the total LCA when comparing two or more systems.

Co-Product Allocation Study

The study involved creating a life cycle inventory for a baseline system (using Excel spreadsheets) then running the inventory through Life Cycle Impact Assessment (LCIA) models to generate impact indicators. These steps were repeated to create scores using different allocation schemes. First, conventional gasoline was analyzed with a reference flow of 1,000 gallons (3,785 liters), then an alternative product, gasoline blended with 8.7% ethanol by volume, was analyzed based on an equivalent functional unit (this volume was calculated from the U.S. federally-mandated oxygen content required in automotive fuel). Ethanol has a lower energy value, hence, more gasoline-blended fuel is needed:

Conventional Gasoline with MTBE:	20.22 mpg	(8.59 kmpl)	1,000 gallons (3,785 l)
Gasoline with Ethanol:	19.95 mpg	(8.48 kmpl)	1,014 gallons (3,838 l)

Due to the similarities of the conventional gasoline and gasoline with ethanol (i.e. the major component of both products is gasoline), a second case study was conducted to compare conventional gasoline with E85 (85% ethanol). The spreadsheets that were used to model the data for the gasoline with ethanol additive were modified to reflect a fuel mix of 85% ethanol by volume, and the reference flow was adjusted to 1,380 gallons (5,224 liters) to equal the energy content of 1,000 gallons of conventional gasoline.

E85 Ethanol Fuel:	14.65 mpg	(6.23 kmpl)	1,380 gallons (5,224 l)

The results of the allocation scheme showed remarkable consistency when comparing the impacts of conventional gasoline to the ethanol alternatives. In each scenario, conventional gasoline resulted in lower impact scores in the global warming, ozone depletion, and human health-noncancer (water) categories; E85 resulted in a lower impact score for human health cancer.

As expected, the individual scores within each impact category varied depending on the allocation method (weight, volume, market value energy, or demand-based allocation) that was used. Conventional gasoline showed the largest variation (approximately 11%) in the ozone depletion category; the other variations were around 6% or less. However, the relative ranking of conventional gas to the alternative ethanol fuels is consistent in all cases.

The results of the study showed that for this particular case study allocation methodology does not alter the final LCA results when comparing systems. In the first case (conventional gasoline versus gasoline with ethanol), it should be recognized that the alternative products are very similar in composition; one product is 100% gasoline while the alternative is approximately 92% gasoline. Therefore, there may not be enough of a difference in the inventory between the two products to affect the final results in the impact modeling. However, in the comparison with E85, varying allocation showed the same result.

These results should not be interpreted as a reflection of the actual environmental impacts related to the three products. The inventory data that were collected on the input and outputs to the systems have not been peer-reviewed. Furthermore, the results of the impact models that were used for the LCIA phase have not been validated nor were any gaps in the modeling ascertained, i.e. it was not verified that all the inventory data were modeled. This should not be equated to a weakness in the research since the 'correctness' of the results in assessing harm to human health and the environment is not the issue here. The goal of the research study was to ascertain the modeling variability which resulted as long as the same impact models were applied to the same starting data.

This supports the hypothesis that the selection of a basis for co-product allocation has no impact on the results of the total LCA when comparing two or more systems as long as the chosen bases are applied consistently. That is, the relative ranking of two products, i.e. Product A is preferable to Product B, will be the same regardless of the allocation methodology that is chosen. This suggests that in creating an LCA it is appropriate to select an allocation rule that is most 'meaningful' to the practitioner, or the rule that the practitioner deems is most appropriate, as long as the rule is applied consistently to all the systems being modeled. These findings should not be extended to other products without further investigation. Additional case studies are needed to support, or not support, this thesis author's findings.

Concluding Discussion and Recommendations

In this thesis, the author covers the many facets of LCA, which is both an emerging tool for environmental management as well as a concept which requires the user to look holistically at environmental issues. The chapters in this thesis describe LCA methodology, with a focus on allocation methodology, and explore the outlying issues that remain unresolved. In addition, the thesis author proposes that life cycle thinking is an important component in achieving sustainability. Based on the chapters of this thesis, a number of topics for further, urgently needed research are identified. They are grouped under four headings: Goal and Scope Definition, Inventory, Impact Assessment, and Life Cycle Management. The recommendations are as follow:

Specific Research Recommendations Related to LCA Goal and Scope Definition:

- Develop general guidance for defining research goals and for establishing appropriate study scope and boundaries for LCAs based on the goals.
- Develop general guidance for synchronizing the goal of the study with inventory data quality objectives, thereby aiding the determination of the level of effort needed to collect data, the use of surrogate data, the age of data, etc.
- Develop general guidance for synchronizing the stated goal of the study to the impact assessment methodology, thereby aiding in the selection of impact categories.
- Explore the validity of assigning 'credits' in system expansion and ascertain the appropriateness of this approach.

Specific Research Recommendations Related to Life Cycle Inventory:

- Establish a publicly-accessible, peer-reviewed database of life cycle inventory data presented in LCA-compatible format. Emphasis should be given to the use stage of products, which has not been studied as often as manufacturing processes.
- Develop general guidance for applying exclusion rules, e.g. small amounts, capital equipment, infrastructure, workers, etc.
- Develop general guidance for conducting uncertainty analysis for life cycle inventory studies.
- Investigate the development of industry-specific guidelines that identify dominant stages, applicable rules of exclusion, etc. to help direct and minimize data collection.

Specific Research Recommendations Related to Life Cycle Impact Assessment:

- Develop general guidance and international consensus on life cycle impact assessment models to bring the practice to a consistent level.
- Generate temporal and spatial meta-data to accompany chemical release and resource use data in order to enhance impact assessment results.
- Generate impact data for chemicals and resources that have not yet been studied and catalogued.
- Develop general guidance for partitioning releases across media (air, water, and soil) and across impact categories.

Specific Research Recommendations for Life Cycle Management:

- Develop a framework which integrates LCA results into the decision-making process, including development of normalization factors.
- Develop general guidance for assessing the social aspects of products, processes and services that can be integrated with the results of an LCA in the decision-making process.
- Conduct research to develop guidance on how to integrate environmental, economic and social impact factors and to assist in determining whether this should be or could be done by integrating the tools or by integrating the information generated by the tools in the decision-making framework.
- Conduct application–focused research to determine how businesses and governments can apply LCA as a support tool for decision-making.

The business climate of the 21st century continues to see increases in globalization, advancements in information technology, rapid process and product innovations and chaotic marketplace demands. All this is taking place on a planet that is quickly approaching its ecological limits. Signs are already being seen in critical areas, such as ozone layer depletion, global climate warming and various types of environmentally induced toxic pollution.

Increasing demands for change are causing significant adjustments in how companies design, produce and deliver products and services to their customers. It can be anticipated that the economy of this evolving century will be characterized by a decrease in the material and energy content of products and an increase in their knowledge content. The challenge for businesses and governments will be to ensure that continued economic development is ecologically and socially sustainable. With the need and desire for equal development opportunities for all, the issue of sustainability continues to be a key issue as we move forward. We are beginning to recognize that the path towards sustainability requires life cycle thinking and the cooperation among the various stakeholders throughout the life cycles of products and services.

If we are to achieve a truly sustainable society, the application of methods and tools to measure and compare the environmental impacts of human activities for the provision of products and services is a requirement. Every product, as well as service, has a life with associated environmental impacts from emissions to the environment and the consumption of resources that occur when extracting resources, producing materials, manufacturing the product, during consumption/use, and at the product's end-of-life. These emissions and use of resources contribute to a wide range of impacts, such as climate change, stratospheric ozone depletion, tropospheric ozone (smog) creation, eutrophication, acidification, toxicological stress on human health and ecosystems, and the depletion of resources, including water use and land use. A clear need exists to take a proactive stand and to provide insights, apart from current regulatory practices, to help to reduce such impacts.

The paradigm of sustainability, with its three aspects of social responsibility, economic performance, and environmental stewardship, has become accepted as the goal of public policy. However, as in the past, approaches to environmental protection continue to be based on 'end-of-pipe' solutions which are focused on single media (e.g., air, water, soil) and within a single stage in the life-cycle stage of a product. These do not always lead to an overall reduction in environmental impacts. Pollution control resources are spent on activities that are required by laws and regulations, and that do not always provide the most efficient use of those resources in terms of reducing impacts. This has often led to unintended consequences, for example, allowing one environmental problem to be solved while generating another. Because they are not designed to address a full understanding of the trade-offs and their implications in a systematic fashion, end-of-pipe approaches often diminish opportunities for achieving net environmental improvements.

Interest in life cycle management continues to grow. Our knowledge and understanding of LCA methodology and application has developed significantly over the past three decades. Further refinement of the methodology is needed but care should be taken to ensure that the pursuit does not end up being one of getting to more and more precise numbers. The life cycle thinking perspective has intrinsic value. As seen in the bio-fuels examples, different results have been presented but the overall message is the same – we must take care in how we acquire biofeedstocks and should be cautious in jumping to conclude that everything natural is automatically good. The future offers both significant challenges and opportunities for LCA and LCA-related approaches to play important decision-making support roles.

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